Chapter 6
Global Cost of Land Degradation

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Abstract Land degradation—defined by the Millennium Ecosystem Assessment report as the long-term loss of ecosystems services—is a global problem, negatively affecting the livelihoods and food security of billions of people. Intensifying efforts, mobilizing more investments and strengthening the policy commitment for addressing land degradation at the global level needs to be supported by a careful evaluation of the costs and benefits of action versus costs of inaction against land degradation. Consistent with the definition of land degradation, we adopt the Total Economic Value (TEV) approach to determine the costs of land degradation and use remote sensing data and global statistical databases in our analysis. The results show that the annual costs of land degradation due to land use and land cover change (LUCC) are about US$231 billion per year or about 0.41 % of the global GDP of US$56.49 trillion in 2007. Contrary to past global land degradation assessment studies, land degradation is severe in both tropical and temperate countries. However, the losses from LUCC are especially high in Sub-Saharan Africa, which accounts for 26 % of the total global costs of land degradation due to LUCC. However, the local tangible losses (mainly provisioning services) account only for 46 % of the total cost of land degradation and the rest of the cost is due to the losses of ecosystem services (ES) accruable largely to beneficiaries other than the local land users. These external ES losses include carbon sequestration, biodiversity, genetic information and cultural services. This implies that the global

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community bears the largest cost of land degradation, which suggests that efforts to address land degradation should be done bearing in mind that the global community, as a whole, incurs larger losses than the local communities experiencing land degradation. The cost of soil fertility mining due to using land degrading management practices on maize, rice and wheat is estimated to be about US$15 billion per year or 0.07 % of the global GDP. Though these results are based on a crop simulation approach that underestimates the impact of land degradation and covers only three crops, they reveal the high cost of land degradation for the production of the major food crops of the world. Our simulations also show that returns to investment in action against land degradation are twice larger than the cost of inaction in the first six years alone. Moreover, when one takes a 30-year planning horizon, the returns are five dollars per each dollar invested in action against land degradation. The opportunity cost accounts for the largest share of the cost of action against land degradation. This explains why land users, often basing their decisions in very short-time horizons, could degrade their lands even when they are aware of bigger longer-term losses that are incurred in the process.

Keywords Land degradation · Total economic value · Land use/cover change · Ecosystem services · Global cost

Introduction

Land degradation—defined as persistent or long-term loss of ecosystem services, has recently gained a more prominent attention in national and international agendas, especially after the food crisis in 2008 with spiking food and land prices (von Braun 2013) and higher demands for biofuels. The rising concern for sustainable development and poverty reduction has also contributed to increased attention to sustainable land management. Land degradation affects the poor the most since they heavily depend on natural resources. Despite the increasing need for addressing land degradation, investments in sustainable land management remain limited—especially in low income countries. An FAO study on agricultural investment showed a declining public investment in agricultural sector in Sub-Saharan Africa (SSA) over the past three decades (FAO 2012), with the public expenditure per worker declining from US$152 in 1980–89 to only US$42 in 2005–07 (ibid).

As part of efforts to raise awareness of the cost of inaction against land degradation, this study is conducted to determine the cost of land degradation across regions and globally. The study makes new contributions to literature by adopting the Millennium Ecosystem Assessment (MEA 2005) definition of land degradation and, therefore, using the Total Economic Value (TEV) approach to determine the value of land degradation (see Nkonya et al. 2013).

This study contributes to literature significantly as it develops analytical methods that use TEV approaches and data that are easily available to allow regular economic assessment of land degradation and improvement. The analytical methods are presented in a simplified language to allow application across disciplines and different analytical skill levels of economics and ecology. The study also covers two major forms of land degradation—namely loss of value of ecosystem services due to land use change/cover (LUCC) of six major biomes and use of land degrading management practices on cropland and grazing lands that do not experience LUCC. The six major biomes include forest, shrublands, grasslands, cropland, bare land, and woodlands and they accounted for about 86 % of land area in 2001 (NASA 2014).

Even though this study uses TEV to reflect the broader concept of land degradation and includes six biomes, it does not comprehensively cover all forms of land degradation. We do not cover some forms of environmental degradation—such as over-application of fertilizers or agrochemicals that lead to eutrophication. We also do not cover degradation of forests, grasslands, shrublands and woodlands that did not experience LUCC. Additionally our study does not consider loss of wetlands—a biome that covers 550 million ha (Spiers 2001), which is about 4 % of global land area. This is due to lack of proper data to analyze loss of wetlands.

Our study does not analyze the impact land degradation on consumers of food, feed, etc. Our study also does not analyze indirect impacts of land degradation such as the increasing prices of land, migration, etc. These omissions are necessary to make the study tractable. Other studies could be commissioned to cover these gaps.

This paper is organized as follows. The next section discusses past studies on the costs of land degradation at regional or global levels. This is followed by a description of the analytical methods and data used in this study. The results section follows and the last concludes with policy implications.

Previous Global Studies on the Costs of Land Degradation

A number of studies have estimated the costs of land degradation at the global level. It is not our aim to conduct a comprehensive review of such studies, rather our objective is to highlight the different estimation methods and consequent wide variation of findings on the global costs of land degradation. The 12 studies reviewed are summarized in Table 6.1. The costs of land degradation range from US$17.58 billion to as high as US$9.4 trillion (both at 2007 values). Two major reasons explain the large variation of these estimates. First, the studies use different methodological approaches. Secondly, some studies evaluate only few biomes while others are more comprehensive and cover all major biomes. Dregne and Chou (1992) were among the earliest to evaluate the global costs of land degradation. Using a loss of productivity approach, they estimated that the global cost of cropland and grassland degradation in 1990 at US$43 billion. A more recent study, based on literature review, estimated the cost of land degradation to be about US $450 billion per year (UNCCD 2013). Using loss of carbon sink as an indicator of

Table 6.1 Global costs of land degradation of past studies

Author(s)	Annual cost reported (US$ billion)	Equiv. annual cost in 2007 US$ billion	Comments
FAO (2007)	40	40.00	Methods not reported
UNCCD (2013)	490	685.40	Review of literature
Trivedi et al. (2008)	43–65	41.4–62.6	Loss of carbon sink due to deforestation of tropical rainforests
Dregne and Chou (1992)	43	54.69	Loss of productivity of cropland and grassland
Basson (2010)	21	20.27	Off-site cost of soil erosion: (i) reduced water storage structures, with the replacement costs of silted-up reservoirs (ii) loss of hydroelectric power (HEP) and damage to HEP infrastructure (iii) reduction of irrigation reservoir
Myers et al. (2000)	300	361.15	Cost of protection of biodiversity loss
Costanza et al. (2014)	9400	9400.00	Benefit transfer approach to estimate the Total economic (TEV) of ecosystem services. Cost of terrestrial land degradation computed as net loss/gain of value of ecosystem services of terrestrial biomes
Trutcost (2014)	6900	6900.00	Literature review and government studies and stylized environmental evaluation methods of environmental pollution
Dodds et al. (2013)	900	800.73	Anthropogenic degradation of freshwater ecosystem services
Chiabai et al. (2011)	261[a]	277.07	Simulation using IMAGE 2.4 model of net present value of forest ecosystem services, 2000–2050

[a]Lower bound of the estimate increase in value of ecosystem services equal to US$61

land degradation, Trivedi et al. (2008) estimated the global cost of deforestation of tropical forests and rainforests was about US$43–65 billion. The cost of avoiding degradation could also be used to measure the cost of land degradation (Requier-Desjardins et al. 2011). Accordingly, Myers et al. (2000) estimated the cost of avoiding the loss of biodiversity to be about US$300 billion. Using replacement costs of silted up reservoirs, loss of hydroelectric power and reduction in irrigated production, Basson (2010) estimated the annual global cost of siltation of water reservoirs to be about $18.5 billion.

A more recent study by Costanza et al. (2014) uses the total economic value approach and estimated the net cost of terrestrial ecosystem services to be about US $9.9 trillion. As shown in Fig. 6.1, a large share of the loss of terrestrial ecosystems in this study came from wetlands degradation.

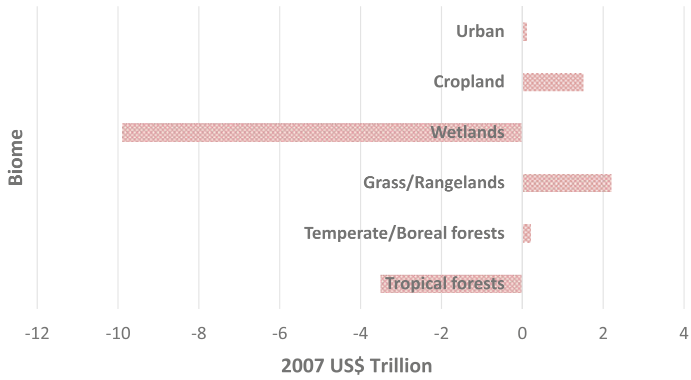


Fig. 6.1 Global value of change of ecosystem services, 1997–2007. *Source* Computed from Costanza et al. (2014)

The net loss of terrestrial ecosystem services is about US$9.4 trillion but the gross loss is US$13.4 of which wetlands loss accounts for 74 % and the remaining loss is accounted for by tropical forests. The other terrestrial biomes included in the study registered gains.

Unlike Costanza et al. (2014), Trucost (2014) directly estimated the environmental impacts of economic activities. Specifically, the environmental impacts were measured by the cost of land use, greenhouse gas emission, water consumption and air pollution. The direct measurement of environmental pollution by companies is a significant contribution of the Trucost (2014) study.

The review above shows that the costs of land degradation include a wide range of costs, an aspect which implies the difficulty of achieving a consensus on one specific costs estimate. As argued by Nkonya et al. (2013), this study approach bears in mind the data availability at the global level and the key elements that need to be taken into account in any global ELD assessment. A standardized procedure could, thus, allow the comparison of ELD values across studies.

To lay ground for the methodological approaches used in this study, the following section discusses the land use types and their major characteristics.

Land Use Types and Their Characteristics

We discuss the terrestrial land use types used in this analysis, highlighting their extent and importance across regions. We focus on seven major terrestrial land use types, namely forests, shrublands, grasslands, cropland, woodlands, urban and bare or barren lands.

Definition and Classification of Terrestrial Biomes and Land Use/Cover Types

There is a number of definition and classification of biomes that reflect the scientists' area of emphasis (McGinley 2014). For example, FAO defines forest as an area with a minimum coverage of 1 ha, with at least 10 % crown cover and with mature trees at least 2 m tall (FAO 2011). The definition explicitly includes open woodlands, such as those found in the African Sahel. This differs from the International Geosphere-biosphere Programme (IGBP) definition, in which a forest is an area with 60 % tree canopy coverage (Table 6.2). Miller (1990) includes shrublands in grasslands while IGBP assigns shrublands a separate biome. In this study we use the IGBP definitions since the MODIS data used are defined according to IGPBP.

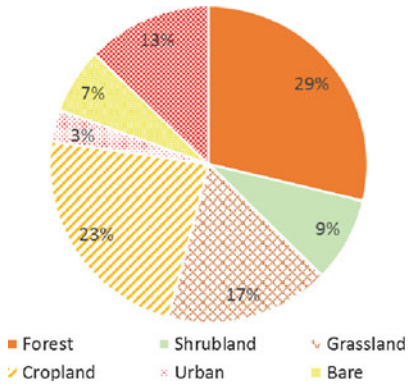
The seven major terrestrial biomes covered in study account for about 86 % of the global land area in 2001. The rest of the area was covered by inland water bodies and wetlands. Wetlands cover less than 5 % of Earth's ice-free land surface (NASA 2014), but they play a key role in carbon and water cycles. For example, Costanza et al. (2014) estimated the cost of wetlands degradation to be about 2007 US$9.4 trillion/year or 50 % of the total annual cost of loss of terrestrial and marine ecosystem services estimated at 2007 US$20.2 trillion. However, we focus our analysis on the seven major biomes mentioned above.

Table 6.2 defines each biome while Fig. 6.2 reports the global extent of each biome in 2001 at the global level. The table below Fig. 6.2 reports the corresponding extent of each biome at region level. We use the Moderate Resolution

Table 6.2 Definition of biomes used in the study

Biome	IGBP definition
Forests	Woody vegetation with height >2 m and covering at least 60 % of land area. Forest trees divided into three categories: (i) Deciduous Broadleaf—broadleaf trees that shed leaves in annual cycles. (ii) Deciduous Needleleaf—as deciduous broadleaf but with narrow leaves. (iii) Evergreen Broadleaf Forests—broadleaf trees that remain green foliage throughout the year. (iv) needleleaf evergreen—like evergreen broadleaf but with narrow leaves
Grassland	Lands with herbaceous types of cover. Tree and shrub cover is less than 10 %
Cropland	Lands covered with temporary crops followed by harvest and a bare soil period (e.g., single and multiple cropping systems). Note, perennial woody crops are classified as forest or shrubland
Bare	Barren or Sparsely Vegetated (Bare Soil and Rocks). Lands with exposed soil, sand or rocks, with less than 10 % vegetated cover throughout the year
Shrublands	Vegetation with mainly shrubs or short trees (shrubs) of less than 2 m. Canopy of shrublands is fairly open and allows grasses and other short plants grow between the shrubs
Woodland	Biome with tree cover of 5–10 %, with trees reaching a height of 5 m at maturity

For more definitions, please see http://earthobservatory.nasa.gov/Experiments/Biome/vocabulary.php



Region	Forest	Shrubland	Grassland	Cropland	Urban	Bare	Woodlands
	Area as percent of total land area						
SSA	21.88	11.21	20.21	18.1	2.58	11.41	14.6
LAC	37.36	8.79	22.48	14.66	5.46	3.16	8.11
NAM	39.16	5.94	8.37	36.11	2.61	0.93	6.89
East Asia	35.7	5.28	9.54	18.37	9.65	4.31	17.14
Oceania	37.4	15.9	17.96	15.19	0.19	3.72	9.65
South Asia	48.51	4.15	5.9	23.08	2.59	2.88	12.9
SE Asia	46.82	0.99	5.22	27.84	2.34	0.95	15.83
East Europe	28.83	4.45	7.85	50.31	2.88	0.63	5.06
West Europe	26	6	21.78	27.44	4.65	1.28	12.85
Global	28.72	8.62	16.75	23.22	3.26	6.69	12.74

Fig. 6.2 Extent of the major terrestrial biomes, 2001. *Note SSA* Sub-Saharan Africa; *LAC* Latin American Countries; *NAM* North America; *SE* South-east. See Appendix for countries in each region. *Source* Calculated from MODIS data

Imaging Spectroradiometer (MODIS) landcover data to analyze the land use and land cover change (LUCC). MODIS data are collected by NASA's two satellites (Terra (EOS AM) and Aqua (EOS PM)) and have three levels of resolutions (250, 500, and 1000 m) (NASA 2014) and were launched in December 1999. For our study we use the 1-km resolution that matches the International Geosphere-Biosphere Program (IGBP) land cover classification. The data include a much greater number of inputs (7 wavelengths, or "bands") as well as the enhanced minimum and maximum annual values of vegetation index, land surface temperature. The MODIS data are quality controlled and ground-truthed (Friedl et al. 2010). The overall accuracy of land use classification is about 75 % (Friedl et al. 2010). As will be discussed below, LUCC will be used as one form of land degradation or improvement.

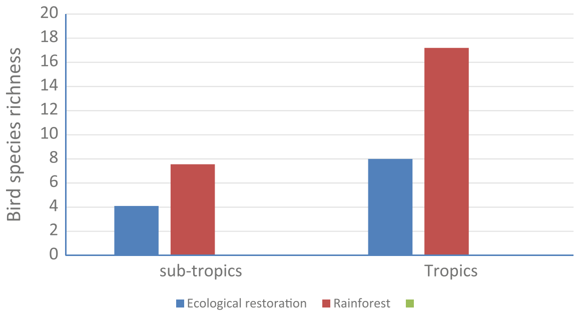


Fig. 6.3 Species bird richness in ecological restoration 10-year trees versus primary rainforest. *Source* Computed from Cateral et al. (2004)

Forest

The forests serve as the biggest terrestrial carbon sink as they store about 861 petragrams of Carbon (PgC) (Pan et al. 2011), which is about half the global terrestrial global carbon stock (FAO 2013). However, due to different definitions of forest by FAO and IGBP, the extent and land use change reported in this study could differ from those reported by FAO. Our analysis will look at the change in forest extent as land degradation/improvement even though other forms of land degradation or improvement may happen through changes of forest density. In the past two decades (1990–2010), global forest density—tree density per hectare—increased (Rautiainen et al. 2011). The increase was most pronounced in North America and Europe and the increase in Africa and South America was only modest. In Asia, forest density increased in 1990–2000 but decreased in 2000–2010 (ibid).

Loss and gain in biodiversity is another important aspect that changes as forest LUCC occurs. Unfortunately, biodiversity builds over many years and cannot be fully restored through reforestation and afforestation programs (CBD 2010). Newly planted forests have fewer tree species and lower fauna and flora biodiversity (Ibid). For example, a study of ecological restoration through replanting of rainforest in Australia showed that birds richness in planted rainforest was only about half of their reference rainforest (Fig. 6.3).

Grassland

According to the MODIS data, grassland covers 17 % of the land area (Fig. 6.2), but grassland could also include shrublands (Miller 1990; FAO 2010). Using the broader definition of grassland, including subtropical deserts,[1] grasslands, tundra,

[1]Subtropical deserts differ from bare deserts since they have vegetation with strong moisture and water conservation mechanisms, which are well-adapted to the low precipitation.

Table 6.3 Land area of grassland (million km^2)

Regions	Savanna	Shrublands	Non-wood grassland	Tundra	Global
Asia (excl NENA)	0.9	3.76	4.03	0.21	8.89
Europe	1.83	0.49	0.7	3.93	6.96
NENA	0.17	2.11	0.57	0.02	2.87
SSA	10.33	2.35	1.79	0	14.46
NAM	0.32	2.02	1.22	3.02	6.58
CAC	0.3	0.44	0.3	0	1.05
South-America	1.57	1.4	1.63	0.26	4.87
Oceania	2.45	3.91	0.5	0	6.86
World	17.87	16.48	10.74	7.44	52.53

Note CAC Central American and Caribbean; *NENA* Near East and North Africa. *Source* White et al. (2000)

woodlands and shrublands (Miller 1990), it is estimated that the biome cover 5 billion ha or 40 % of global land area and store about 30 % of carbon stock (Tennigkeit and Wilkies 2008) Grasslands, account for 70 % of the global agricultural area, and about 20 % of the soil carbon stocks (Ramankutty et al. 2008). However, not all grasslands are used for livestock production. FAO (2012) estimates that 26 % of the ice-free land area is used for livestock production, supporting about one billion people, mostly pastoralists in South Asia and SSA. Livestock provides about a quarter of protein intake and 15 % of dietary energy by global human population (Ibid). Table 6.3 reports the distribution of grassland across regions in 2000.

Shrublands and Woodlands

We discuss shrublands and woodlands together, similarly to previous literature (e.g. see MEA 2005). The major difference between them is the tree height. Shrublands are covered by shorter trees (shrubs) and woodland is a biome with tree cover of 5–10 %—with trees reaching 5 meters height at maturity (FAO 2010). Shrublands account for 9 % of the global land area while woodlands cover about 13 % of the land area (Fig. 6.2). Shrublands and woodlands serve as pasture and provide many other forms of ecosystem services (MEA 2005).

Cropland

According to the MODIS data used in this study, cropland is the second largest biome as it covers 23 % of land area (Fig. 6.2). Extent of cropland area in 1992–2009 decreased by 0.3 % but increased by 4 % in SSA—the largest increase in the world (FAOSTAT 2014; Foley et al. 2011). Consequently SSA experienced the highest deforestation rate in the world (Gibbs et al. 2010). Cropland mainly provide provisioning services though it also provides regulating and cultural services, supporting services, regulation of water and climate systems and aesthetic services (Swinton et al. 2007).

Bare Lands

Covering about 7 % of the land area, bare land has exposed soil, sand or rocks, with less than 10 % vegetative cover throughout the year. This includes the deserts and degraded lands. This also includes the Polar Regions permanently covered with snow or ice. In our LUCC analysis, the bare biome analysis will focus on bare land that could have been affected by anthropogenic changes and will exclude Polar Regions and other uninhabited areas.

Urban

The urban areas have been expanding rapidly in the past few decades, covering 3 % of the global land area in 2001 (Foley et al. 2005). For the first time, the urban population surpassed the rural population in 2009 (UN 2010). We do not include the urban areas in ecosystem services valuation due to their complex nature.

Analytical Approach

We use the Total Economic Value (TEV) approach, which assigns value to both tradable and non-tradable ecosystem services. There is a considerable debate on the usefulness of the TEV approach (e.g. see a review by Nijkamp et al. 2008; Seppelt et al. 2011). Given the complex nature of ecosystem services, double-counting is a major problem of TEV approach (Balmford et al. 2008). Another problem is assigning value to non-tradable ecosystem services. For example an attempt to assign value to some of the ecosystem services—e.g. the air we breathe—could be futile as such resources may not be amenable to valuation and could put unnecessary cost burden on producers. For example, Trucost (2014) evaluated the global social cost of loss of ecosystem services to be about US$4.7 trillion per year and concluded that the top 20 production sectors that lead in ecosystem services degradation would not make profit if they took into account the lost ecosystem services.[2] Despite this, there is a strong realization of the importance of using the broader MEA (2005) definition of land degradation and this justifies the use of the TEV approach to determine the cost of land degradation. Our approach uses methods that avoid double counting or assigning values that may be contestable.

We divide the causes of land degradation into two major groups and evaluate the cost for each:

1. Loss of ecosystem services can be due to LUCC that replaces biomes that have higher ecosystem value with those that have lower value. For example, change from one hectare of forest to one hectare of cropland could lead to loss of ecosystem services since the TEV of a forest is usually higher than the value of cropland. We focus on five major land use types: cropland, grassland, forest, woodland, shrublands and barren land. Even though Costanza et al. (2014) report that wetlands degradation accounts for about 50 % of total annual land degradation, we do not include wetlands because of their small extent (5 %) and limited data availability.
2. Using land degrading management practices on a static land use, i.e. land use did not change from the baseline to endline period. Due to lack of data and other constraints, we focus on cropland and livestock only.

We focus on anthropogenic land degradation, but due to the lack of relevant TEV data, we use a value transfer approach, which assigns ES values from existing case studies to ES valuation in other areas with comparable ES (Desvousges et al. 1998; Troy and Wilson 2006). The value transfer approach has its weaknesses (e.g. see Defra 2010), but lack of data makes it the only feasible approach for global or regional studies.

Land Degradation Due to LUCC

The cost of land degradation due to LUCC is given by

$$C_{LUCC} = \sum_{i}^{K} (\Delta a_1 * p_1 - \Delta a_1 * p_2) \tag{6.1}$$

[2]Coal power generation (Eastern Asia); Cattle ranching (South America); coal power generation (North America); Wheat farming (Southern Asia); Rice farming (Southern Asia); Iron and steel mills (Eastern Asia); Cattle ranching (Southern Asia); Water supply (Southern Asia); Wheat farming (North Africa); Rice farming (Eastern Asia); Water supply (western Asia); Fishing (global); Rice farming (Northern Africa); Maize farming (Northern Africa); Rice farming (SE Asia); Water supply (Northern Africa); Sugar (Southern Asia); Natural gas extraction (Eastern Europe); and Natural gas generation (Northern America).

where CLUCC = cost of land degradation due to LUCC; a_1 = land area of biome 1 being replaced by biome 2; P_1 and P_2 are TEV biome 1 and 2, respectively, per unit of area.

By definition of land degradation, $P_1 > P_2$.

This means, LUCC that does not lead to lower TEV is not regarded as land degradation but rather as land improvement or restoration. To obtain the net loss of ecosystem value, the second term in the equation nets out the value of the biome 1 replacing the high value. i = biome i, i == 1, 2, … k.

The ecosystem services included in the TEV and their corresponding value are reported in Table 6.4. Discussion on how data were processed to avoid double-counting is done in the data section below.

Land Degradation Due to Use of Land Degrading Management Practices on a Static Cropland

The provisioning services of crops are well known and directly affect rural households. What is less known are the ecosystem services provided by cropland. One such service is carbon sequestration, which we measure in this study by comparing sequestration due to sustainable land management (SLM) with that arising from land degrading practices.

We use DSSAT-CENTURY (Decision Support System for Agrotechnology Transfer) crop simulation model (Gijsman et al. 2002) to determine the impact of SLM practices on crop yield and soil carbon. Among the most widely used crop models globally, DSSAT employs a process-based approach to model the growth of crops and their interaction with soils, climate, and management practices. DSSAT combines crop, soil, and weather databases for access by a suite of crop models enclosed under one system. When calibrated to local environmental conditions, crop models can help understand the current status of farming systems and test hypothetical scenarios. DSSAT model was modified by incorporating a soil organic matter and residue module from the CENTURY model. The combined DSSAT-CENTURY model used in this study was designed to be more suitable for simulating low-input cropping systems and conducting long-term sustainability analyses.

DSSAT has been calibrated using many experiments around the world. However, the DSSAT and other process-based models have a number of disadvantages as reported by Lobell and Burke (2010). Process-based crop models give point estimates and do not include all relevant biological processes. For example DSSAT cannot simulate the effect of salinity, soil erosion, phosphorus, potassium, intercropping and other processes that could affect yield. As a part of efforts to address these disadvantages, we also estimate empirical models that are based on previous studies. The empirical models incorporate the effect of salinity and soil

Table 6.4 Terrestrial ecosystem services and their global average value (2007 US$/ha/year)

Ecosystem services	Inland wetlands	Tropical forest	Temperate forest	Woodlands	Grasslands
Provisioning services	1659	1828	671	253	1305
Food	614	200	299	52	1192
Water	408	27	191		60
Raw materials	425	84	181	170	53
Genetic resources		13			
Medicinal resources	99	1504			1
Ornamental resources	114			32	
Regulating services	17,364	2529	491	51	159
Air quality regulation		12			
Climate regulation	488	2044	152	7	40
Disturbance moderation	2986	66			
Regulation of water flows	5606	342			
Waste treatment	3015	6	7		75
Erosion prevention	2607	15	5	13	44
Nutrient cycling	1713	3	93		
Pollination		30		31	
Biological control	948	11	235		
Habitat services	2455	39	862	1277	1214
Nursery service	1287	16		1273	
Genetic diversity	1168	23	862	3	1214
Cultural services	4203	867	990	7	193
Esthetic information	1292				167
Recreation	2211	867	989	7	26
Inspiration	700				
Cognitive development			1		
Total economic value	25,682	5264	3013	1588	2871

Extracted from Groote et al. (2012)

erosion (Nkonya et al. 2013). To capture the long-term impacts of land management practices, the DSSAT model will be run for 40 years.

We use two crop simulation scenarios:

1. SLM practices are the combination of organic inputs and inorganic fertilizer. Integrated soil fertility management (ISFM)—combined use of organic inputs, judicious amount of chemical fertilizer and improved seeds (Vanlauwe and

Giller 2006) is considered an SLM practice. Long-term soil fertility experiments have shown that ISFM performs better than the use of fertilizer or organic input alone (Vanlauwe and Giller 2006; Nandwa and Bekunda 1998).

2. Business as usual (BAU). The BAU scenario reflects the current management practices practiced by majority of farmers. These could be land degrading management practices or those which are not significantly different from the performance of ISFM.

Long-term soil fertility experiments have shown that even when using ISFM at recommended levels, yields decline due to decrease of soil organic matter (Nandwa and Bekunda 1998). This is also an indication of land degradation that will be taken into account as shown below.

$$\mathrm{CLD} = \left(y^c - y^d\right)P * (A - A^c) + \left(y^c_{t=1} - y^c_{t=40}\right) * A^c\big)P - \tau\Delta\mathrm{CO}_2 \tag{6.2}$$

where CLD = cost of land degradation on cropland, y_c = average yield with ISFM in the 10 years, y_d = average yield with BAU in the last 10 years, A = total area that remained under in baseline and endline periods, A_c = cropland area under ISFM. P = price of crop i; $y^c_{t=1}, y^c_{t=40}$ are average yield under ISFM in in the first 10 years and last 10 years respectively; ΔCO_2 = change in the amount of carbon sequestered under SLM and BAU and τ = price of CO_2 in the global carbon market.

We compute the net carbon sequestration after considering the amount of CO_2 emission from nitrogen fertilization and from manure application. One kilogram of nutrient nitrogen requires about 77.5 MJ for its production using the Haber-Bosch process, packaging, transportation, distribution, and application (Stout 1990). Of the 3553 PJ energy used in agriculture in 1998, nitrogen alone accounted for 64 % of the energy. The remaining energy in agriculture was used by (with their percent contribution in brackets) farm machinery (26 %), irrigation pumps (3 %) and pesticides (1 %) (Vlek et al. 2004).

We focus on three major crops: maize, rice and wheat, which cover about 42 % of cropland in the world (FAOSTAT 2014). The three crops also consume the largest share of fertilizer use in all regions (Table 6.5).

Table 6.5 Fertilizer use by the three most important crops in the world

Region	Maize	Rice	Wheat	Total
	% of total consumption of N, P and K			
SSA	26	8	7	41
LAC	25	6	8	39
South Asia	2	32	23	56
SE Asia	8	51	0	58
NENA	7	3	37	46
Global	17	17	22	55

Notes: *SSA* Sub-Saharan Africa; *LAC* Latin America, *SE* South-east, *NENA* Near East and North Africa
Source FAO (2006)

DSSAT will simulate maize, rice and wheat yields at a half degree resolution, i.e., about 60 km.

Land Degradation on Static Grazing Land

We use methods discussed in Chap. 8 and for brevity, we only summarize the discussion in this chapter. The models used to determine cost of land degradation is:

$$\begin{aligned} CLD_m &= \sum_{i=1}^{I} [DMI_{t=2001} - DMI_{t=2010}]\theta_m x_t P_m \\ DMI_t &= biom_t \gamma \kappa \end{aligned} \tag{6.3}$$

where DMI_t = dry matter intake (tons) in year t in pixel i; $\boldsymbol{\theta_m}$ = Conversion factor of grass DMI to the fresh weight of milk; P_m = price of milk per ton; biomt = grass biomass production (DM) in year t; γ = contribution of grass to total feed intake; xt = number of milking cows in pixel i; and $\boldsymbol{\kappa}$ = share of above ground grass biomass actually consumed by livestock.

Likewise, the loss of meat production due to land degradation (CLD_b) is given by

$$CLD_b = [DMI_{t=2001} - DMI_{t=2010}]\theta_b x_t \tau_t P_b, \tag{6.4}$$

where P_b = price of meat per ton; θ_b = conversion factor of grass DMI to the fresh weight of meat; τ_t = off-take rate; other variables are as defined above.

The total cost of static grassland degradation (LLD) is given by:

$$\mathrm{LLD} = \mathrm{CLD_m} + \mathrm{CLD_b}. \tag{6.5}$$

We only consider on-farm losses including milk production and off-take rate for meat and ignore the loss of live weight of livestock not slaughtered or sold since such loss is not liquidated and eventually affects human welfare. Due to lack of data, we also ignore the impact of degradation on livestock health, parturition, and mortality rates as well as loss of carbon sequestration and other environmental and ecological services provided by grasslands. This results in conservative estimates.

Total Cost of Land Degradation

We combine the total cost of land degradation from LUCC and from static land use as follows:

$$TCLD = \sum_{i}^{H} [CLD + LLD] + C_{LUCC}, \tag{6.6}$$

where TCLD = total cost of land degradation; CLUCC is cost of land degradation from LUCC; H = number of crops considered, H = 1, 2, 3, 4 (see Table 6.5).

Other variables are as defined in equation in (6.1)–(6.5). We will express the total land degradation per year basis and assume that the rate of land degradation is linear. Hence the annual cost of land degradation will be expressed as:

$$TCLD_a = \frac{TCLD}{T}. \tag{6.7}$$

where $TCLD_a$ = annual cost of land degradation; T = time from baseline to endline period. It should be noted that the annual cost of land degradation increases cumulatively as extent of land degradation increases. Thus, $TCLD_a$ reflects the long-term average—as stated in the definition of land degradation.

Cost of Taking Action Against Land Degradation

The approach for determining the cost of action for degradation due to LUCC has to consider the cost of reestablishing the high value biome lost and the opportunity cost of foregoing the benefits drawn from the lower value biome that is being replaced (Torres et al. 2010). For example, if a forest were replaced with cropland, the cost of planting trees or allowing natural regeneration (if still feasible) and cost of maintaining the new plantation or protecting the trees until they reach maturity has to be taken into account. Additionally, the opportunity cost of the crops being foregone to replant trees or allow natural regeneration has to be taken into account. This means the cost of taking action against land degradation due to LUCC is given by

$$CTA_i = A_i \frac{1}{\rho^t} \left\{ z_i + \sum_{t=1}^{T} (x_i + p_j x_j) \right\}. \tag{6.8}$$

where CTAi = cost of restoring high value biome i; ρt = land user's discount factor; Ai = area of high value biome i that was replaced by low value biome j; z_i = cost of establishing high value biome i per ha; xi = maintenance cost of high value biome i per ha until it reaches biological maturity—i.e., the age at which biome is capable of reproducing and bearing seeds (hereafter referred to as maturity); x_j = productivity of low value biome j per hectare; p_j = price of low value biome j per unit (e.g. ton); t = time in years and T = Land user's planning horizon. The term $p_j x_j$

represents the opportunity cost of foregoing production of the low value biome j being replaced.

The cost of inaction will be the sum of annual losses due to land degradation

$$CI_i = \sum_{t=1}^{T} C^i_{LUCC}, \tag{6.9}$$

where CI_i = cost of not taking action against degradation of biome i; C^i_{LUCC} is the cost of land degradation due to LUCC for biome i. Other variables are as defined in Eq. (6.1). As Nkonya et al. (2013) note, land users will take action against land degradation if $CTA_i < CI_i$.

The cost of action given in Eqs. 6.8 and 6.9 assumes all degradation effects are fully reversible but as discussed earlier, such assumption does not hold. For example, Fig. 6.3 shows that biodiversity of restored forests is lower than that of the natural forests. This is due to the loss of species habitat and biomes that take centuries to be restored. Given that the benefit of restoring degraded land goes beyond the maturity period of biome i, we have to use the land user's planning horizon to fully capture the entailing costs and benefits. Poor farmers tend to have shorter planning horizon while better off farmers tend to have longer planning horizon (Pannell et al. 2014). The planning horizon also depends on the type of investment. For example, tree planting requires longer planning horizon than annual cropland. For brevity however, we will assume a 30 year planning horizon for all the biomes considered.[3] Our assumption implies that during this time, farmers will not change their baseline production strategies dramatically. It is important to consider the biome establishment period since it has important implications on decision making. Poor land users are less likely to invest in restoration of high value biomes that take long time to mature. For example, trees take about 4–6 years to reach maturity (Wheelwright and Logan 2004). Given this we assume a 6 year maturity for trees. For grasslands, we assume a 2 year maturity age for natural regeneration or planting. The assumption is based on perennial grass like Rhodes grass (Chloris gayana), which reach full maturity after 2 years (Heuzé et al. 2015). Replanting is necessary if the LUCC involved excessive weeding of grass. Natural regeneration may take longer than 3 years but for simplicity we assume a three natural regeneration period.

As expected both the cost of action and inaction differ significantly across space and time. For example, reforestation costs are lower in low income regions than in high income countries (Benítez et al. 2007). However low government effectiveness and other challenges exist in low income countries and these could lead to even higher costs to maintain improvement. Our analysis will take into account such differences by using actual costs that have been observed in projects/programs in two major economic groups—high and low income countries.

[3]The 30 year planning period for land degradation due to LUCC should not be confused with the 40 year used in the crop simulation.

We also take into account the cost of land degradation across agroecological zones. For example, establishing a biome in a semi-arid area is more difficult than would be the case in humid and subhumid regions. Pender (2009) illustrate this using the survival rate of planted trees in the Niger, which was only 50 %. Other challenges also face farmers in arid and semi-arid areas (with annual average rainfall below 700 mm) when compared to land users in humid and subhumid areas (with annual precipitation above 700 mm) (IISD 1996). Hence for any given region, we assume that the cost of establishing any biome in arid and semi-arid areas is twice the corresponding cost in the humid and subhumid regions in the same economic group.

There are alternative land rehabilitation strategies available to land users. For example, action against deforestation could be taken using the traditional tree planting approach, which unfortunately is expensive but could achieve faster results. Assisted natural regeneration is also used and is cheaper than the conventional tree planting. For example, Bagong Pagasa Foundation (2011) found that the cost of the traditional replanting trees on deforested area was US$1079/ha compared to only US$579 for assisted natural regeneration. We will use the most common strategy in any given region and economic group.

Data

LUCC

Table 6.6 reports the extent of each biome in 2001 and the corresponding change in 2009. Figures 6.4 and 6.5 spatially report the corresponding changes. Extent of forest biome increased by almost 6 % globally with much of the increase occurring in temperate regions while almost all tropical regions experienced deforestation (Fig. 6.3). During the same period (2000–10), FAO (2011) reported an annual global deforestation rate of 0.1 %. As observed above, the disagreement between the MODIS land cover and FAO (2011) could be due to the differences in definition of forests.

While the extent of shrublands and cropland increased, the changes are quite different across different regions. The extent of cropland increased by 32 % in Oceania and by 12 % in SSA, but decreased in the Americas, Europe and SE Asia. Forest accounted for over 30 % of cropland expansion in Oceania and South Asia (Table 6.7). The source of cropland expansion in SSA was mainly shrublands and woodlands while forests accounted for only 19 % of cropland expansion (Table 6.7). This is contrary to Gibbs et al. (2010) who observed that forests contributed the largest share of crop expansion in SSA. Again, the difference could be explained by inclusion of woodlands and shrublands in the forest biome. MODIS data used in this study treats forest, shrublands and woodlands as separate biomes (Tables 6.2 and 6.6; Fig. 6.5).

Table 6.6 Land area of terrestrial biomes 2001 and change in 2009

Region	Forest	Shrubland	Grassland	Cropland	Bare	Woodlands
	Area of biome in 2001 (million ha)					
SSA	493.41	640.63	1402.09	300.99	2761.62	821.59
LAC	854.43	180.1	465.77	131.7	51.22	143.83
NAM	717.83	444.38	323.88	559.81	100.64	276.62
East Asia	442.56	137.32	305.29	327.95	302.69	547.9
Oceania	313.63	3230	2570.83	87.46	14.98	2044.14
South Asia	191.96	22.82	21.72	194.52	20.65	81.97
SE Asia	182.61	3.13	12.21	60.2	1.03	72.9
East Europe	586.77	510.75	165.96	310.86	15.89	268.3
West Europe	141.7	57	96.82	156.19	202.2	103
Total	3924.9	5226.12	5364.59	2129.69	3470.92	4360.23
	Change in area in 2009 as % of area in 2011					
SSA	1.15	−6.30	−2.08	−12.08	2.26	4.37
LAC	5.15	−2.41	−18.80	8.18	0.98	24.74
NAM	−18.79	1.56	4.38	13.50	1.12	11.82
East Asia	−5.27	45.97	−5.11	−12.14	7.99	−2.85
Oceania	8.17	−3.03	8.50	−32.67	−120.69	−5.00
South Asia	1.81	−6.35	−16.71	−2.18	15.98	3.11
SE Asia	7.65	−44.41	−4.34	9.52	63.11	−25.69
East Europe	−23.19	−7.43	42.44	2.60	−22.28	35.47
West Europe	−14.34	5.86	7.51	5.22	0.82	−1.59
Total	−5.65	−2.10	3.24	−0.03	2.08	1.46

Notes 1 % change in area = $\frac{a_1 - a_2}{a_1} * 100$

SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East

See Appendix for countries in each region

Source MODIS data

Total Economic Value Data

We derive the TEV from the economics of ecosystems and biodiversity (TEEB) database, which is based on more than 300 case studies—reporting more than 1350 ES values (de Groot et al. 2012). The spatial distribution of the terrestrial biome studies is shown in Fig. 6.6. Studies on coastal, coastal and inland wetlands, coral reefs, freshwater, and marine are excluded in accordance to our study's focus on the seven major terrestrial biomes. It is clear that the studies are well-distributed even in SSA. Areas with limited coverage include Russia, central Asia and NENA. However, there are few studies conducted in these regions that will serve as representative of the regions. Due to a large variation of the data source and methods used, data were standardized to ensure that the reported values are comparable. The criteria used for including studies were: the study has to be original, i.e., not based on literature review; reported value of ES value per ha for specific biome and

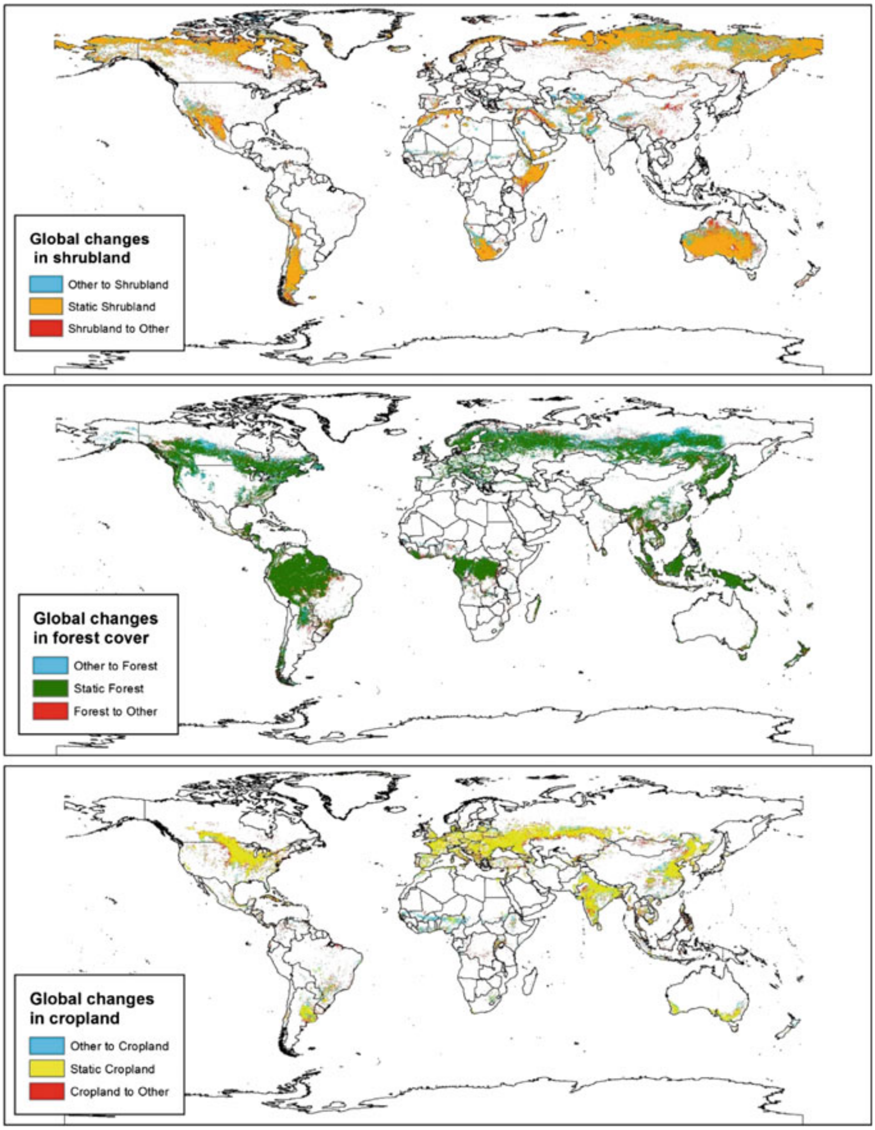


Fig. 6.4 Change of extent of shrubland, forest and cropland, 2001–09. *Source* Calculated from MODIS land cover data

specific time period, valuation method is included, and surface area studied is reported (de Groot et al. 2012). Only 665 of the 1350 case studies met these conditions (de Groot et al. 2012).

The data were converted to 2007 US$ to allow value comparison across time. One of the major weaknesses of the ES values included in the database was the

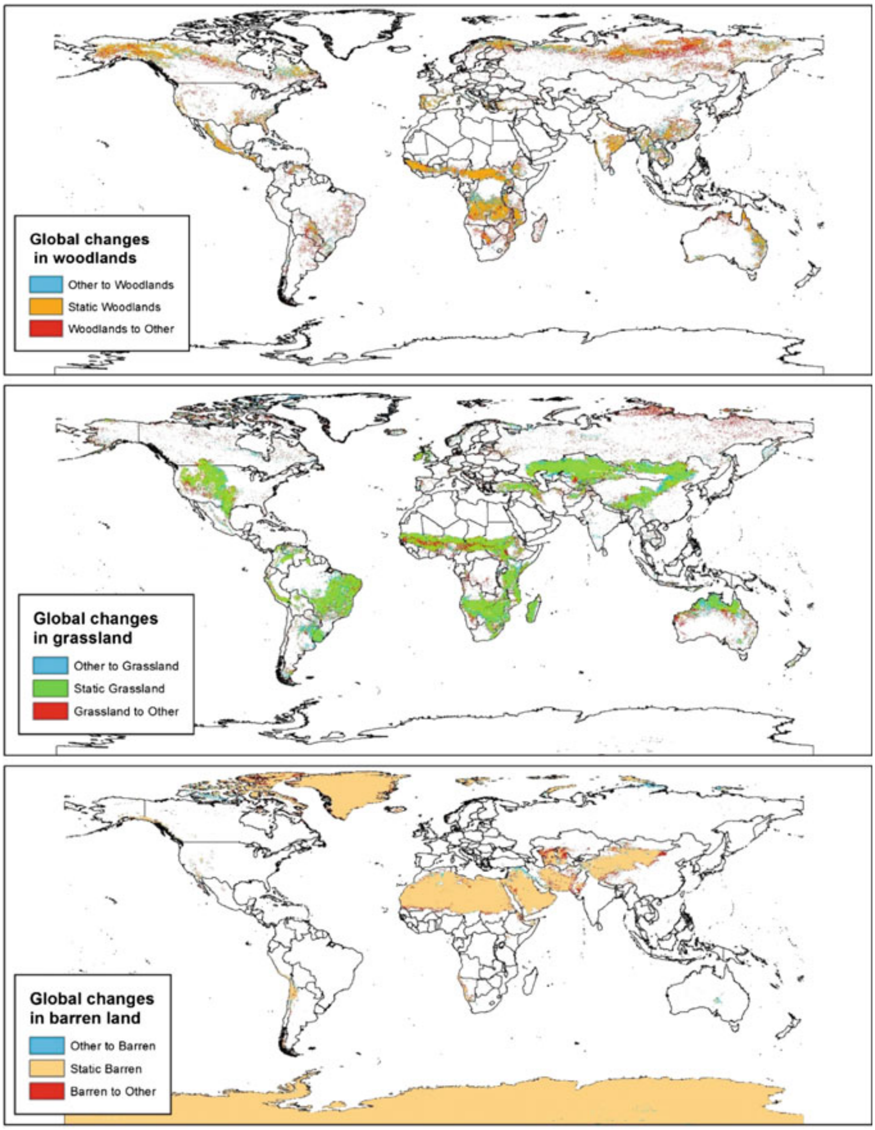


Fig. 6.5 Change of extent of woodlands, grassland and barren land, 2001–09. *Source* Calculated from MODIS land cover

wide variation of the ES values. For example value of tropical forests ranges from less than US$1 to US$9412/ha/year. Likewise, the value of grasslands varies from less than US$1 to US$ 6415/ha/year. De Groot et al. (2012) attribute the wide variation to five major reasons (i) locations attach different values to different biome ES (ii) different valuation methods were used but over 60 % used annual TEV

Table 6.7 Sources of cropland expansion

Source	SSA	East Asia	Oceania	South Asia
	Percent contribution			
Forest	19	17	36	36
Grassland	18	20	18	11
Shrubland	37	19	29	20
Bare	4	1	1	1
Woodlands	22	43	16	33

Note Includes regions that experienced cropland area expansion reported in Table 6.6

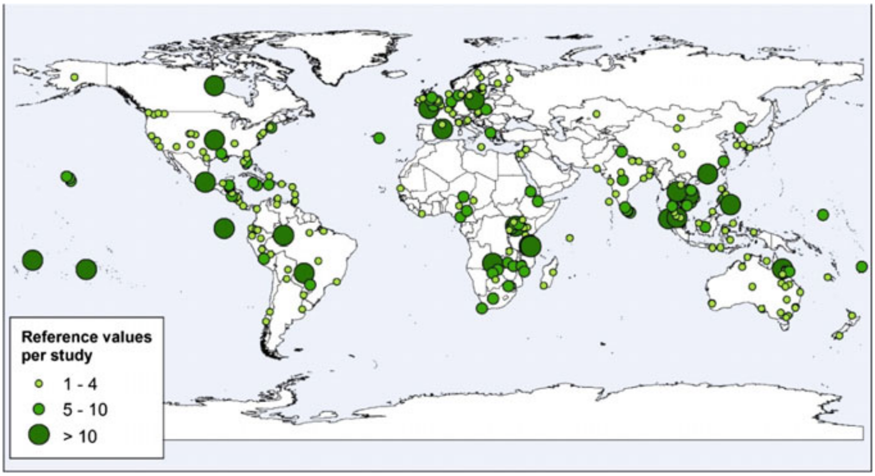


Fig. 6.6 Location of TEEB database of terrestrial ecosystem service valuation studies. *Source* Derived from TEEB database, the TEV of the five major biomes is shown below

(Table 6.8) (iii) different sub-biomes were considered in different studies (iv) attribution of ES values to different services, which could lead to double-counting when ES are aggregated and (v) ES values are time specific (e.g. see Costanza et al. 2014).

Additionally, most studies used did not exhaustively cover all ES and therefore the average values reported are conservative estimates of the total value (Ibid). To address this problem, we only included studies that used TEV.

TEV and Double-Counting Challenge

Double counting—i.e., assigning value of an ecosystem service at two different stages of the same process providing human welfare is a common problem in ecosystem valuation using TEV approaches. The potential for double-counting is

Table 6.8 Analytical methods of terrestrial biome ES evaluation

Analytical method	# of studies	%
Annual	827	63.5
Benefit Transfer	165	12.7
Direct market pricing	100	7.7
Net Present Value	56	4.3
Total Economic Value	46	3.5
Contingent Valuation	25	1.9
Avoided Cost	21	1.6
Replacement Cost	20	1.5
Others	42	3.2
Total	1302	100.0

Others include: Capital/stock value, factor income/production function, group valuation, hedonic pricing, marginal value, mitigation and restoration cost, one time payment/WTP, PES and present value
Source Compiled from TEEB database

hard to completely rule out due to the complex interlinkages of ecosystem services and processes (Fu et al. 2011). For instance if there are pollination services value of forest (or other biomes) these are certainly reflected in the value of crop harvests and hence adding them up is a double counting. The same applies to nutrient cycling, disease and climate regulation, flood and erosion regulation, etc. The potential for double-counting leads to overestimation of the cost of land degradation.

de Groot et al. (2012) use different standardization methods to address these issues. These include assigning value to final products of regulating and supporting services (Fisher et al. 2008). Other measures used to avoid and/or reduce double-counting include: Use case studies with consistent ES classification systems and selecting annual TEV valuation methods which are widely used in the ES literature (Fu et al. 2011).

Comparison of TEV of Biomes Across Studies and with Conventional GDP

Comparison of the TEEB average ES values with Chiabai et al. (2011) and CBD (2001)—both of which are global studies—reveal that TEEB average values are lower (e.g. see Fig. 6.7). Chiabai et al. (2011) value of tropical forests is about 10,000/ha/year compared to about US$5000 for TEEB and US$6000 for CBD value (Fig. 6.7). TEEB's value for temperate forests is the highest however but comparable to the value reported by Chiabai et al. (2011). Hence even though we believe that the values used are conservative, the values should be interpreted with these differences in mind.

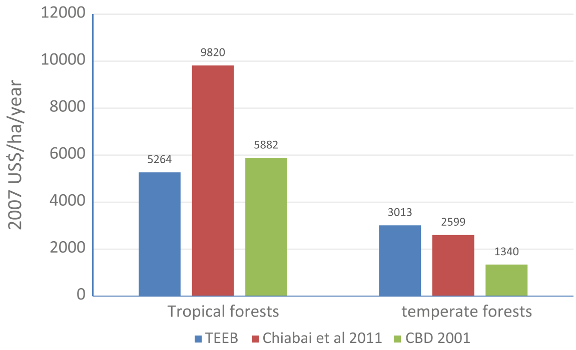


Fig. 6.7 Comparison of TEV of tropical and temperate forests across three studies. *Source* Computed from CBD (2001), Chiabai et al. (2011), de Groot et al. (2012)

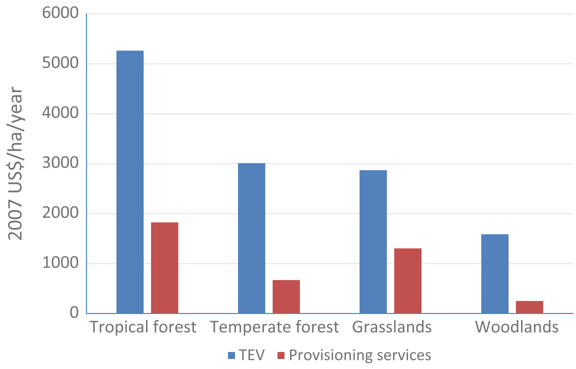


Fig. 6.8 TEV of major biomes. *Source* de Groot et al. (2012)

Figure 6.8 reports the average TEV of the major terrestrial biomes. Figure 6.8 also reports the corresponding value of provisioning services to reflect the traditional assessment of cost of land degradation that considered only provisioning services. In all cases, the TEV is more than twice the corresponding value of provisioning services.

We compare the ecosystem value endowment and the corresponding GDP per capita of each country to reflect the large differences between the traditional valuation methods that only takes into account tangible marketable services and the TEV approach. It can be easily seen that countries considered among the poorest have equivalent or greater TEV than high income countries (Figs. 6.9 and 6.10). For example, if TEV were used to group countries in three “income” groups, majority of SSA countries could be regarded as “middle-income” countries while majority of West European countries would fall in the “low-income” countries. North America, China, Russia, Australia and Brazil would fall in the “high income countries” largely due to their large land area and rich endowment of high-value biomes—

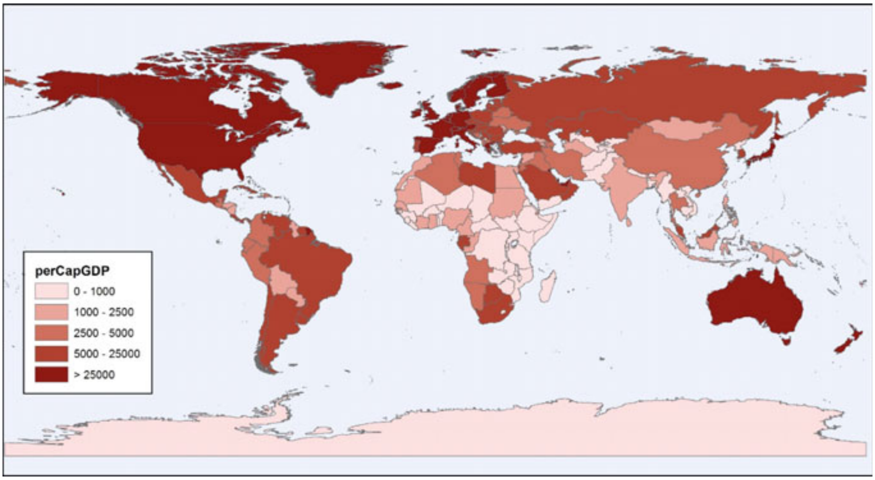


Fig. 6.9 Gross domestic product per capita, 2007 US$

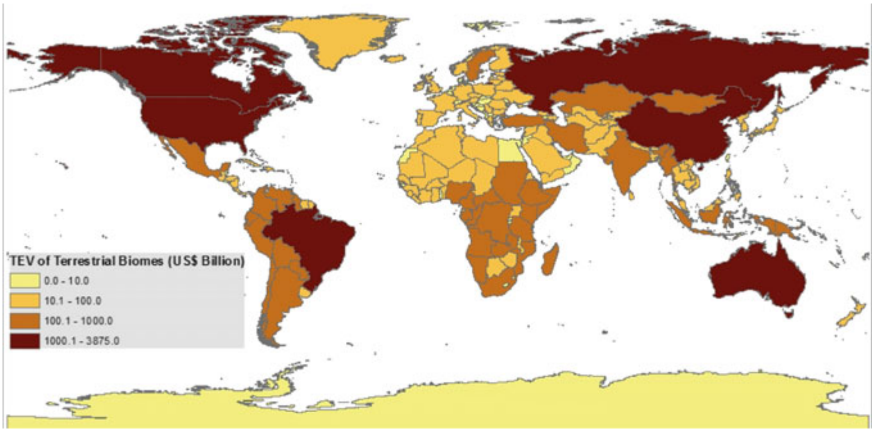


Fig. 6.10 TEV endowment at country level in 2001

namely forest or grasslands. Taking population into account but dropping countries577 with fewer than one million people, only three countries (Australia, Canada and Russia) classified as high income countries are among the top 12 countries with highest per capita TEV of terrestrial biomes and the rest in list are low income countries with sparse population (Table 6.9). However, given that a large share of the TEV benefits of ecosystems cannot be internalized in the resident country, such endowment does not reflect the welfare of the people in the country or community around the biome. Never-the-less, the spatial distribution helps to determine where the world needs to concentrate its effort to protect ecosystem services.

Table 6.9 Top 12 countries with highest per capita TEV of terrestrial biomes

Country	2007 GDP (billion US$)	Per capita TEV (2007 US$ 000)	Cost of land degradation (2007 US$ billion)
Kazakhstan	104.85	21.52	23.73
Russia	1299.71	26.57	193.98
Papua New Guinea	6.33	29.12	−0.04
Central African Republic	1.70	33.96	5.35
Bolivia	13.12	43.36	24.25
Congo	8.39	43.82	7.77
Botswana	10.94	64.70	3.15
Mongolia	4.23	64.73	18.96
Canada	1424.07	72.83	114.26
Namibia	8.81	75.26	14.72
Australia	853.86	93.93	117.97
Gabon	11.57	110.94	1.89

Notes Countries with fewer than one million people are excluded

Land Degradation on Static Cropland

DSSAT Crop Simulation

The DSSAT crop simulation baseline land management practices were based on a compilation of global dataset and literature review. Given that there is a large difference between irrigated and rainfed land management practices, both the baseline and ISFM scenarios for irrigated and rainfed systems are simulated separately. In the irrigated simulation, a water management scenario is only applied to areas where water management is practiced.

We compare the amount of nitrogen used in the DSSAT simulation (Table 6.10) and the corresponding application rate obtained from FAOSTAT data (Table 6.11 and 6.12). We also compare the simulated and actual yield under irrigated and rainfed production systems. Table 6.9 shows that the average application rates of fertilizer in most regions is much lower than rates used in the DSSAT model. For example, while average application rate in SSA is 6 kgN/ha, it is 22 kgN/ha for rainfed maize. This large difference could be due to the fact that FAOSTAT nitrogen rate was computed by assuming that all cropland received fertilizer. Calibration of DSSAT model fertilizer rate assumed application rate at crop level, rather than entire cropland. However, FAO fertilizer application rate for each crop of the three crops considered in this study (maize, rice and wheat) is much higher than the corresponding average for all crops combined in each region (Table 6.9).

Table 6.10 Fertilizer application rates on cropland across regions

Region	Maize (KgN/ha)		Rice (KgN/ha)		Wheat (KgN/ha)	
	Irrigated	Rainfed	Irrigated	Rainfed	Irrigated	Rainfed
SSA		22.7	134.5	20.5	100.0	20.4
LAC	184.5	44.7	153.6	40.9		58.5
NAM			214.5			
East Asia						59.7
SE Asia		31.2	136.0			80.0
Oceania		70.3	184.5			59.9
South Asia	147.4	40.3	154.1			55.0
East Europe		60.0		90.0		60.0
West Europe	200.0	150.0			150.0	59.6
Central Asia	147.5		147.5			
NENA	149.8	60.0	141.7	20.0	141.8	60.0
Total	155.0	37.3	151.0	37.2	123.5	42.8

Note Empty cells imply that the production system is not applicable in the corresponding region
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East
See Appendix for countries in each region

Table 6.11 Application rate of Nitrogen used in DSSAT simulation

Region	N	P_2O_5	K_2O	NPK
	Average application (2001–10) Kg/ha			
SSA	6.04	3.00	1.83	10.86
NAM	59.6	21.1	21.1	101.8
LAC	29.7	23.9	23.0	76.6
South Asia	82.3	30.7	12.6	125.6
South-east Asia	60.2	15.5	22.3	98.1
East Asia	254.5	94.5	44.4	393.4
Central Asia	13.0	3.1	0.6	16.7
Oceania	25.0	31.0	5.5	61.6
East Europe	20.5	6.5	7.5	34.5
West Europe	95.8	26.8	29.9	152.5
NENA	42.2	14.3	3.7	60.2

Computed from FAOSTAT raw data
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East
Note See Appendix for countries in each region

For example application rate on maize and rice in north America is respectively 257 and 184 kgNPK/ha while the equivalent average amount for all crops is only 101 kgNPK/ha. The regional average may also mask the large differences within each region (Table 6.10).

Table 6.12 Application rate of NPK by crop

Region	Wheat	Maize	Rice
	kgNPK/ha		
NAM	84	257	184
LAC	76	67	90
West Europe	213	276	279
East Europe	95	40	–
USSR	25	294	107
Africa	63	55	19
Asia	144	117	140
World	116	136	134

Notes: *NAM* North America; *LAC* Latin American countries
Source FAO (2006)

Another challenge is to determine the adoption rate of ISFM in each country. We reviewed literature and used secondary data to determine adoption rate reported in Table 6.11. We then use the DSSAT simulation results at each pixel (half degree resolution) to determine the yield under ISFM and BAU scenarios and use the realistic adoption rates to determine the cost of land degradation on static cropland.

The secondary data used to determine adoption rate of ISFM include household surveys in SSA and conservation agriculture data reported by AQUASTAT website. Conservation agriculture is the practice that has soil cover throughout the year, minimizes soil disturbance through minimum tillage and spatio-temporal diversification of crops (Kassam et al. 2009; FAO 2008). Hence in countries with high fertilizer use, conservation agriculture could effectively mean ISFM since the crop residue component and crop rotation significantly increases soil carbon and yield. However, the impact of conservation agriculture on yield and profitability is heterogeneous (Pannell et al. 2014) but some of its components have been shown to have consistent positive impact. Zero tillage has been shown to significantly increase yield over long-term period in North America (Fulton 2010) and Australia (Llewellyn et al. 2012). Likewise, maize-legume rotation has been shown to increase yield of up to 25 % higher than monoculture (Brouder and Gomez-Macpherson 2014). Based on a global literature review, Palm et al. (2014) show that it increases biodiversity, topsoil organic matter and reduces soil erosion and runoff—leading to improved water quality.

The global adoption rate of conservation agriculture is 124 million ha or 9 % of the global cropland (Friedrich et al. 2012), 87 % of which is in Argentina, Australia, Brazil, Canada, and US (Brouder and Gomez-Macpherson 2014). The adoption rate in SSA and South Asia is generally low (Pannell et al. 2014; Brouder and Gomez-Macpherson 2014).

Due to the low adoption of conservation agriculture and fertilizer in SSA, conservation agriculture may not be equivalent to ISFM in the region. Hence we use household survey data to determine the adoption rate of ISFM in SSA. The average ISFM adoption rates in each region are reported in Tables 6.13 and 6.14.

Table 6.13 Adoption rates of SLM practices across regions

Region	Management practices	Adoption rate
SSA	Low-cost, productivity enhancing land management practices	3 % or 5 million ha on 191 million ha of cropland (Pender 2009)
Global (Kassam et al. 2009)	Conservation agriculture	10.2
LAC		37
SSA		0.7
LAC		26.6
NAM		20.6
Pacific		15.1
East Europe		1.7
Central Asia		5.7
West Europe		3.4
NENA		0.1
East Asia		10.0

Note See Appendix for countries in each region

Table 6.14 Adoption rates of inorganic and organic inputs and ISFM in SSA: Household survey

Country	ISFM	Organic inputs	Fertilizer	Nothing	Institution that collected data, data type and year survey conducted
	Adoption rate (percent)				
Mali	18	39	16	27	Direction nationale de l'informatique (DNSI). Recensement general de l'agriculture, 2004/2005
Uganda	0	67.61	0.96	31.42	Uganda Bureau of Statistics. Uganda national panel survey 2009/10
					Agriculture module
Kenya	16	22.3	17.44	43.66	ASDSP/KARI/UONa
					Kenya agricultural sector household baseline survey
Nigeria	1.28	28.23	23.31	47.17	IFPRI. Fadama III household survey, 2012
Malawi	7.52	2.77	51.58	38.14	National Statistics Office. Third integrated household survey, 2010/11, agricultural module
Tanzania	0.56	2.89	0.58	95.19	National bureau of statistics. National panel survey, agriculture module
Overall adoption rate (%)	6.2	19.1	24.6	49.8	

Notes: *ASDSP* Agriculture sector development support program; *KARI* Kenya Agricultural Research Institute; *UON* University of Nairobi

Results

Table 6.15 and Fig. 6.11 report the loss of ecosystems due to LUCC. Table 6.15 shows that the global annual average cost of land degradation due to LUCC was 2007 US$230.76 billion/year or 0.4 % of the global GDP in 2007. If the cost of land degradation were a country's GDP, it would be about the 8th richest country in the world. The total value of land degradation surpasses the GDP in 2007 of all countries in SSA. Figure 6.12 shows that SSA accounted for about 26 % of the cost of land degradation—underscoring the severity of land degradation in the region. Accordingly, the cost of land degradation is about 7 % of SSA's GDP—the highest level in the world. However, measured as percent of ecosystem total economic value (1.24 %), SSA's cost of land degradation is the second highest after NENA's, which is about 1.62 %. NAM, Pacific and East and West Europe experienced the lowest TEV loss of ecosystem services. In the humid and subhumid regions—where land degradation is more pronounced than in the arid and semi-arid regions (Bai et al. 2008), the Pacific region did remarkably well (Table 6.15). The results in

Table 6.15 Terrestrial ecosystem value and cost of land degradation due to LUCC

Region	GDP	Ecosystem value	% of TEV	Cost of land degradation		Cost of LD (TEV) as % of		
				TEV	Provisioning services only			
	2007 US$ billion/year			2007 US$ billion/year		GDP	TEV of ES	Total cost of LD
SSA	879.15	4844.17	18.82	60.290	30.34	6.86	1.24	26.13
LAC	3880.41	5958.52	23.15	52.551	22.31	1.35	0.88	22.77
NAM	15904.3007	3776.08	14.67	26.443	13.48	0.17	0.70	11.46
East Asia	10182.76	1552.63	6.03	16.704	5.87	0.16	1.08	7.24
Pacific	1001.55	1982.66	7.70	13.928	8.90	1.39	0.70	6.04
South Asia	1784.75	1065.43	4.14	9.664	2.55	0.54	0.91	4.19
SE Asia	861.12	562.02	2.18	5.793	1.82	0.67	1.03	2.51
Central Asia	180.4	492.30	1.91	5.743	12.58	3.18	1.17	2.49
West Europe	17144.86	684.37	2.66	5.252	2.14	0.03	0.77	2.28
East Europe	3023.14	4180.28	16.24	23.957	2.89	0.79	0.57	10.38
NENA	2040.19	643.99	2.50	10.436	3.74	0.51	1.62	4.52
Global	56882.69	25742.44	100	230.761	106.63	0.41	0.90	100

Notes: *SSA* Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East
See Appendix for countries in each region
Source GDP—World Bank data, TEV and land degradation—authors

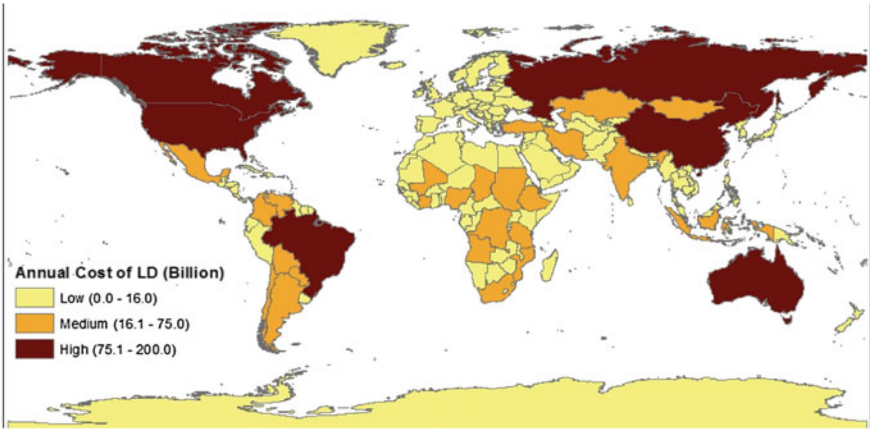


Fig. 6.11 Global cost of land degradation (2007 US$ billion), 2001–09

West Europe and NAM are consistent with Costanza et al. (2014) who reported increasing forest cover in these regions. The results in Europe are also consistent with Environmental performance index (EPI) ranking, which ranks region's performance in environmental health and ecosystem sustainability as highest in the world (EPI 2012). Nine of the countries with highest EPI ranking were European. European country with the lowest EPI ranking is Malta, which is the 87th of the total of 130 countries ranked.[4]

Who Bears the Burden of the Cost of Land Degradation?

We compare the cost of land degradation by separating the ES losses into two major components:

Provisioning services, which have direct impact on land users, and which account for the largest share of benefits that drive their decision making. This is the portion that has been used in many studies that do not use the TEV approach.

The value of the rest of ecosystem services—regulating, habitat and cultural services. These ecosystem services include both global benefits—such as carbon sequestration and biodiversity—and indirect local benefit, that land users may not assign low priority in their decision making process.

Figure 6.13 shows that loss of provisioning services account for only 38 % of the cost of land degradation—suggesting that the largest share of the cost of land degradation is borne by the global community. For example value of regulating

[4]Malta is included in the West Europe group in the EPI ranking but under NENA in this study.

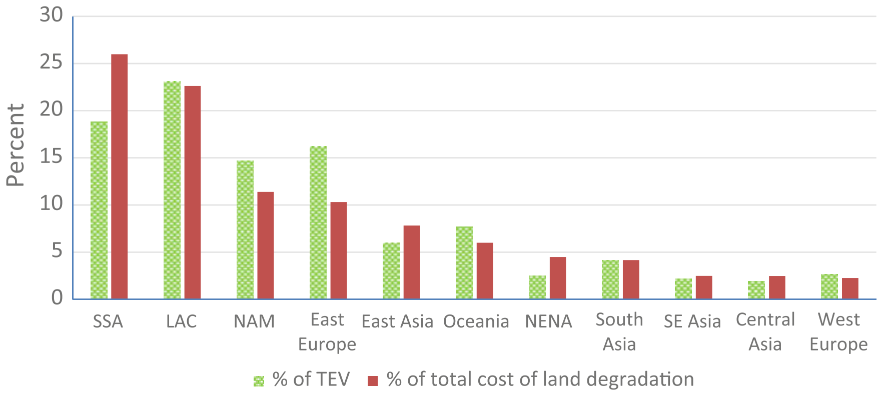
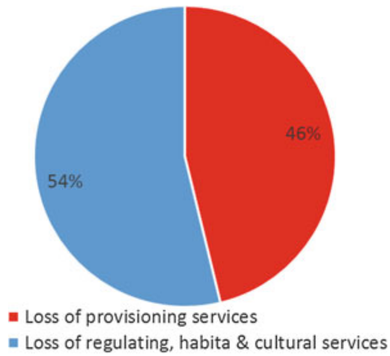


Fig. 6.12 Regional contribution of total economic value of terrestrial ecosystem services and cost of land degradation. *Note* See Appendix for countries in each region. *SSA* Sub-Saharan Africa; *LAC*

services accounts for the largest share of total economic value (TEV) of both tropical and temperate forests (Fig. 6.14). Provisioning services account for the lowest or second lowest share of TEV of both tropical and temperate forest TEV (Fig. 6.14). Thus if land holders are managing forests, the value of provisioning services will play the biggest role in decision making while regulating services will be given a low priority despite its large value. This suggests that land degradation is a global problem that requires both global and local solutions. Some studies that have compared the local benefits for protected areas showed that the benefit of converting forests to small-scale farming was greater than the benefit local communities draw from protected forests in Cameroon (Yaron 1999) or to unsustainably harvest timber in Malaysia (Shahwahid et al. 1999).

Fig. 6.13 Who bears the burden of the cost of land degradation?



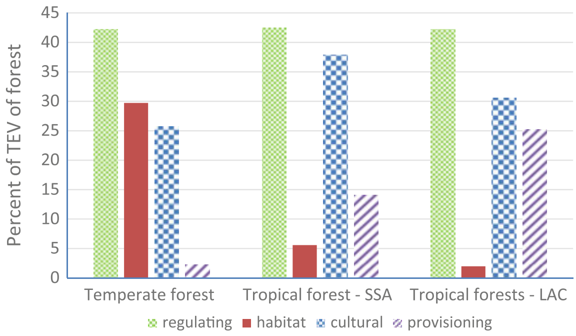


Fig. 6.14 Type of ecosystem services and their contribution to total value of forest biomes. *Notes* Average of total economic value (2007 US$) is 5264 (tropical forests) an 3013 (temperate forests). *Source* Calculated from TEEB database

Cost of Land Degradation Due to Use of Land Degrading Practices on Cropland

Table 6.16 shows that use of land degrading management practices in SSA on rainfed maize leads to a 25 % fall in yield compared to yield in the past 30 years. This is the highest loss of productivity of the cropland in the world. However, yield levels observed from the FAOSTAT shows an increase in yield in all regions for all crops in the corresponding periods simulated (Table 6.17). The reason for the inconsistency is that FAOSTAT yield includes yields from cropland expansion on forests and other virgin lands (Table 6.7) that is higher than yield on continuously cultivated cropland. Additionally, there has been an increase of fertilizer use and other inputs that mask the loss of productivity of land reflected in the simulation model (Le et al. 2014). The increase in use of fertilizer and improved technologies leads to higher yield despite the degraded lands. For example, Vlek et al. (2010) report land degradation in SSA. In NAM, East and West Europe and central Asia however, we see an increase in yield and consistent with the FAOSTAT yield trend. This could be a result of the higher use of fertilizer rates under BAU than yield under ISFM. But greater fertilizer use under BAU masks the environmental degradation due to eutrophication (enrichment of surface waters with plant nutrients) and other forms of water pollution (Glibert et al. 2006) that is not included in this study.

For irrigated rice, we see a fall in yield in all regions—as expected—except in Central Asia (Table 6.18). Surprisingly, the largest loss is experienced in NAM followed by LAC. Losses in SSA are only 20 %, the fifth largest in the world. For rainfed wheat, we see a yield decline in all regions except South Asia, central Asia and Asia and pacific (Table 6.19).

The cost of land degradation on static cropland is reported in Table 6.20 and is divided according to the components described in Eq. 6.2, i.e., loss of provisioning services and carbon sequestration under BAU and continuous cropping under ISFM.

Table 6.16 Change in rainfed maize yield under business as usual and ISFM—DSSAT results

Maize	BAU		ISFM		Yield change (%)		Change due to degradation/ improvement (%)
	Baseline	Endline	Baseline	Endline	BAU	ISFM	
	Yield (tons/ha)				$\% \Delta y = \frac{y_2 - y_1}{y_1} * 100$		$\% D = \frac{y_2^c - y_2^d}{y_2^d} * 100$
SSA	2.2	1.7	2.5	2.1	−23.2	13.9	25
LAC	3.4	3.1	3.8	3.6	−10.5	−6.7	16
NAM	6.1	6.4	5.7	6.2	4.2	10.1	−2
South Asia	3.4	3.1	3.6	3.4	−9.3	−5.8	11
Asia and Pacific	4.4	4.4	4.5	4.4	1.4	−0.4	1
East Europe	3.3	3.6	2.7	3.2	7.8	19.3	−12
Central Asia	5.1	5.5	4.1	4.9	7.1	18.3	−11
West Europe	5.3	5.6	4.4	5.1	5.4	15.6	−9
NENA	4.4	4.3	4.0	4.4	−1.1	7.9	1

Note Y1 = Baseline yield (average first 10 years); Y2 = Yield endline period (average last 10 years)
y_2^c = ISFM yield in the last 10 years; y_2^d = BAU yield, last 10 years
See Appendix for countries in each region
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa

Table 6.17 Actual crop yield and change

Region	Maize		Rice		Wheat	
	Baseline yield (Tons/ha)	Change (%)[a]	Baseline yield (Tons/ha)	Change (%)[a]	Baseline yield (Tons/ha)	Change (%)[a]
SSA	1.28	44.28	2.38	3.67	1.88	37.62
LAC	2.1	94	2.3	93.6	1.8	43
NAM	6.8	37.9	6	28.6	2.3	22.1
East Asia	3.9	33.4	5.3	18.7	2.9	51.5
Oceania	1.5	58.35	3.3	16.5	1.4	8.2
South Asia	1.4	74.3	2.3	42.1	1.8	44.9
SE Asia	1.7	101.6	2.9	36.5	1.4	15.3
East Europe	–	–	–	–	1.9	31.1
West Europe	5.53	37.4	5.45	11.55	5	11.73
Central Asia[b]	3.33	53.2	2.45	30.6	1.11	35.1
NENA	3.47	48	4.47	40.73	1.7	40.7

Note [a]Change ($\% \Delta y$) is computed $\% \Delta y = \frac{y_2 - y_1}{y_1} * 100$
[b]Baseline period for Central Asia is 1992–2001 and 1981–90 for the rest of regions. Endline for all regions is 2001–10
See Appendix for countries in each region
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East
Source FAOSTAT raw data

Table 6.18 Change in irrigated rice yield under business as usual and ISFM

Region	BAU		ISFM		Yield change (%)		Change due to degradation/improvement
	Yield (Tons/ha)						$\% \Delta y = \frac{y_2 - y_1}{y_1} * 100$
Baseline							$\% D = \frac{y_2^c - y_2^d}{y_2^d} * 100$
Endline	Baseline	Endline	BAU	ISFM			
SSA	4.4	3.2	4.9	3.9	−26.8	−19.9	20
LAC	7.6	5.5	8.8	7.1	−27.5	−19.2	29
NAM	4.8	5.9	6.1	7.8	22.5	28.0	33
South Asia	6.5	4.9	7.5	6.1	−24.9	−18.0	25
Asia and Pacific	3.5	6.1	4.2	7.8	75.6	86.5	27
East Europe	2.1	5.3	2.1	5.4	147.3	157.0	2
Central Asia	0.8	1.4	0.8	1.4	69.6	68.0	−1
West Europe	7.7	5.1	8.1	5.7	−33.4	−29.2	11
NENA	1.0	2.8	1.1	3.2	189.3	180.8	15

Note Y1 = Baseline yield (average first 10 years); Y2 = Yield endline period (average last 10 years)
y_2^c = ISFM yield in the last 10 years; y_2^d = BAU yield, last 10 years
See Appendix for countries in each region
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa

Table 6.19 Change in rainfed wheat yield under business as usual and ISFM—DSSAT results

	BAU		ISFM		Yield change (%)		Change due to degradation/ improvement
	Baseline	Endline	Baseline	Endline	$\% \Delta y = \frac{y_2 - y_1}{y_1} * 100$		$\% D = \frac{y_2^c - y_2^d}{y_2^d} * 100$
SSA	1.4	1.2	1.4	1.3	−15.2	−10.7	8
LAC	1.8	1.6	1.8	1.7	−8.8	−6.7	1
NAM	2.3	2.2	2.4	2.2	−7.7	−6.3	3
South Asia	1.4	1.3	1.3	1.1	−8.8	−11.8	−12
Asia and Pacific	2.0	1.8	1.9	1.8	−6.1	−5.6	−1
East Europe	1.3	1.1	1.4	1.2	−10.9	−9.7	7
Central Asia	0.8	0.8	0.8	0.8	0.8	2.2	−7
West Europe	2.1	2.0	2.2	2.1	−7.7	−7.2	6
NENA	1.3	1.2	1.3	1.2	−7.7	−4.9	2

Note Y1 = Baseline yield (average first 10 years); Y2 = Yield endline period (average last 10 years)
y_2^c = ISFM yield in the last 10 years; y_2^d = BAU yield, last 10 years
See Appendix for countries in each region
SSA Sub-Saharan Africa; *LAC* Latina American countries; *NAM* North America; *NENA* Near East and North Africa

The global cost of land degradation for the three crops is about US$56.60 billion per year (Table 6.20), of which, East and South Asia accounted for the largest share of loss. However when the loss is expressed as percent of GDP, South Asia experiences the most severe cost of land degradation on cropland. The cost of land degradation shown is generally low than what has been reported in other studies largely due to DSSAT's assumption of much higher BAU fertilizer application rates. This reduces the actual cost of land degradation. Additionally, DSSAT assumes no salinity or soil erosion. This further demonstrates the underestimation of land degradation on static cropland. The total cost due to the loss of carbon sequestration accounts for 67 % of the total cost at global level—suggesting the cost of land degradation on static cropland is borne more heavily the global community than the farmers. The results also underscore the great potential of ISFM in carbon sequestration.

The three crops account for about 42 % of the cropland in the world. If all cropland is assumed to experience the same level of degradation, the total cost of land degradation on cropland is about 0.25 % of the global GDP.

As discussed in the introduction section, our estimates are conservative since we do not take into account other costs of land degradation. For example we do not include off-site cost of pesticide use, which are quite high. Pimental et al. (1995) estimated that the environmental and social costs were about US$8 billion per year, of which $5 billion are external social costs. The social costs considered were human health and the environmental effects were, pest resistance, loss of natural enemies, groundwater contamination, and loss of pollinating insects and other agents (Ibid).

We also do not consider the point and nonpoint pollution of inorganic fertilizer that leads to eutrophication and other forms of surface and underground water

Table 6.20 Cost of soil fertility mining on static maize, rice and wheat cropland

Region	Cost of land degradation (2007 US$) due to		Type of ecosystem loss			Total cost	Cost of LD as % of GDP
	BAU	Continuous ISFM	Provisioning services	CO_2 sequestration			
				BAU	continuous ISFM		
SSA	0.689	0.126	0.815	1.604	0.947	3.367	0.38
LAC	0.433	0.194	0.627	2.006	2.015	4.648	0.12
NAM	0.275	0.165	0.44	5.00	1.013	6.453	0.04
East Asia	4.331	0.244	4.575	7.071	1.708	13.354	0.13
Oceania	0.03	0.045	0.075	0.365	0.47	0.909	0.09
South Asia	4.724	0.5	5.224	4.541	4.093	13.858	0.78
SE Asia	1.439	0.22	1.659	0.516	1.651	3.827	0.44
East Europe	0.144	0.034	0.178	3.045	0.275	3.498	0.12
West Europe	0.16	0.027	0.187	1.872	0.161	2.219	0.01
Central Asia	0.007	0.004	0.011	0.257	0.076	0.344	0.19
NENA	0.261	0.04	0.301	3.373	0.448	4.122	0.20
Total	12.493	1.599	14.092	29.651	12.856	56.599	0.10

Note See Appendix for countries in each region
SSA Sub-Saharan Africa; *LAC* Latin American countries; *NAM* North America; *NENA* Near East and North Africa; *SE* South East
Source Authors

pollution. About 47 % of nitrogen applied is lost annually to the environment through leaching, erosion, runoff, and gaseous emissions (Roy et al. 2002). Agriculture is the leading cause of eutrophication and other forms of freshwater pollution (Ongley 1996). Pretty et al. (2003) estimated the cost of eutrophication in the United Kingdom to be about £75.0–114.3 million or 2003 US$ 127 to 193 million and 2.2 billion in US (Dodds et al. 2003). Another study estimated that water pollution costs from agriculture in the United Kingdom is US$141–300 million per year or about 1–2 % of the value of gross agricultural output (DEFRA 2010; Pretty et al. 2003). At a global level, Dodds et al. (2013) estimated the loss of freshwater ecosystems due to human activities is 2013 US$900 billion per year. In general, our estimates are conservative due to the limitation of the crop modeling used and future studies are required to take into account the gaps in this study.

Cost of Land Degradation on Grazing Biomass

The cost of land degradation on grazing land that takes into account only loss of milk and meat production is about 2007US$7.7 billion (Table 6.21). As discussed in Chap. 8, loss of milk production accounts for the largest share of total cost. NAM

Table 6.21 Cost of loss of milk and meat production due to land degradation of grazing biomass

Regions	Milk	Meat	Total	Gross total	Percent of total cost
	2007 US$ Million				
SSA	1018.02	127.26	1145.28	1489.46	15
LAC	1082.78	82.46	1165.23	1494.67	15
NAM	2633.68	283.49	2917.17	3495.73	38
East Asia	13.62	5.08	18.70	22.66	0
Oceania	336.75	190.33	527.08	565.25	7
South Asia	16.00	0.90	16.90	21.54	0
SE Asia	156.76	2.30	159.05	178.11	2
East Europe	271.44	364.11	635.55	360.92	8
West Europe	586.93	252.10	839.03	941.58	11
Central Asia	102.51	6.63	109.14	126.38	1
NENA	15.07	113.80	128.88	42.71	2
Global	6233.56	1428.45	7662.01	8739.02	

Note: *NAM* North America, *LAC* Latin American Countries, *SSA* Sub-Saharan Africa, and *NENA* Near East and North Africa

accounts for 38 % of the total cost due to the high productivity of livestock system in the region and the severe land degradation that occurred. Other regions that experienced severe grazing land degradation are SSA and LAC.

Summary of Cost of Land Degradation

Table 6.22 shows that the total cost of land degradation due to LUCC and use of land degrading management practices on static cropland and grazing land is about US$300 billion. LUCC accounts for the largest of total cost of land degradation. This is largely due to its broader coverage of biomes and ecosystems services. Likewise, SSA and West Europe respectively accounts for the largest and smallest share of the global total cost of land degradation.

We now turn to cost of action against land degradation in order to determine whether action could be justified economically. As Nkonya et al. (2013) note, an action against land degradation will be taken if the cost of inaction is greater than the cost of taking action.

Cost of Action Against land degradation

We computed the cost of taking action against land degradation using Eq. (6.5). The components of taking action against land degradation, namely the cost of establishing and maintaining degraded biome, and the opportunity cost of taking

Table 6.22 Summary of cost of land degradation

Region	Type of land degradation			Total cost of LD	Cost of LD as percent of	
	LUCC	Use of land degrading management practices on:				
		Cropland	Grazing lands			
	2007 US$ billion				GDP	Total cost
SSA	60.29	3.367	1.49	65.15	7.4	22.0
LAC	52.551	4.648	1.49	58.69	1.5	19.8
NAM	26.443	6.453	3.50	36.39	0.2	12.3
East Asia	16.704	13.354	0.02	30.08	0.3	10.2
Pacific	13.928	0.909	0.57	15.40	1.5	5.2
South Asia	9.664	13.858	0.02	23.54	1.3	8.0
SE Asia	5.793	3.827	0.18	9.80	1.1	3.3
East Europe	23.957	3.498	0.36	27.82	0.9	9.4
Central Asia	5.743	2.219	0.94	8.90	4.9	3.0
West Europe	5.252	0.344	0.13	5.72	0.0	1.9
NENA	10.436	4.122	0.04	14.60	0.7	4.9
Global	230.761	56.599	8.74	296.10	0.5	

Note: *LD* Land degradation
Sources Tables 6.15, 6.20 and 6.21

action—are explained in detail in the methods section. This section only presents the results. To completely rehabilitate land degradation due to LUCC in all regions, a total of US$4.6 trillion will be required in 6 years (Table 6.22). But if action is not taken to rehabilitated degraded lands, the world will incur a loss of US$14 trillion during the same.

During the entire 30-year planning horizon, the cost of action is at most 34 % of the cost of inaction. The opportunity cost accounts of taking action accounts for over 90 % of the total cost of action in the first 6 years in all but one region (NENA). This suggests there is a large opportunity cost of taking action against land degradation and such opportunity cost explains the economic rationale of land degradation for private land users. Over the 30 year planning horizon, the cost of action falls dramatically once the opportunity cost is dropped at the establishment period.[5] This means it is the establishment period that matters most and not the rest of the planning horizon (Table 6.23).

The returns to taking action against land degradation are quite high. In the first 6 years, land users will get at least US$2 for every dollar they spend on rehabilitating degraded lands. At the end of land user's 30-year planning horizon, the

[5]Please see discussion in the methods section on why the opportunity cost is dropped at the end of the establishment period.

Table 6.23 Cost of action and inaction against LUCC-related land degradation during the rehabilitation period and planning horizon

Region	Cost of action	Cost of inaction	Cost of action	Cost of inaction	Cost of action as % of cost of inaction[a]		Opportunity cost as % of cost of action, (1st 6 years)	Returns to action against LD		
	First 6 years		30-year planning horizon		1st 6 years	30-year planning horizon				
	2007 US$ billion							6 years	30 years	Without opportunity cost
SSA	795	2696	797	3343	29	24	96	3	4	80
LAC	752	2309	754	2977	33	25	98	3	4	167
NAM	739	2251	751	4545	33	17	93	3	6	45
East Asia	495	1278	508	2594	39	20	98	3	5	150
Oceania	399	1247	407	2442	32	17	97	3	6	105
South Asia	210	493	210	646	43	33	98	2	3	137
SE Asia	134	304	135	400	44	34	98	2	3	148
East Europe	765	2366	777	4813	32	16	92	3	6	36
West Europe	178	451	181	926	39	20	96	3	5	57
Central Asia	53	230	53	277	23	19	97	4	5	130
NENA	80	395	80	504	20	16	81	5	6	27
Total	4600	14021	4653	23465	33	20	94	3	5	50

[a]The inverse of the corresponding percent is the returns on investment
Note See Appendix for countries in each region

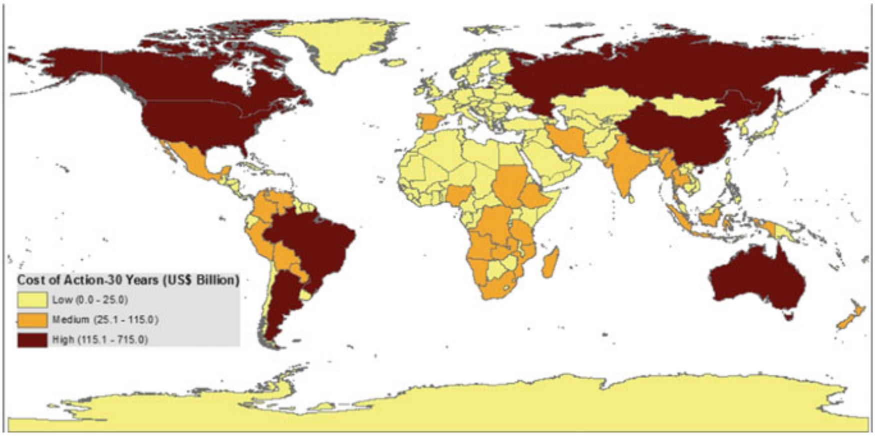


Fig. 6.15 Cost of action against land degradation, 30-year planning horizon

returns to taking action against land degradation increases to at least US$3 for each dollar invested. If we ignore the opportunity cost and consider only the actual cost incurred by land users to address land degradation, the returns are at least US$27 per dollar invested (Table 6.23). The results suggest that the large returns to investment in addressing land degradation but also raise important question as to why many land users do not take action despite the high returns. The chapter on drivers of land degradation addresses this question.

The global distribution of the cost of taking action against land degradation (Fig. 6.15) is consistent with the pattern revealed in the cost of land degradation (Fig. 6.11).

Contrary to Bai et al. (2008), land degradation is severe in both temperate and tropical regions. However the corresponding cost of taking action is highest in high income countries due to their high value of land and labor costs and other factors discussed by Benítez et al. (2007). Country-level cost of taking action against land degradation are highest in North America, Russia, China, Australia Brazil and Argentina. However, regional analysis show that SSA contributes the largest share (17 %) of cost of taking action against land degradation (Fig. 6.16) despite having the small unit cost of biome restoration (Sathaye et al. 2006). This is due to the extent and severity of land degradation in the region. East Europe, North America and LAC also contribute large shares of cost of taking action against land degradation while West Europe, NENA and Central Asia contribute smallest shares. The results underline the global nature of land degradation and the corresponding cost of taking action to address the problem and where large costs are expected to be incurred.

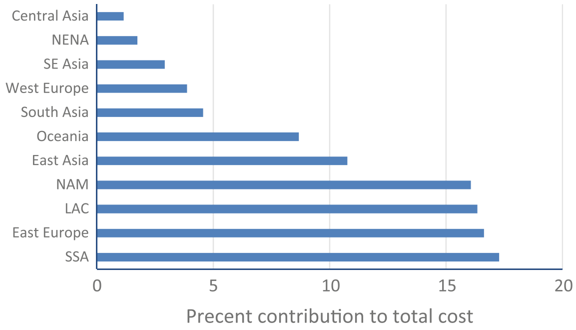


Fig. 6.16 Regional cost of taking action against land degradation

Conclusions and Policy Implications

Land degradation is a global problem that requires both local and global policies and strategies to address it. The global community bears the largest share of land degradation while the local land users where biomes are located bears a smaller share of the cost. As expected, the cost of taking action against land degradation is lower than the cost of inaction even when one considers only the first 6 years of rehabilitation. Returns to investment in action against land degradation is at least twice the cost of inaction in the first six years. But when one takes into account the 30-year planning horizon, the returns are five dollars per dollar invested in action against land degradation. The opportunity cost of taking action accounts for the largest share of the cost and this contributes to inaction in in many countries. Furthermore, the prices of land (and shadow prices) are expected to increase as the world gets wealthier and more crowded moving from 7 to 9 billion in the coming generation. Any further degradation of land and soils will increase even more with the increase of the value of the degraded resources.

Strategies should be developed that give incentives to better manage lands and reward those who practice land management that provide significant global ecosystem services. The payment for ecosystem services (PES) mechanisms that saw large investments in carbon markets should be given a new impetus to address the loss of ecosystem services through land use/cover change (LUCC) which accounts for the largest cost of land degradation.

SSA accounts for the largest share of land degradation and the corresponding cost of action. The global community needs to pay greater attention to addressing land degradation in SSA, since the region accounts for the largest share of total value of ecosystem services and that its highest level of poverty and other challenges reduces its capacity to achieve United Nations Convention to Combat desertification (UNCCD)'s target of zero net land degradation by year 2030. The new strategies need to learn from past success stories and failed projects. There are success stories that have proven that even poor farmers could practice sustainable land management practices. The case of Niger and the re-greening of the Sahel demonstrates this. The top-down programs implemented in developing countries prove that they rarely work.

The extent of land degradation high cost of taking action against land degradation in high income countries also requires greater attention. However, the large endowment of financial and human capital and greater government effectiveness give the high income a greater opportunity to achieving UNCCD's target of zero net land degradation by year 2030.

Appendix

Countries, Sub-regions and Regions

Central Africa	Caribbean countries	East Asia	East Europe	Central Asia
Cameroon	Puerto Rico	Macao	Kosovo	Kyrgyzstan
Gabon	Cayman Islands	Hong Kong	Armenia	Azerbaijan
Equatorial Guinea	Dominica	China	Ukraine	Uzbekistan
Central African Rep.	Bahamas	South Korea	Iceland	Turkmenistan
Sao Tome and Principe	Saint Vincent and the Grenadines	Mongolia	Slovakia	Tajikistan
DRC	Cuba	Taiwan	Slovenia	Kazakhstan
Congo	Turks and Caicos Islands	North Korea	Poland	**NENA**
Eastern Africa	Dominican Republic	Japan	Belarus	Lebanon
Eritrea	Guadeloupe	**Pacific/Oceania**	Croatia	Jordan
Burundi	Barbados	Niue	Hungary	Morocco
Kenya	Haiti	Tuvalu	Latvia	Malta
Ethiopia	French Guiana	Papua New Guinea	Czech Republic	Syria
Uganda	Jamaica	Tonga	Lithuania	Tunisia
Somalia	Antigua and Barbuda	New Zealand	Romania	West Bank
Sudan	Montserrat	Fiji	Albania	Algeria
Rwanda	Belize	French Polynesia	Montenegro	Libyan Arab Jamahiriya
Djibouti	Saint Helena	Guam	Czechoslovakia	Cyprus

(continued)

(continued)

Central Africa	Caribbean countries	East Asia	East Europe	Central Asia
Indian Ocean	Saint Kitts and Nevis	Micronesia	Bulgaria	Israel
Mayotte	Guyana	Marshall Islands	Yugoslavia	Western Sahara
Comoros	Anguilla	Samoa	Bosnia and Herzegovina	Egypt
Mauritius	Suriname	Cook Islands		Gibraltar
Madagascar	Grenada	Palau	Russia	UAE
Seychelles	Netherlands Antilles	Wallis and Futuna Islands	Macedonia	Turkey
Reunion	Martinique	Kiribati	Estonia	Yemen
Southern Africa	British Virgin Islands	New Caledonia	Serbia and Montenegro	Iraq
Botswana	Trinidad and Tobago	Vanuatu	Georgia	Kuwait
Malawi	Bermuda	Solomon Islands	Modova	Qatar
Zambia	Saint Lucia	Northern Mariana Islands	Serbia	Palestinian Territory
Namibia	Aruba	Pitcairn Islands	**West Europe**	Afghanistan
Zimbabwe	**Central America**	American Samoa	Isle of Man	Saudi Arabia
Angola	Honduras	Norfolk Island	Belgium	Oman
Lesotho	Costa Rica	Tokelau	Norway	Pakistan
Mozambique	Nicaragua	Pacific Islands Trust Territory	Andorra	Bahrain
South Africa	El Salvador	Australia	Saint Pierre and Miquelon	Iran
Swaziland	Guatemala	**Southern Asia**	United Kingdom	
Tanzania	Mexico	Nauru	Greece	
Western Africa	Panama	India	Liechtenstein	
Chad	**North America**	Sri Lanka	France	
Ghana	Canada	Nepal	Sweden	
Sierra Leone	US Virgin Islands	Bhutan	Ireland	
Gambia	USA	Maldives	San Marino	
Burkina Faso	Southern America	Bangladesh	MONACO	
Guinea-Bissau	Chile	Indonesia	Netherlands	
Benin	Colombia	Viet Nam	Greenland	
Côte d'Ivoire	Falkland Islands (Malvinas)	Cambodia	Portugal	
Mauritania	Uruguay	Philippines	Luxembourg	

(continued)

(continued)

Central Africa	Caribbean countries	East Asia	East Europe	Central Asia
Guinea	Argentina	Lao People's Democratic Republic	Switzerland	
Mali	Ecuador	Timor-Leste	Finland	
Togo	Brazil	Singapore	Spain	
Senegal	Bolivia	Malaysia	Denmark	
Nigeria	Peru	Brunei Darussalam	Italy	
Niger	Venezuela	Thailand	Channel Islands	
Liberia	Paraguay	Myanmar	Faroe Islands	
Cape Verde			Germany	
			Austria	

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