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Chapter G2 Carbon emissions from land use and land-cover change

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Abstract

The net flux of carbon from land use and land-cover change (LULCC) is significant in the global carbon budget but uncertain, not only because of uncertainties in rates of deforestation and forestation, but also because of uncertainties in the carbon density of the lands actually undergoing change. Furthermore, there are differences in approaches used to determine the flux that introduce variability into estimates in ways that are difficult to evaluate, and there are forms of management not considered in many of the analyses. Thirteen recent estimates of net carbon emissions from LULCC are summarized here. All analyses consider changes in the area of agricultural lands (croplands and pastures). Some consider, also, forest management (wood harvest, shifting cultivation). None of them includes the emissions from the degradation of tropical peatlands. The net flux of carbon from LULCC is not the same as "emissions from deforestation", although the terms are used interchangeably in the literature. Means and standard deviations for annual emissions are 1.14 ± 0.23 and $1.13 \pm 0.23\,\mathrm{Pg\,C\,yr^{-1}}$ ($1\,\mathrm{Pg} = 10^{15}\,\mathrm{g}$ carbon) for the 1980s and 1990s, respectively. Four studies also consider the period 2000–2009, and the mean and standard deviations for these four are 1.14 ± 0.39, 1.17 ± 0.32, and $1.10 \pm 0.11\,\mathrm{Pg\,C\,yr^{-1}}$ for the three decades. For the period 1990–2009 the mean global emissions from LULCC are $1.14 \pm 0.18\,\mathrm{Pg\,C\,yr^{-1}}$. The errors are smaller than previously estimated, as they do not represent the range of error around each result, but rather the standard deviation across the mean of the 13 estimates. Errors that result from data uncertainty and an incomplete understanding of all the processes affecting the net flux of carbon from LULCC have not been systematically evaluated but are likely to be on the order of $\pm 0.5\,\mathrm{Pg\,C\,yr^{-1}}$.

1 Introduction

The sources and sinks of carbon from land use and land-cover change (LULCC) are significant in the global carbon budget. The contribution of LULCC to anthropogenic carbon emissions were about 33 % of total emissions over the last 150 yr (Houghton, 1999), 20 % of total emissions in the 1980s and 1990s (Denman et al., 2007), and 12.5 % of total emissions over 2000 to 2009 (Friedlingstein et al., 2010). The declining fraction is largely the result of the rise in fossil fuel emissions.

The flux of carbon from LULCC does not represent the net flux of carbon between land and atmosphere. Unmanaged terrestrial ecosystems also contribute to changes in the land-atmosphere net flux. There are large annual exchanges of CO_2 between ecosystems (plants and soils) and the atmosphere due to natural processes (photosynthesis, respiration) with substantial interannual variability related to climate variability. The land is currently a net sink despite LULCC emissions. This net sink is likely attributable to the affects of environmental changes on plant growth, such as the fertilizing effects of rising CO_2 in the atmosphere and nitrogen (N) deposition, and changes in climate, such as longer growing seasons in northern extra-tropical regions. These environmental drivers affect both managed and unmanaged lands and make attribution of carbon fluxes to LULCC difficult. LULCC, in theory, includes only those fluxes of carbon attributable to direct human activity and excludes those fluxes attributable to natural or indirect human effects. In practice, however, attribution is difficult, in part because of the interactions between direct and indirect effects. It is difficult to establish how much of the carbon accumulating in a planted forest, for example, can be attributed to management, as opposed to increasing concentrations of CO_2 in the atmosphere.

Recent estimates of the flux of carbon from LULCC are shown in Fig. 1 and summarized briefly in Table 1. A few of the estimates are not strictly global but include only tropical regions (DeFries et al., 2002; Achard et al., 2004). Nevertheless, these estimates for the tropics appear to fit within the range of global estimates because the net annual flux of carbon due to LULCC from regions outside the tropics has been nearly zero over the last decades (Houghton, 2003). This near neutrality may be misleading, however. It does not indicate a lack of activity outside the tropics. Indeed, annual gross sources and sinks of carbon from LULCC are nearly as great in temperate and boreal regions as they are in the tropics (Richter and Houghton, 2011). Rates of wood

harvest, for example, are nearly the same in both regions. The main difference between the two regions is that forests are being lost in the tropics, while forest area has been expanding in Europe, China, and North America.

The mean annual net flux of carbon from LULCC based on these recent estimates is 1.14 ± 0.23 and 1.13 ± 0.23 Pg C yr^{-1} (10^{15} g carbon yr^{-1}) for the 1980s and 1990s, respectively. The four estimates for 2000–2009 yield mean net sources of 1.14 ± 0.39, 1.17 ± 0.32, and 1.10 ± 0.11 Pg C yr^{-1} for the 1980s, 1990s, and 2000–2009, respectively. Only one of these estimates (Houghton, 2010) is based on the recent data for deforestation rates (FAO, 2010). The three others are forced by scenarios after 2000 or 2005. For the longer interval 1990–2009 the mean net flux for all analyses is 1.14 ± 0.18 Pg C yr^{-1}. The errors refer to the standard deviations across the model estimates; they do not reflect the larger uncertainty within each estimate due to uncertainty in data and uncertainty in understanding and accounting for multiple processes, such as forest management (see Sects. 4 and 5, below). Few estimates include an assessment of the inherent uncertainty although there have been limited studies of uncertainty in estimating LULCC emissions (Houghton, 2005; Ramankutty et al., 2007). It is the expert judgment of the authors here that uncertainty is in the region of ±0.5 Pg C yr^{-1}.

The discussion below focuses on identifying the reasons for differences among these recent estimates. Differences are grouped into three major categories: data on rates and area of LULCC and carbon density of soils and vegetation before and after change, the types of LULCC processes included, and the treatment of environmental change (e.g. CO_2 and N fertilization, changes in temperature and moisture).

Note that LULCC affects climate through emissions of chemically and radiatively active gases besides CO_2, including other carbon compounds. Further, LULCC affects climate biogeophysically as well as biogeochemically through effects on surface albedo, surface roughness, and evapotranspiration (e.g. Pongratz et al., 2010). Non-CO_2 gases and biophysical effects are not considered here.

2 Approaches and data

2.1 What is land use and land-cover change?

Ideally, land use and land-cover change would be defined broadly to include not only human-induced changes in land cover, but all forms of land management (e.g. tillage, fertilizer use, shifting cultivation, selective logging, draining of peatlands, use or exclusion of fire). The reason for this broad ideal is that the net flux of carbon attributable to management is that portion of a terrestrial carbon flux that might qualify for credits and debits under a post-Kyoto agreement. However, it is not possible to separate management effects from natural and indirect effects (CO_2 fertilization, N deposition, and the effects of climate change) from measurements alone, and model estimates vary considerably. Furthermore, the ideal of monitoring the impacts of all forms of land management activity requires more data, at higher spatial and temporal resolution, than has been practical (or possible) to assemble at the global level. Thus, most analyses of the effects of LULCC on carbon have focused on the dominant (or documentable) forms of management and, to a large extent, ignored others.

In this chapter the term "land use" refers to management within a land-cover type. For example, the harvest of wood does not change the designation of the land as forest although the land may be temporarily treeless. "Land-cover change", in contrast, refers to the conversion of one cover type to another, for example, the conversion of forest to cropland. The largest emissions of carbon have been from land-cover change, particularly the conversion of forests to non-forests, or deforestation. All of the analyses reviewed here have included change in forest area, and most have included other changes in land cover (e.g. natural grassland to pastureland). However, most analyses include little if any land use (management) despite the effects of land use on terrestrial carbon storage.

All of the approaches for calculating the emissions of carbon from LULCC consider the areas affected (e.g. deforested or reforested) (Sect. 2.2) and emissions coefficients (carbon lost or gained per hectare following a change in land management) (Sect. 2.3).

The approaches differ, first, in the way managed areas are identified and measured; and, second, in the way carbon stocks and changes in carbon stocks are estimated (some are modeled, others are specified from observations). Approaches also differ in the types of land use and land-cover change considered (Sect. 4).

2.2 Changes in area

Three approaches have been used to document changes in the area of ecosystems or changes in land cover: nationally-aggregated land-use statistics, satellite data on land cover, and satellite data on fires.

2.2.1 National census data

Some analyses, especially for historic changes, are based on aggregated, non-spatial data on LULCC, as reported in national and international statistics. The UN's Food and Agriculture Organization (FAO) provides two data sets that have been used to estimate changes in land cover over recent decades. One data set (FAOSTAT, 2009) reports annual areas in croplands, pastures, forests, and other lands. The other data set (Forest Resource Assessments; FRAs; FAO, 2001, 2006, 2010) pertains to forests alone. These FAO data sets can be used in combination to assign deforested areas to either croplands or pastures, as in the Houghton data set (Houghton, 2003), or to constrain assumptions about whether agricultural expansion occurs at the expense of grasslands or forests, as in the SAGE and HYDE data sets (Ramankutty and Foley, 1999; Klein Goldewijk, 2001; Pongratz et al., 2008). The distinctions are important because different land uses have different carbon stocks, and the carbon flux resulting from LULCC depends on assumptions about land cover before and after change. The FAO data sets have also been used in combination to estimate rates of deforestation for shifting cultivation (Houghton and Hackler, 2006), a rotational use of land with repeated clearing and subsequent regrowth of fallow forests.

The data from the FAO include nearly all countries and, hence, enable global estimates to be calculated. The data are not spatially explicit, however, and require independent data or allocation rules to assign deforestation to particular ecosystem types (with specific carbon densities). The FAO data report annual areas of different land covers (forest, agriculture and pasture), which provide the basis for calculating annual rates of land-cover change, but these are net changes, not gross changes. Net changes in land cover underestimate gross sources and sinks of carbon that result from simultaneous clearing for, and abandonment of, agricultural lands, thus underestimating subsequent areas of secondary forests and their carbon sinks.

The FAO data rely on reporting by individual countries. They are more accurate for some countries than for others and are not without inconsistencies and ambiguities (Grainger, 2008). Revisions in the reported rates of deforestation from one 5-yr FRA assessment to the next may be substantial due to different methods or data being used. FAO estimates of deforestation rates over the last few decades have been substantially reduced by incorporating satellite data (FAO, 2001, 2006, 2010).

2.2.2 Satellite data on land cover

A second approach for estimating LULCC is to use a time-series of satellite data to estimate the spatio-temporal dynamics of forest area change. In general, satellite data alleviate the concerns of bias, inconsistency, and subjectivity in country reporting (Grainger, 2008). Satellite data can also distinguish between gross and net forest area loss, although increases in forest area are more difficult to observe with satellite data than decreases because forest growth is a more gradual process. Furthermore, although satellite data are good for measuring changes in forest area, they have generally not been used to distinguish the types of land use following deforestation (e.g. croplands, pastures, shifting cultivation). Exceptions include the regional studies by Morton et al. (2006) and Galford et al. (2008).

Satellite-based methods include both high-resolution sample-based methods and wall-to-wall mapping analyses. Sample-based approaches employ systematic or

stratified random sampling to quantify gains or losses of forest area at national, regional and global scales (Achard et al., 2002, 2004; Hansen et al., 2008, 2010). Systematic sampling provides a readily implementable and easily understood framework for forest area monitoring. The UN-FAO Forest Resource Assessment Remote Sensing Survey will use samples at every latitude/longitude intersection to quantify biome and global-scale forest change dynamics from 1990 to 2005 (FAO, 2007). Other sampling approaches stratify by intensity of change, thereby reducing sample intensity. Achard et al. (2002) provided an expert-based stratification of the tropics to quantify forest cover loss from 1990 to 2000 using whole Landsat image pairs. Hansen et al. (2008a, 2010) employed MODIS data as a change indicator to stratify biomes into regions of homogeneous change for a 2000 to 2005 study. Within each stratum, 18.5 km sample blocks of Landsat image pairs were characterized to derive estimates of gross forest-cover loss at biome, continental and national scales.

Sampling methods such as described above provide regional forest area and change estimates with uncertainty bounds, but they do not provide a spatially explicit map of forest extent or change. Wall-to-wall mapping does. While coarse-resolution data sets (>4 km) have been calibrated to estimate wall-to-wall changes in area (DeFries et al., 2002), recent availability of moderate spatial resolution data (<100 m), typically Landsat imagery (30 m), allows a more finely-resolved approach. Historical methods rely on photointerpretation of individual images to update forest cover on annual or multi-year bases, such as with the Forest Survey of India (Global Forest Survey of India, 2008) or the Ministry of Forestry Indonesia products (Government of Indonesia/World Bank, 2000). Advances in digital image processing have led to the operational implementation of mapping annual forest cover loss with the Brazilian PRODES (INPE, 2010) and the Australian National Carbon Accounting products (Caccetta et al., 2007). These two systems rely on cloud-free data to provide single-image/observation updates on an annual basis. Persistent cloud cover has limited the derivation of products in regions such as the Congo Basin and Insular Southeast Asia (Ju and Roy, 2010). For such areas, Landsat data can be used to generate multi-year estimates of forest-cover extent and loss (Hansen et al., 2008b; Broich et al., 2011a). For regions experiencing forest change at an agro-industrial scale, MODIS data provide a capability for integrating Landsat-scale change to annual time-steps (Broich et al., 2011b).

In general, moderate spatial resolution imagery is limited in tropical forest areas by data availability. Currently Landsat is the only source of data at moderate spatial resolution available for tropical monitoring, but to date an uneven acquisition strategy along with varying bioclimatic regimes limit the application of generic biome-scale methods with Landsat. No other system has the combination of (1) global acquisitions, (2) historical record, (3) free and accessible data, and (4) standard terrain-corrected imagery, along with robust radiometric calibration, that Landsat does. Future improvements in moderate spatial resolution tropical forest monitoring can be delivered largely by increasing the frequency of data observations.

The primary weakness of satellite data is that they are not available before the satellite era (Landsat began in 1972). Long time-series are required for estimating legacy emissions of past land-use activity (Sect. 3.2). Although maps, at varying resolutions, exist for many parts of the world, spatial data on land cover and land-cover change become available at a global level only after 1972, at best. In fact, there are many holes in the coverage of the earth's surface until 1999 when the first global acquisition strategy for moderate spatial resolution data was undertaken with the Landsat Enhanced Thematic Mapper Plus sensor (Arvidson et al., 2001). The long-term plan of Landsat ETM+ data includes annual global acquisitions of the land surface. However, cloud-cover and phenological variability limit the ability to provide annual global updates of forest extent and change. The only other satellite system that can provide global coverage of the land surface at moderate resolution is the ALOS PALSAR radar instrument, which also includes an annual acquisition strategy for the global land surface (Rosenquist et al., 2007). However, large area forest-change mapping using radar data has not yet been implemented.

A variant of the satellite-based approach to land-cover change combines remote sensing-based information on recent land-cover change with regional tabular statistics,

such as from FAO, to reconstruct spatially explicit land-cover reconstructions covering more than the satellite era (Ramankutty and Foley, 1999; Pongratz et al., 2008; Klein Goldewijk, 2001). Two spatial data sets, in particular, have been used in most of the analyses included in Fig. 1: the SAGE data set, including cropland areas from 1700–1992 (Ramankutty and Foley, 1999), and the HYDE data set, including both cropland and pasture areas (Klein Goldewijk, 2001). These data sets have been updated and extended to the preindustrial past (Pongratz et al., 2008; Klein Goldewijk et al., 2011). Their differences account for about a 15 % difference in flux estimates over the period 1850–1990 (Shevliakova et al., 2009) and 1920–1990 (Fig. 1; Table 1).

2.2.3 Satellite data on fires

A third approach, applied so far only in tropical forests, uses satellite detection of fires in forests to estimate emissions from deforestation (van der Werf et al., 2010). The approach provides an estimate of gross forest loss but does not identify uses of land where fire is absent, for example, wood harvest. Nor does it distinguish between intentional deforestation fires and escaped wildfires. The approach combines estimates of burned area (Giglio et al., 2010) with complementary observations of fire occurrence (Giglio et al., 2003). At province or country level, clearing rates calculated this way capture up to about 80 % of the variability and also 80 % of the total clearing rates found by other approaches (Hansen et al., 2008a; INPE, 2010). One advantage of the fire-counting approach is that it allows for an estimate of interannual variability (see Sect. 7, below).

2.3 Carbon stocks and changes in them

Three approaches have been used to estimate carbon density (Mg C/ha) and changes in carbon density as a result of LULCC: non-spatial literature values, satellite-based estimates, and modeled estimates.

2.3.1 Selected field studies

One method uses ground-based measurements reported in forestry and agricultural statistics and the ecological literature. Inventory data are available on the carbon density of vegetation and soils in different ecosystem types, and the changes in them following disturbance or management. These data can be used with data on changes in land cover to track changes in carbon using empirical bookkeeping models. For example, conversion of native vegetation to cropland (i.e. cultivation) causes 25–30 % of the soil organic carbon in the top meter to be lost (Post and Kwon, 2000; Guo and Gifford, 2002; Murty et al., 2002). The conversion of lands to pastures, generally not cultivated, has less of an effect on soil carbon, on average. This approach is appropriate for non-spatial models. It assigns an average carbon density for biomass and for soils to all land within a small number of particular ecosystem types (e.g. deciduous forest, grassland). Considerable uncertainty arises because, even within the same forest type, the spatial variability in carbon density is large, in part because of variations in soils and microclimate, and in part because of past disturbances and recovery. Furthermore, the literature-based estimates of carbon density are representative of a specific time and do not capture changes in carbon density that may arise from indirect anthropogenic or natural effects.

2.3.2 Satellite-based estimates

A second approach uses new satellite techniques to estimate aboveground carbon densities. Examples of mapping aboveground carbon density over large regions include work with MODIS (Houghton et al., 2007), multiple satellite data (Saatchi et al., 2007), radar (Treuhaft et al., 2009), and lidar (see Goetz et al., 2009, for a review). While the accuracy is lower than site-based inventory measurements (inventory data are generally used to calibrate satellite algorithms), the satellite data are far less intensive to collect, can cover a wide spatial area, and thus can better capture the spatial variability in aboveground carbon density. By assigning a specific carbon density to

the forests actually deforested, this second approach increases the accuracy of flux estimates over the non-spatial approach described above (Baccini et al., 2012).

The capability of measuring changes in carbon density through monitoring is in its infancy, but such a capability would enable a method for estimating carbon sources and sinks that is more direct than identifying disturbance first, and then assigning a carbon density or change in carbon density (Houghton and Goetz, 2008). The approach would require ecosystem models and ancillary data to calculate changes in soil, slash, and wood products, and estimation of change, by itself, would not distinguish between deliberate LULCC activity and indirect anthropogenic or natural drivers. Nevertheless, estimation of change in aboveground carbon density has clear advantages for calculating sources and sinks of carbon.

2.3.3 Modeled estimates

A third approach uses process-based ecosystem models that calculate internally the carbon density of vegetation and soils in different types of ecosystem based on climate drivers and other factors within the models (see e.g. McGuire et al., 2001; Friedlingstein et al., 2006 for model intercomparisons). These models simulate spatial and temporal variations in ecosystem structure and physiology. Models differ in detail with respect to number of plant functional types (e.g. tropical evergreen forest, temperate deciduous forest, grassland) and number of carbon pools (e.g. fast and slow decaying fractions of soil organic matter). They simulate changes in carbon density by accounting for disturbances and recovery, whether natural or anthropogenic.

Net primary productivity is simulated in these ecosystem models as a function of the vegetation or plant functional type (PFT), local radiative, thermal, and hydrological conditions of the soil and the atmosphere, as well as the atmospheric composition. Soil organic matter decomposition is commonly controlled by temperature and soil moisture. The ecosystem models therefore respond to changes in climate and atmospheric composition. The models emphasize different aspects of ecosystem dynamics, with some accounting for competition between PFTs, nutrient limitation, and natural disturbances.

Anthropogenic land-cover change is usually prescribed from maps based on spatially explicit data sets, such as HYDE or SAGE. The land-cover change leads to a change in the fraction of PFT and a subsequent re-allocation of carbon to the atmosphere and to soil and product pools, where carbon decomposes with different turnover rates. Models differ widely with respect to implementation of land use (management), e.g. wood harvest, grazing, and other management activities. Regrowth follows abandonment of managed land, with some models accounting for degradation and succession. In the absence of detailed information on land conversion, specific allocation rules have to be applied to determine which natural vegetation type is reduced or expanded when managed land expands or is abandoned. Common rules are a proportional reduction of natural vegetation (Pitman et al., 2009) or a preferential allocation of pasture to natural grassland (Pongratz et al., 2008).

In contrast to bookkeeping models that specify changes in soil and vegetation carbon density based on a limited number of observations, process-based models determine internally vegetation and soil carbon density and changes in them. Both NPP and soil decomposition adjust over time in response to climate change or the fertilizing effects of changes in atmospheric CO_2 and N. The process-based models can therefore reflect much greater spatial and temporal variability in carbon density and response to environmental conditions than bookkeeping models, but their modeled carbon stocks may differ markedly from observations.

The sensitivity of carbon fluxes to the choice of model has been assessed in two studies. McGuire et al. (2001) applied four different process-based ecosystem models to similar data on cropland expansion; resulting land-cover emissions ranged from 0.6 to 1.0 Pg C yr^{-1} for the 1980s or from 56 to 91 Pg C for 1920–1992 (Fig. 1). Reick et al. (2010) applied a process-based model (JSBACH) and a bookkeeping approach (based on Houghton, 2003) to identical LULCC data and found that land-cover emissions were 40 % higher for the bookkeeping approach than the process-based approach (153 vs. 110 Pg C for 1850–1990) (see Fig. 1 and Table 1). The difference could be attributed almost entirely to differences in soil carbon changes; the

bookkeeping model assumed a 25 % loss of soil carbon to the atmosphere, while the process-based model calculated soil carbon changes based on changes in NPP and the input of organic material associated with the change in land use. Differences in the way models treat environmental change is addressed in Sect. 6.

2.3.4 Carbon emissions from fires

When satellite-based observations of fires in tropical forests are used to estimate rates of deforestation, the associated emissions of carbon are estimated by combining the fire-determined clearing rates with modeled carbon densities (van der Werf et al., 2010). Aboveground carbon densities are modeled (as in Sect. 2.3.3 above), but the changes in carbon density as a result of fire are calculated differently from the methods described above. The fraction of aboveground biomass lost to fire is based on a predefined range of combustion completeness using literature values and a scaling factor based on the fire persistence. This metric describes how many times a fire is seen in the same grid cell, and is related to the completeness of conversion; multiple fire events are needed for complete removal of biomass, resulting in high fire persistence (Morton et al., 2008) and high combustion completeness (van der Werf et al., 2010).

Over the period 1997–2010, average fire emissions from deforestation and degradation in the tropics with this approach were 0.4 Pg C yr^{-1}, with considerable uncertainty. Fires from peatlands added another 0.1 Pg C yr^{-1} (Sect. 5.1), for a total of 0.5 Pg C yr^{-1}. This estimate does not include emissions from respiration and decay of residual plant material and soils, nor does it account for changes in land use that do not rely on fire. To account for decay, fire emissions were doubled (Barker et al., 2007; Olivier et al., 2005), yielding an annual average estimate of ~1 Pg C yr^{-1}, in line with other estimates (Fig. 1), although none of these global estimates included emissions from drained and burned peatlands. Future research is needed to determine the exact ratio between fire and decay, something that is highly variable depending on post-deforestation land use. The main advantage of using fire to study deforestation emissions is that the fire emissions can be constrained using emitted carbon monoxide, which is routinely monitored by satellites and provides a much larger departure from background conditions than emitted CO_2 (e.g. van der Werf et al., 2008).

The approach underestimates carbon emissions for uses of land, such as wood harvest, that do not involve fire; and it overestimates LULCC carbon emissions if they include natural fires. Changes in forest area as determined from satellite data are not clearly attributable to management, as opposed to natural, processes. By definition, the sources and sinks of carbon for LULCC should not include the sources and sinks from natural disturbances and recovery. The latter are part of the residual terrestrial net flux. Fires, in particular, are difficult to attribute to natural processes, indirect effects (e.g. anthropogenic climate change), or direct management. The point here is that natural disturbances and recovery may be accidentally included in satellite-based analyses of LULCC.

3 Components of the annual flux of carbon from LULCC

The net flux of carbon from LULCC consists of several component fluxes that are not treated consistently among analyses, adding to the differences among flux estimates. To help illustrate the effects of these components, it is helpful to distinguish the net annual flux of carbon from the gross sources and sinks that comprise it. Using Houghton's analysis (the same as reported in Friedlingstein et al., 2011) as an example, the mean net flux of carbon from LULCC was a global source of 1.1 $\mathrm{Pg\,C\,yr^{-1}}$ over the period 2000–2009. Gross sources and sinks of carbon were about three times greater (Fig. 2a, b) and probably underestimated because deforestation was driven by net (rather than gross) changes in agricultural area, thereby underestimating the areas of secondary forests.

3.1 Instantaneous versus delayed fluxes

All estimates include and distinguish between instantaneous (emissions in the year of the disturbance) and delayed carbon fluxes. The loss of vegetation and soil carbon with LULCC is allocated to pools with different turnovers, and the fractions of initial carbon density assigned to these different turnovers vary among analyses. For example, burning releases carbon to the atmosphere immediately, while soils and products decay at different rates. While this difference does not affect cumulative emissions over a long time period, short-term emission fluxes can vary substantially (Ramankutty et al., 2007).

The fraction of biomass removed as a result of LULCC varies depending on the land use following clearing (Morton et al., 2008). Mechanized agriculture generally involves more complete removal of above- and below-ground biomass than clearing for small-scale farming or pasture. For example, in the southern Amazon state of Mato Grosso, estimated average emissions for 2001–2005 were 116 $\mathrm{Mg\,C\,ha^{-1}}$ when forests were converted to cropland and 94 $\mathrm{Mg\,C\,ha^{-1}}$ when they were converted to pasture (DeFries et al., 2008). Incorporating post-clearing land cover in estimating carbon emissions from land-use change will reduce uncertainties (Galford et al., 2010).

3.2 The importance of legacy fluxes

The existence of delayed fluxes implies that estimates of current fluxes must include data on historical land-cover activities and associated information on the fate of cleared carbon. However, such historical data are not included in all analyses, especially in studies using remote-sensing data where information is available only since the 1970s at best. This leads to the question of how far back in time one needs to conduct analyses in order to estimate current emissions accurately, or, alternatively, how much current emissions are underestimated by ignoring historical legacy fluxes. The answer depends on various factors including: (1) the rates of past clearing; (2) the fate of cleared carbon (including combustion completeness, repeat fires, etc.); (3) the fate of product and slash pools; and (4) the rate of forest growth following harvest or agricultural abandonment. If the rate of clearing in historical time periods is negligible, it is clear that legacy fluxes will be small. If most of the carbon cleared during previous land uses is burnt (and immediately lost to the atmosphere during those historical times), legacy fluxes will also be small. However, if a significant amount of historically cleared carbon remains in the soil to decompose or is turned into products which oxidize slowly, legacy fluxes will be higher today (unless soil decomposition rates or product oxidation rates are also high). The same reasoning applies to rates of growth of secondary forests.

Ramankutty et al. (2007) explored these issues using a sensitivity analysis in the Amazon. Their "control" study used historical land-use information since 1961, assumed a constant annual fraction of 20 % of cleared carbon being burnt, 70 % going to slash pools, 8 % to product pools, and 2 % to elemental carbon, and calculated annual actual fluxes from 1961 to 2003. When they repeated the analysis ignoring historical land use prior to 1981, they underestimated the 1990–1999 emissions by 13 %, while ignoring data prior to 1991 underestimated emissions by 62 %. However, if the assumption of the fate of cleared carbon was altered to 70 % burnt annually and 20 % left as slash, the underestimated emissions for ignoring pre-1981 data and pre-1991 data went down to 4 % and 21 %, respectively.

Globally, the contribution of instantaneous and legacy fluxes to the mean net flux 2000–2009 is shown in Fig. 2c. Instantaneous (fast) and legacy effects contribute about equally to gross emissions in this study. In contrast, gross sinks are almost entirely legacy fluxes, resulting from the uptake of carbon by secondary forests established in previous years following harvests and agricultural abandonment.

Most studies of LULCC have estimated the "actual" carbon flux, composed of legacy fluxes from past LULCC and instantaneous fluxes from current LULCC. While this approach is relevant for understanding the effects of LULCC on atmospheric carbon dioxide concentrations, a "committed" flux approach may be useful in some cases, e.g. for comparing alternative choices of land-use activities with regard to their total anticipated

emissions (Fearnside, 1997). The committed flux cumulates all emissions related to a specific land-use activity, both instantaneous and delayed emissions that will occur in the future, over a given time horizon. It can thus be calculated without knowing historical land-use changes. Actual and committed approaches have different intended uses, and they should not be directly compared, as demonstrated by Ramankutty et al. (2007).

4 Additional LULCC processes not included in all analyses

As discussed above (Sect. 2), variability in the estimates of flux from LULCC results, in large part, because of differences in data used to estimate deforestation rates and carbon density (see also Houghton, 2005, 2010). The variability also results from the types of land use included. All of the analyses reviewed here have included deforestation, either with satellite data or by inferring changes in forest area by combining data on expansion and abandonment of agricultural area (cropland and pasture) with information on natural vegetation (the latter approach also accounts for carbon fluxes from conversion of non-forest natural vegetation). Additional fluxes, not included in all of the analyses in Fig. 1, are outlined in the following section.

4.1 Forest degradation

The net flux of carbon from LULCC is not the same as "emissions from deforestation", although the terms are used interchangeably in the literature. A major difference among the estimates reviewed here is whether or not they included wood harvest and/or shifting cultivation, both of which reduce the carbon density of forests without changing forest area, a change defined here as forest degradation.

Logging in Amazonia, for example, added 15–19 % to the emissions from deforestation alone (Huang and Asner, 2010). For all the tropics, harvests of wood and shifting cultivation, together, added 28 % to the net emissions calculated on the basis of land-cover change alone (Houghton, 2010). They added 32–35 % to the global net flux from deforestation (Shevliakova et al., 2009). These last two estimates of carbon loss are net losses, including both the losses of carbon from oxidation of wood products and logging debris and the uptake of carbon in secondary forests recovering from harvest. Those analyses that have not included wood harvest and shifting cultivation may underestimate the net flux by 25–35 %.

It should be noted that rotational land uses are a source of carbon only if the activity or the area involved is increasing. Constant rates of logging (and subsequent recovery) should eventually lead to a net flux of zero. Declining rates will lead to a temporary net sink.

Using Houghton's bookkeeping method over the period 2000–2009, the net emissions from forest degradation accounted for about 11 % of the net flux (Fig. 2d). On the other hand, they accounted for about 66 % of gross emissions (Fig. 2e). Not surprisingly, the gross sources (decay of debris and wood products) and sinks (regrowth) from wood harvest and shifting cultivation are large compared to the net flux. Indeed the gross fluxes from these rotational land uses explain most of the gross fluxes. Net and gross emissions from deforestation are not identical in this accounting because the net land-use flux includes the effects of both deforestation and reforestation.

4.2 Agricultural management

The changes in soil organic carbon (SOC) that result from the cultivation of native soils are included in most analyses, but the changes in SOC that result from cropland management, including cropping practices, irrigation, use of fertilizers, different types of tillage, changes in crop density, and changes in crop varieties, are not generally included in global analyses. Studies have addressed the potential for management to sequester carbon, but fewer studies have tried to estimate past or current carbon sinks. One analysis for the US suggests a current sink of 0.015 $\mathrm{Pg\,C\,yr^{-1}}$ in croplands (Eve et al., 2002), while a recent assessment for Europe suggests a small net source or near-neutral conditions (Ciais et al., 2010). In Canada, the flux of carbon from cropland

management is thought to be changing from a net source to a net sink, with a current flux near zero (Smith et al., 2000). The effects of agricultural management have also been included in regional analyses at high spatial resolution (e.g. Kutsch et al., 2010; West et al., 2010). Globally, the current flux is uncertain but probably not far from zero (Table 2).

4.3 Fire management

The emissions of carbon from fires associated with deforestation are included in the emissions of carbon from LULCC, but wildfires have been ignored, first, because they are not directly a result of management and, second, because, in the absence of a change in disturbance regimes, the emissions from burning are presumably balanced by the accumulations in ecosystems recovering from fire. Fire management (outside deforestation), on the other hand, is a management activity that affects carbon storage, yet it has been largely ignored in global analyses of LULCC despite the fact that fire exclusion, fire suppression, and controlled burning are practiced in many parts of the world. Fire management may cause a terrestrial sink in some regions (Houghton et al., 1999; Marlon et al., 2008) and a source in others. In particular, the draining and burning of peatlands in Southeast Asia are thought to add another 0.3 $\mathrm{Pg\,C\,yr^{-1}}$ to the net emissions from land-use change (not included in the estimates reported here) (see Sect. 5.1.1) (Hooijer et al., 2010).

4.4 Land degradation

Most forms of management other than harvest of wood have received little attention in global estimates of carbon flux from LULCC. An exception is the net release of carbon estimated to have occurred in China between 1900 and 1980 (Houghton and Hackler, 2003). During this interval, the net loss of forest area was more than three times greater than the net increase in croplands and pastures. Assuming the data are accurate, the loss may have resulted from unsustainable harvests, from deliberate removal of forest cover (for protection from tigers or bandits), and from the deleterious effects of long-term intensive agriculture on soil fertility. Annual emissions of carbon were between 0.1 and 0.3 Pg C yr^{-1} during this interval but very uncertain. The area in degraded lands is rarely enumerated (Oldeman, 1994), yet the losses of carbon may be significant (Lal, 2001).

5 Additional LULCC processes not included in any analyses

The three processes described below are not included in any of the global estimates of LULCC. The first process will increase estimates of net carbon emissions, the second is likely to decrease estimates, and the third is uncertain as to its net effect.

5.1 Peatlands, wetlands, mangroves

5.1.1 Drainage and burning of peatlands

Peatlands occur on all continents in the tropics, but the largest tropical peatlands and that that have received most attention from a carbon perspective are those in Southeast Asia, mostly in Indonesia. Here peatlands are overgrown with forests that are often called peat swamp forests. Peatlands cover only a small fraction of the Earth's surface but store large amounts of carbon; estimates start at 42 Pg C for SE Asian peatlands compared to 70 Pg C for Amazon aboveground biomass (Hooijer et al., 2010). While peatlands in general are a carbon sink, drainage of these peatlands for agriculture and forestry often results in emissions, either via fire or via decomposition. In Borneo, peat swamp forests experienced deforestation rates of about 2.2 % yr^{-1} between 2002 and 2005, higher than other types of forests (Langner et al., 2007).

Fire emissions during the 1997–1998 El Niño in Indonesia were first estimated to be between 13 and 40 % of global fossil fuel emissions (Page et al., 2002). More recent studies (Duncan et al., 2003; van der Werf et al., 2008) confirmed the significant

contribution of peatlands to the global carbon cycle, and indicated that emissions were probably close to the lower estimate of Page et al. (2002). Fire emissions from the burning of peatlands are generally lower than during the 1997–1998 El Niño when the region experienced a long and intense dry season, but on average they are still comparable to fossil fuel emissions in the region (van der Werf et al., 2008).

Emissions of carbon from oxidation of peatlands as a result of drainage are not as well studied, yet may be more important. Quantifying these fluxes requires extensive fieldwork to monitor annual changes in peat extent, although new LIDAR-based estimates may provide estimates of the loss rates of peatlands when focusing on a longer timeframe or for larger burns (Ballhorn et al., 2009). The most extensive estimate so far is probably by Hooijer et al. (2006) who estimated annual emissions of between 97 and 233 Tg C yr^{-1} for all of Southeast Asia, with 82 % from Indonesia. These emissions vary less from year to year than fire emissions do, although oxidation rates are related to water table depth and thus to precipitation rates, which vary considerable from year to year (Wösten and Ritzema, 2001).

The combined emissions from both oxidation through drainage (165 ± 68 Tg C yr^{-1}) and fire (124 ± 70 Tg C yr^{-1}) in Southeast Asian peatlands are 289 ± 138 Tg C yr^{-1} (or 0.3 Pg C yr^{-1}) (Table 2) (Hooijer et al., 2010; van der Werf et al., 2008). The estimate is likely a global underestimate because other areas besides Southeast Asia may also be exploiting peatlands (Lähteenoja et al., 2009).

5.1.2 Mangroves

A recent study estimated that deforestation of mangroves released 0.02 to 0.12 Pg C yr^{-1} (Donato et al., 2011). The high releases resulted from the carbon-rich soils, which range from 0.5 to more than 3 m in depth. The carbon emissions from these and other wetlands have not been included in global estimates of emissions from land-cover change.

5.2 Erosion/redeposition

Reviews have consistently shown that 25–30 % of the soil organic carbon in the top meter is lost with cultivation (Post and Kwon, 2000; Guo and Gifford, 2002; Murty et al., 2002). This loss is generally assumed to have been released to the atmosphere. However, some of it may have been moved laterally to a different location (erosion), perhaps buried in an anoxic environment, and thereby sequestered. Comparison of erosion rates with the amount of organic carbon in freshwater sediments suggests that some of the carbon lost through erosion accumulates in riverbeds, lakes, and reservoirs (Stallard, 1998; Smith et al., 2001; Berhe et al., 2007). Recent estimates suggest that as much as 0.6 Pg C may be buried this way (Tranvik et al., 2009; Aufdenkampe et al., 2011). To the extent that soil carbon is not released to the atmosphere, but moves laterally, the emissions reported here, e.g. by Houghton (2003) and Shevliakova et al. (2009), may be overestimated. If this flux is part of a steady-state flux related to farming, the sink of 0.6 Pg C yr^{-1} helps explain a significant fraction of the residual terrestrial sink.

5.3 Woody encroachment

The expansion of trees and woody shrubs into herbaceous lands is increasing carbon storage on land in many regions. Scaling it up to a global estimate is problematical, however (Scholes and Archer, 1997; Archer et al., 2001), in part because the areal extent of woody encroachment is unknown and difficult to measure (e.g. Asner et al., 2003). Also, the increase in carbon density of vegetation observed with woody encroachment is in some cases offset by losses of soil carbon (Jackson et al., 2002). In other cases the soils may gain carbon (e.g. Hibbard et al., 2001) or show no discernable change (Smith and Johnson, 2003). Finally, woody encroachment may be offset by its reverse process, woody elimination, an example of which is the fire-induced spread of cheatgrass (*Bromus tectorum*) into the native woody shrublands of the Great Basin in the western US (Bradley et al., 2006). The net effect of woody encroachment

and woody elimination is, thus, uncertain, not only with respect to net change in carbon storage, but also with respect to attribution. It may be an unintended effect of management, or it may be a response to indirect or natural effects of environmental change.

6 Treatment of environmental change

While bookkeeping models use rates of growth and decay that are fixed for different types of ecosystems, process-based models simulate these processes as a function of climate variability and trends in atmospheric composition. Because effects are partly compensating (e.g. deforestation under increasing CO_2 leads to higher emissions because CO_2-fertilization has increased carbon stocks, but regrowth is also stronger under higher CO_2 concentrations), a CO_2 fertilization effect is not likely a major factor in differences among emission estimates (McGuire et al., 2001). Over the industrial era, the combined effects of changes in climate and atmospheric composition by one estimate have increased LULCC emissions by about 8 % (Pongratz, this study, Fig. 1, and Table 1).

To compare the emissions determined from bookkeeping models that do not include the effects of a temporally varying environment with emissions determined from process-based models that do include the effects (Piao, Pongratz LUC + CO_2, and Van Minnen in Fig. 1), the process-based models are usually run with and without LULCC, and the difference between the two runs is taken to yield the net effects of LULCC. However, if CO_2 fertilization has a greater effect on growing forests than on grown forests, then this CO_2 effect is included in the estimated LULCC emissions. There are doubtlessly other interactions, as well, between environmental changes and management, making comparisons and attribution difficult.

There is another (indirect) effect of deforestation. The woody biomass of forests has a greater capacity than the herbaceous biomass of crops and grassland to store carbon, and this capacity is reduced as forests are converted to non-forest lands. In models, the strength of this effect depends on the atmospheric CO_2 concentration as well as the area of forest lost. This effect has been called the “loss of additional sink capacity” (Pongratz et al., 2009), or, including also delayed emissions from past land use, the “net land-use amplifier effect” (Gitz and Ciais, 2003) and “replaced sinks/sources” (Strassmann et al., 2008). Estimates vary from ~4 Pg C for 1850–2000 (Pongratz et al., 2009) and 8.5 Pg C for 1950–2100 (Sitch et al., 2005), to ~0.2 Pg C yr^{-1} for 1990–2000 (Strassmann et al., 2008) and 125 Pg C for 1700–2100 (Gitz and Ciais, 2003) including delayed emissions.

Note that none of the estimates of the carbon flux from LULCC in Fig. 1 includes the fluxes driven by environmental effects on natural vegetation, or those ecosystems that are not affected by LULCC. Both managed and natural ecosystems may be responding similarly to environmental changes, but only the net source/sink from those lands affected by LULCC should be included in comparing estimates of the flux of carbon attributable to LULCC.

7 Interannual variability and trends

Since most assessments of LULCC have focused on 5 to 10-yr changes, interannual variability has not received much attention. However, satellite-based observations of forest-cover loss and fires demonstrate the interannual variability in deforestation rates (Fig. 3). This variability may be driven by commodity prices, institutional measures, and climate conditions. Over the period 2001–2004 clearing rates in the Brazilian state of Mato Grosso were correlated with soy prices (Morton et al., 2006). Longer and more extreme dry seasons, allowing for a more effective use of fire, have been linked to higher clearing rates in Indonesia (van der Werf et al., 2008) and the Amazon (Chen et al., 2012). The large climate shifts related to ENSO in Southeast Asia contribute to large interannual variability, with emissions during dry El Niño years being one or even two orders of magnitude larger than emissions during wet La Niña years.

Regarding a trend in global emissions from LULCC, no trend stands out in the family of curves in Fig. 1. Nevertheless, those analyses that extend to 2010 suggest a recent downturn in net emissions, not statistically significant but consistent with decreased rates of deforestation reported in the FAO 2010 Forest Resources Assessment and with declining rates of deforestation observed in the two countries with the highest rates (Fig. 3). The recent downward trend in net emissions may thus be real. As discussed above (Sect. 2.2.1), revisions in the rates of tropical deforestation reported in the FRAs (FAO, 2001, 2006, 2010) contribute substantially to the variability of flux estimates. The revisions make it difficult to detect a trend, especially if different analyses have used different assessments to drive deforestation. The latest FRA (2010), for example, lowered rates of deforestation for the period 2000–2005, especially in tropical Asia. If these most recent data are more accurate than previous estimates, then all estimated emissions based on the earlier estimates of deforestation are too high between 2000 and 2005, and they may distort or obscure a downward trend in emissions.

8 Summary of uncertainties

The contributions of different factors to the uncertainty of flux estimates are summarized in Table 2 along with estimates of the fluxes from activities or processes that are (1) not included in all analyses and (2) not included in any analyses. The rate of change in land cover appears to be the largest single source of uncertainty (±0.4 Pg C yr^{-1}), but this observation, based on Houghton (2005), is dated. The decadal standard deviation reported here is ~0.2 yr^{-1} for the 1990–2009 period. Better reporting of deforestation rates by the FAO has narrowed the range of estimates cited by Houghton (2005) and the IPCC (2007) and is likely to reduce the uncertainty still more in the future. A similar reduction in the uncertainty of biomass estimates is also likely.

Overall, the error for emissions of carbon from LULCC is estimated to be ±0.5 Pg C yr^{-1}. Most of that uncertainty comes from processes not considered in the analyses reviewed here (Table 2). By chance, the effects of these processes seem to be offsetting and thus unlikely to bias estimates of flux from LULCC. That observation has considerable uncertainty, however. The estimated errors in Table 2 are often little more than guesses, obtained from regional or national studies (e.g. Houghton et al., 1999; Houghton and Hackler, 2006) but never evaluated globally. The estimates (both fluxes and errors) for these processes are tentatively advanced here for purposes of discussion.

9 Conclusions

Scientists working on defining the role of terrestrial ecosystems in the global carbon cycle recognized long ago the importance of satellite data for documenting changes in forest area (Woodwell et al., 1984). Satellite data for carbon density are also becoming available. The co-location of land-cover change and biomass density data, both at relatively high resolution, offers a new opportunity for estimating terrestrial sources and sinks of carbon at greater accuracy, reducing the potential bias from interaction between the two variables. Recent analyses have taken advantage of this opportunity (Baccini et al., 2012), although not at a spatial resolution necessary for capturing LULCC. But the analyses are underway; they will be increasingly used in the future. Challenges include identification of the fate of cleared land, attribution for observed changes in biomass density, and accounting for the all of the carbon (i.e. changes in belowground carbon density and harvested wood products).

Another advance in reducing variability among estimates might include an inter-comparison of the models used to estimate LULCC. Dealing quantitatively with the differences among approaches (as opposed to qualitatively, as discussed here) might benefit from a coordinated, systematic inter-comparison, where models used the same set of input variables (e.g. McGuire et al., 2001). The IPCC's Fifth Assessment Report is a step in this direction, although it has been shown that the implementation of the same LULCC data may vary greatly across models (Pitman et al., 2009). Other adjustments might be made off line. For example, estimated emissions from analyses not

considering wood harvest might be increased by 20–35 %, or those analyses not explicitly including emissions from the draining and burning of tropical peatlands might be increased by 0.1–0.3 $\mathrm{Pg\,C\,yr^{-1}}$. "Corrections" for CO_2 or nitrogen feedbacks would be more difficult, as the feedbacks may increase both sources and sinks, with an unclear effect on the net balance.

More important than comparisons among models, of course, is comparisons of model estimates with data, a non-trivial comparison when emissions over large regions are concerned. The global carbon budget offers little constraint as long as the residual terrestrial sink is calculated by difference. One goal of the research is to explain more and more of this residual sink or, to put it another way, to make it vanish. Process-based terrestrial models, collectively, may be able to explain the residual terrestrial sink (Le Quéré et al., 2009), but differences among model estimates under future environmental conditions do not inspire confidence that the important processes are fully understood (Friedlingstein et al., 2006).

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Table 1a. Key characteristics of the data sets shown in Fig. 1. Note that several studies provide a range of different estimates of land-use emissions; the datasets shown in this study were chosen as the ones closest to a bookkeeping approach or to isolate certain processes.

Study (Fig. 1)	Reference	Approach	LULCC types	LULCC source	Carbon fluxes	Beginning of accounting (AD)[a]	Spatial detail[b]	Emissions 1920 to 1999 (Pg C yr^{-1})	Emissions 1990 to 1999 (Pg C yr^{-1})
Achard	Achard et al. (2004)	Bookkeeping model	De/reforestation, forest degradation, peat fires	Remote sensing, FAO Remote Sensing Survey	Actual direct	1990	Explicit (only tropics)	–	1.10
Arora	Arora and Boer (2010)	Process model (CTEM)	Cropland	Ramankutty and Foley (1999)	Actual direct	1850	Explicit	0.92	1.06
DeFries	DeFries et al. (2002)	Bookkeeping model	De/reforestation	Remote sensing	Actual direct	1982	Explicit (only tropics)	–	0.90
Houghton	Houghton (2010)	Bookkeeping model	Ag[c] incl.	FAO and shifting cultivation in Latin America/tropical Asia, and wood harvest	Actual direct national censuses	1850	Regional	1.21	1.50
Piao	Piao et al. (2009)	Process model (ORCHIDEE)	Ag Foley (1999) (cropland), HYDE2.0 (pasture), IMAGE (after 1992)	Ramankutty and including effects of observed CO_2 and climate change	Actual direct	1900	Explicit	1.31	1.24
Pongratz LUC	Pongratz et al. (2009)	Process model (JSBACH)	Ag al. (2008)[d]	Pongratz et	Actual direct	800	Explicit	0.90	1.14
Pongratz LUC + CO_2	Pongratz et al. (2009)	Process model (JSBACH)	Ag al. (2008)[c]	Pongratz et including effects of simulated CO_2 and climate change	Actual direct	800	Explicit	0.99	1.30
Reick process	Reick et al. (2010)	Process model (JSBACH)	Ag	Pongratz et al. (2008)[d]	Actual direct	800	Explicit	1.03	–
Reick bookkeeping	Reick et al. (2010)	Bookkeeping model	Ag	Pongratz et al. (2008)[d]	Actual direct	800	Explicit	1.34	–
Shevliakova HYDE/SAGE	Shevliakova et al. (2009)	Process model (LM3V)	Ag incl. shifting cultivation in tropics, and wood harvest	Hurtt et al. (2006)[e]	Actual direct	1700	Explicit	1.44	1.31

Table 1b. Continued.

Study (Fig. 1)	Reference	Approach	LULCC types	LULCC source	Carbon fluxes	Beginning of accounting (AD)[a]	Spatial detail[b]	Emissions 1920 to 1999 (Pg C yr^{-1})	Emissions 1990 to 1999 (Pg C yr^{-1})
Shevliakova HYDE	Shevliakova et al. (2009)	Process model (LM3V)	Ag incl. shifting cultivation in tropics, and wood harvest	Hurtt et al. (2006)[f]	Actual direct	1700	Explicit	1.28	1.07
Strassmann	Strassmann et al. (2008)	Process model (LPJ in BernCC)	Ag, urban	HYDE 2.0 adjusted	Actual direct	1700	Explicit	1.39	0.75
Stocker	Stocker et al. (2011)	Process model (LPJ in BernCC, updated since Strassmann, 2008)	Ag, urban	HYDE3.1 adjusted	Actual direct	10 000 BC	Explicit	1.31	0.93
Van Minnen	Van Minnen et al. (2009)	Process model (IMAGE2)	Ag, wood harvest	HYDE (ag), IMAGE2 (w.h.)	Actual direct including effects of CO_2, climate change, and management[d]	1700		1.16	1.33
Zaehle	Zaehle et al. (2011)	Process model (O-CN)	Ag, urban	Hurtt et al. (2006)	Actual direct	1700		1.32	0.97

[a] I.e. legacy emissions of earlier time periods not considered.

[b] Unless otherwise noted, studies considered all land area.

[c] "Ag" stands for changes in land cover caused by expansion or abandonment of agricultural area; agriculture includes both cropland and pasture.

[d] Based on SAGE cropland and SAGE pasture with rates of pasture changes from HYDE, preferential allocation of pasture on natural grassland.

[e] Based on SAGE cropland and HYDE pasture, proportional scaling of natural vegetation.

[f] Based on HYDE cropland and HYDE pasture, proportional scaling of natural vegetation.

[g] An "autonomous growth factor" approximates increase in plant productivity due to nitrogen fertilization and forest management changes.

Table 2. Summary of the factors contributing to uncertainty in estimates of emissions from LULCC and summary of processes missing from at least some of the analyses.

	Decadal uncertainty ($Pg\,C\,yr^{-1}$)	Reference
Uncertainty		
Land-cover change	±0.4	Houghton et al. (2005)*
Model and method	±0.2	McGuire et al. (2001); Reick et al. (2010)
Biomass	±0.3	Houghton et al. (2005)*
Processes included in some analyses		
Forest degradation	∼+0.4 ± 0.2	
Agricultural management	∼0 ± 0.2	
Fire Management	∼ −0.3 ± 0.2	
Land Degradation	∼+0.1 ± 0.2	
Processes included in none of the analyses		
Peatland Drainage	∼+0.3± 0.1	
Erosion/redeposition	∼ −0.6 ± 0.3	
Woody Encroachment	∼ −0.1 ± 0.3	

* Based on numbers in Table 2, including the preliminary FAO data used in Friedlingstein et al. (2010).

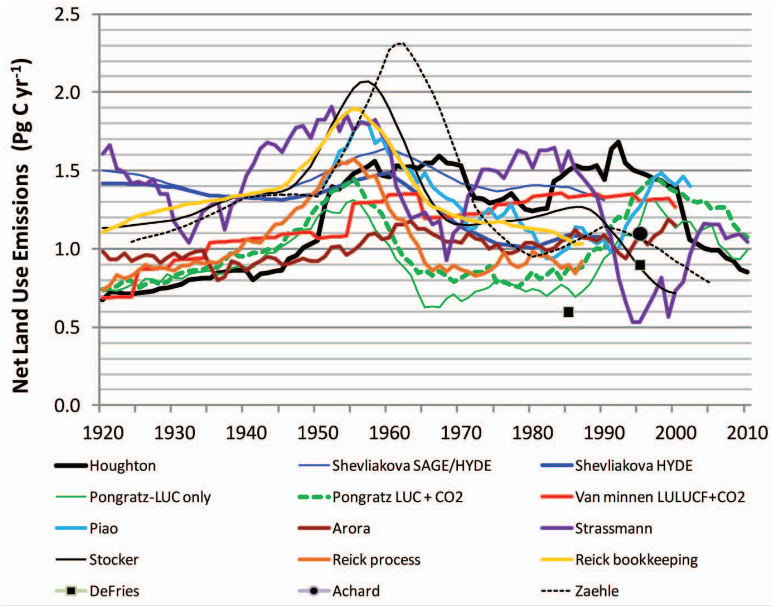

Fig. 1. Recent estimates of the net annual emissions of carbon from land use and land-cover change. The closed boxes (DeFries et al., 2002) and circle (Achard et al., 2004) represent 10-yr means for the 1980s or 1990s.

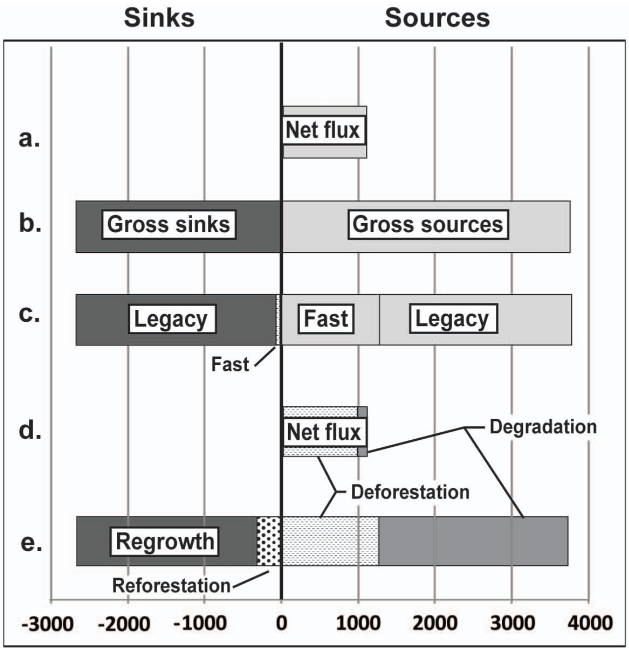

Fig. 2. Net and gross sources and sinks of carbon 2000–2009 attributable to different processes (from Houghton's analysis as reported in Friedlingstein et al., 2011). "Legacy" in 2c refers to the sinks (regrowth) and sources (decomposition) from activities carried out before 2000.

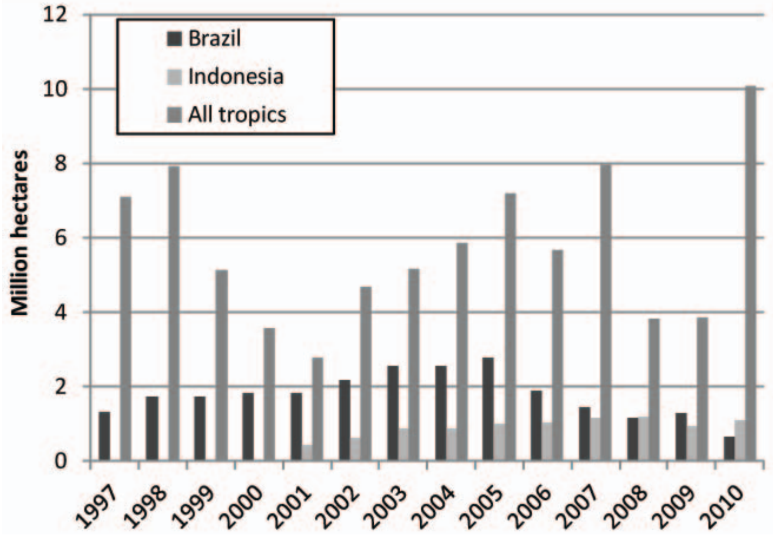

Fig. 3. Interannual variation in rates of deforestation in Brazil (dark bars) (INPE, 2010) in Indonesia (light bars) (Hansen et al., 2009 and updated) and in all tropical forests (van der Werf et al., 2010). The values for Brazil include only the loss of intact forest within the Legal Amazonia, while for Indonesia they include the loss of all forests meeting the definition 30 % cover and 5-meter-tall canopy at 60 m spatial resolution (approximately half of these Indonesian forests are intact). The pan-tropical estimates are based on burned area and active fire detections in forested areas.