**Title**

# The effect of land-use change on the net exchange rates of greenhouse gases: a global compilation of estimates

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**Abstract**

One of the environmental impacts of land-use change (LUC) is a change in the net exchange of the greenhouse gases (GHGs) carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$). Here we summarize global data of changes in soil organic carbon (SOC) stocks and soil $CH_4$ and $N_2O$ fluxes associated with LUC. We combine that with estimates of biomass carbon (C) stock changes and enteric $CH_4$ emissions following LUC. Data were expressed in common units by converting net $CH_4$ and $N_2O$ fluxes to $CO_2$ equivalents ($CO_2$ eq) using established global warming potentials, and carbon-stock changes were converted to annual net fluxes by averaging stock changes over 100 years. Conversion from natural forest to cropland resulted in the greatest increase in net GHG fluxes, while conversion of cropland to secondary forest resulted in the greatest reduction in net GHG emissions. Specifically, LUC from natural forest to crop and grasslands led to net emissions of 7.6±1.3 (Mean ± 95% confidence intervals) and 6.2±0.8 t $CO_2$ eq $ha^{-1}$ $y^{-1}$ to the atmosphere, respectively. Conversely, conversion from crop and grasslands to secondary forest led to a net reduction in emissions by 5.7±4.7 and 3.6±0.7 t $CO_2$ eq $ha^{-1}$ $y^{-1}$, respectively. In all land-use changes involving forests, changes in biomass carbon dominated the overall change in net GHG emissions. A retrospective analysis indicated that LUC from natural forests to agricultural lands contributed a cumulative 1647 ± 521 Gt $CO_2$ eq between 1765 and 2005, which is equivalent to average emissions of 6.9±2.2 Gt $CO_2$ eq per year. This study demonstrates how specific LUCs can positively or negatively affect net GHG fluxes to the atmosphere.

## 1. Introduction

Fossil-fuel emissions are clearly the dominant factor responsible for the enhanced greenhouse effect (Forster et al., 2007), but land-use change (LUC) also leads to important additional greenhouse gas (GHG) exchanges between the atmosphere and the terrestrial biosphere (Houghton et al., 2012; Kirschbaum et al., 2013). Globally, 13 million ha were deforested annually between 1990 and 2009 (FAOSTAT, 2013), with annual mean global C emissions from land-use change estimated to be 4.0 Gt $CO_2$ $yr^{-1}$ between 1980 and 2000 (Houghton et al., 2012) and 4.1 Gt $CO_2$ $y^{-1}$ between 1870 and 2013 (Le Quéré et al., 2013). Over the same time period, areas of cropland and grassland have increased annually by 1 and 2 million ha, respectively (FAOSTAT, 2013). Such LUC is likely to have large environmental impacts, including changes in the net flux of $CO_2$, $CH_4$ and $N_2O$ through altered biogeochemical processes (Forster et al., 2007; Kirschbaum et al., 2012; Wang et al., 2012). The enhanced greenhouse effect is currently dominated by the increase in $CO_2$ concentration, which contributes a radiative forcing of about 1.66 W $m^{-2}$, and increases in $CH_4$ and $N_2O$ add a further 0.48 W $m^{-2}$ and 0.16 W $m^{-2}$, respectively (Forster et al., 2007). With on-going concern about global climate change, the effect of LUC on the emission of all these GHGs needs to be critically considered.

The effect of LUC on $CO_2$ fluxes is directly related to changes in soil organic carbon (SOC) and C in vegetation since any loss of biospheric C stocks increases atmospheric $CO_2$. While the changes in SOC following LUC are mainly attributable to shifts in the balance between carbon-input rates and specific decomposition rates of organic matter (e.g., Murty et al., 2002; Guo and Gifford, 2002; Don et al., 2011), soil erosion may play a role in erosion-prone landscapes (e.g., Lal, 2003; Post et al., 2004; Gaiser et al., 2008) and, where fire is associated with LUC, it may also deplete SOC stocks (e.g., van der Werf et al., 2006, 2010).

The effect of land-use change on $CH_4$ fluxes is related to enteric fermentation by grazing animals and any soil processes that produce or consume $CH_4$. The net $CH_4$ flux in the soil is the result of the balance between methanogenesis (microbial $CH_4$ production mainly under anaerobic

conditions) and methanotrophy (microbial $CH_4$ consumption) (Dutaur and Verchot, 2007; Kirschbaum et al., 2012). Methanogenesis occurs via the anaerobic degradation of organic matter while methanotrophy occurs by methanotrophs metabolizing $CH_4$ as their source of carbon and energy (Hanson and Hanson, 1996). Forest soils are generally the most active $CH_4$ sink, followed by grasslands and cultivated soils (e.g., Topp and Pattey, 1997; Le Mer and Roger, 2001; Dutaur and Verchot, 2007). In cases where LUC involves changes to or from grazed grasslands, there can be large changes in $CH_4$ emissions by enteric fermentation of grazing animals (e.g., Kelliher and Clark, 2010; Cottle et al., 2011). Other than for conversions including wetlands, including rice paddies, changing contributions from enteric fermentation are likely to dominate the overall change in net $CH_4$ emissions. Land-use changes that involve wetlands are special cases of land-use changes that can lead to large on-going changes in $CH_4$ emissions and soil organic C storage (e.g., Janssens et al., 2005). The issues associated with these land-use changes were too complex to be included as part of the present study and have therefore consciously been omitted. It would require a dedicated paper on its own to adequately deal with all GHG implications of land-use changes involving wetlands.

$N_2O$ is produced in soils through three main processes: 1) nitrification, the oxidation of ammonia ($NH_3$) to nitrate ($NO_3^-$) (Kowalchuk and Stephen, 2001); 2) denitrification, the stepwise conversion of $NO_3^-$ to $N_2O$ and ultimately $N_2$, by anaerobic bacteria that use $NO_3^-$ as electron acceptors for respiration under anaerobic conditions (Knowles, 1982); and 3) nitrifier denitrification by $NH_3$-oxidizing bacteria that convert $NH_3$ to $N_2O$ and $N_2$ (Wrage et al., 2001). N input, land use and its management, and climatic conditions are generally considered to be the major controlling factors of $N_2O$ fluxes in soils (e.g., Snyder et al., 2009; Smith, 2010; Kirschbaum et al., 2012).

There has been increasing interest in the effect of LUC on SOC, and previous review papers have comprehensively summarized the effect of various LUCs on SOC (e.g., Murty et al., 2002; Guo and Gifford, 2002; Laganiére et al., 2010; Don et al., 2011; Poeplau et al., 2011; Liao et al., 2012; Li et al., 2012). A growing number of studies have also reported the effect of LUC on $CH_4$

and $N_2O$ fluxes. This may reflect the current interest in the losses and gains of C, and the increase or decrease in the emission of other GHGs related to global climate change. However, we are not aware of any previous comprehensive and quantitative summary reports that have combined the effect of LUC on changes in biomass C, SOC, $CH_4$ and $N_2O$ fluxes. This review is novel in that it takes a comprehensive approach in dealing with the effect of LUC on the exchange of GHGs between land and atmosphere through quantifying changes in all these important fluxes. It thus tries to bring together as much of the published literature as we were able to obtain, summarize the findings, and express them in common and comparable units. The work thus tries to estimate the total GHG impact of specified LUCs based on empirical observations as far as they are available.

Our specific objectives were to: 1) summarize the effect of LUC on exchange of GHGs between the land and the atmosphere; 2) convert the individual net emissions from different gases to common units and derive total integrated net GHG impact related to each LUC; and 3) discuss the underlying mechanisms and drivers of responses.

## 2. Methodology

### 2. 1. Types of land-use change assessed in this study

In this study, we have considered the following types of LUC:

- Land-use change from natural forest to cropland, grassland, or secondary forest, or secondary forest to cropland
- Land-use change from cropland to grassland or secondary forest
- Land-use change from grassland to cropland or secondary forest.

Natural forest includes all naturally growing forests in tropical, temperate, and boreal regions. Secondary forests can be local indigenous forests that are naturally regenerating or forests planted for specific human purposes, and they may include indigenous or introduced species. Croplands exclude rice paddies, while grasslands include both extensively and intensively managed grasslands.

## 2. 2. Quantifying the impact of land-use change on net greenhouse gas exchange

The impact of LUC on net GHG exchange was determined through quantifying changes in biomass C, SOC, $CH_4$ production through enteric fermentation, and net soil fluxes of $CH_4$ and $N_2O$. They were expressed in common units of $CO_2$ equivalents through multiplication by the respective global warming potentials (GWP) of different GHGs.

### 2. 2. 1. Quantifying changes in biomass carbon stocks

Global average biomass C stocks in natural and secondary forests, including above and below-ground biomass, dead wood and litter, were estimated based on information available in FAO (2010) and WBGU (1998) (Table 1). Firstly, global average C stocks for all forests were calculated as 99.8 t C $ha^{-1}$ (FAO, 2010). This included both undisturbed natural and secondary forests, and, according to information compiled by WBGU (1998) biomass carbon stocks of secondary forests is, on average, about 50% of that of primary forests. Considering that 27.3% of global forests can be considered as natural and undisturbed (FAO, 2010), it follows that the global average C stock for all forests would consist of 156.8 t C $ha^{-1}$ in natural forests and 78.4 t C $ha^{-1}$ in secondary forests (Table 1). Biomass C stocks (including both shoot and root) in cropland and grassland (Table 1) were taken from IPCC (2001).

The change in biomass for a transition from one kind of land use *i* to land use *j* was then calculated simply as:

$$\Delta_{B,ij} = B_i - B_j \qquad \text{(Eq. 1)}$$

where $B_i$ and $B_j$ are the average biomass C stocks per unit of land of vegetation types *i* and *j*.

Table 2 gives estimated biomass C stock changes calculated based on these assumptions for various types of LUC.

A problem arises in that carbon-stock changes are one-off carbon-stock changes whereas changes in the flux of the other GHGs constitute on-going changes. Upon land-use change, carbon

stocks change only once. Upon deforestation, a site might lose carbon, especially if biomass carbon is burnt. That loss remains regardless of the length of time that one considers. In contrast, a change in $CH_4$ or $N_2O$ emissions is affective over each year that a land use remains under the new land use. So the relativities between the importance of carbon-stock changes versus on-going emission changes of $CH_4$ and $N_2O$ change very much with the length of time that is considered. To bring these changes to common units, we chose to analyse the changes over a time frame of 100 years, as this is a commonly used time frame in GHG accounting, such as in the calculation of GWPs. However, there is no substantive reason for choosing a 100-year calculation interval rather than any other. Had a longer integration interval been chosen, it would have reduced the inferred importance of carbon-stock changes while the numbers for net changes in $CH_4$ and $N_2O$ fluxes would have remained the same. Conversely, shortening the integration interval would have increased the inferred importance of carbon-stock changes. Greenhouse gas fluxes from these biomass C changes were then calculated simply as:

$$F_{b,ij} = (44/12)\, \Delta_{B,ij} / 100 \qquad \text{(Eq. 2)}$$

where $F_{b,ij}$ is the GHG flux (t $CO_2$ $ha^{-1}$ $y^{-1}$) due to biomass C changes of land-use transition *i* to *j*. The constants 44 and 12 are the molecular weights of $CO_2$ and C, respectively. The division by 100 apportions an overall one-off C-stock change equally over a period of 100 years.

### 2. 2. 2. Quantifying change of soil organic carbon stocks

The change of SOC stocks for LUC from land use *i* to land use *j* over 100 years ($\Delta_{S,ij}$) was estimated as:

$$\Delta_{S,ij} = S_i \times \Delta S_{ij(100)} \qquad \text{(Eq. 3)}$$

where, $S_i$ are the average pre-LUC soil-organic carbon stocks associated with land use *i*, and $\Delta S_{ij(100)}$ is the fractional SOC change estimated over 100 years, following a LUC from land use *i* to *j*.

The SOC in land prior to LUC and the change rates (Δ) of SOC in various types of LUC (Table 3; Fig. 1) were obtained by combining the global meta-data of Murty et al. (2002), Don et al.

(2011), Poeplau et al. (2011), and Power et al. (2011) that together include over 230 studies published from 1963 to 2010. As used here, these include data without bulk-density corrections. It should be noted, however, that errors introduced through uncorrected bulk-density changes lead to errors in opposite directions for measurements based on carbon stocks and carbon concentrations (e.g. Murty et al., 2002) so that across the range of observations, these errors partly negate each other. Soil organic carbon appeared to reach new equilibrium values following different time courses under different LUCs. We tried to estimate those time courses from inspection of reported changes in SOC stocks reported for different time periods after LUC and fitting appropriate relationships to each transition. For the conversion of cropland to secondary forest, we used a linear relationship to describe the time course of change as:

$$\Delta S_{ij(t)} = s_{ij}\, t \qquad \text{(Eq. 4)}$$

where $\Delta S_{ij(t)}$ is the change in SOC (%), at time $t$ (years), and $s_{ij}$ are fitted parameters.

For other LUC types, we used first-order exponential relationships:

$$\Delta S_{ij(t)} = \Delta_{ij,max}\,(1-e^{-kij\,t}) \qquad \text{(Eq. 5)}$$

where $\Delta_{ij,\,max}$ and $k_{ij}$ are fitted parameters for each LUC.

The change in SOC (%) after 100 years ($\Delta S_{ij(100)}$) were determined with Eqs. 4 and 5. Greenhouse gas fluxes in units of $CO_2$ related to these SOC changes were then calculated simply as:

$$F_{s,ij} = (44/12) \times \Delta S_{ij(100)} / 100 \qquad \text{(Eq. 6)}$$

where $F_{s,ij}$ is the annual $CO_2$ flux associated with a change in SOC from change of land use from $i$ to $j$, and $\Delta S_{ij(100)}$ is the change in SOC after 100 years associated with that land-use change. For the conversions of natural forest to grassland, and natural forest to secondary forest, the data provided no meaningful estimates of time courses of change, and we simply used the mean change of all observations.

### 2. 2. 3. Quantifying changes in $CH_4$ fluxes from enteric fermentation

Land-use changes to or from grazed grasslands alter $CH_4$ emissions from enteric fermentation by animals, especially if pastures are grazed by ruminants. Consequently, it is crucial to include changes in $CH_4$ emissions from enteric fermentation in assessing the overall effect of LUC on net GHG exchanges. Therefore, area-based annual $CH_4$ emission rates from enteric fermentation occurring on grazed pastures, $E_p$, were estimated as:

$$E_p = E_{t,p} / A_{t,p} \quad \text{(Eq. 7)}$$

where $E_{t,p}$ is an estimate of total global $CH_4$ emissions from enteric fermentation from pasture-fed livestock, and $A_{t,p}$ the estimated total area of grazed pastures.

Global $CH_4$ emissions from pasture-fed ruminant livestock in 2003 were estimated to be 44 Mt $CH_4$ $y^{-1}$ (35 Mt from cattle and 9 Mt from other domesticated ruminants including sheep, goats, buffalo and camelids; Clark et al., 2005; Kelliher and Clark, 2010), and the total area of grazed grasslands (including permanent meadows, pastures and extensive rangelands) in 2003 was estimated as $3.39 \times 10^9$ ha (FAOSTAT, 2013), giving $E_p$ = 13.0 kg $CH_4$ $ha^{-1}$ $y^{-1}$. Different animals convert different fractions of feed intake into $CH_4$ (camels: 7%; cattle and sheep: 6%; goats: 5%; horses 2.5%; IPCC, 1997) so that the $CH_4$ load of land converted to grazing is also affected by the type of animal grazing on it. However, because globally, grazing is dominated by sheep and cattle (FAO, 1996), we used the same average $CH_4$ emission rates for all grazed lands.

Any LUC that involved a change from or to grazed grassland was estimated to lead to an increase or decrease of the enteric fermentation flux $\Delta_{e,ij}$ by plus or minus 13.0 kg $CH_4$ $ha^{-1}$ $y^{-1}$ which was converted to GHG fluxes in units of $CO_2$ equivalents as:

$$F_{e,ij} = 25\ \Delta_{e,ij} \quad \text{(Eq. 8)}$$

where $F_{e,ij}$ is the GHG flux related to the change in enteric fermentation related to a specific land-use change, $\Delta_{e,ij}$ is the change in $CH_4$ flux rate from enteric fermentation, and 25 is the greenhouse warming potential of $CH_4$ (Forster et al., 2007).

### 2. 2. 4. Quantifying changes in soil $CH_4$ and $N_2O$ emissions

Data were acquired by searching the existing peer-reviewed literature published between 1970 and 2013 using the Web of Science and Google Scholar with search terms such as "land-use change", "land-use conversion", a description of different land use types (e.g., natural forest, cropland, grassland, or secondary forest), and the name of different GHG emissions ($CH_4$ or $N_2O$). We compiled $CH_4$ (n = 34) and $N_2O$ (n = 37) emissions data obtained from paired study sites with different land-use types (Supplementary Information Tables A to E). It should be noted that our data compilation includes a wide variety of studies that were conducted under diverse biophysical conditions using a range of methodologies for quantifying GHG emissions (e.g., sampling protocols, chamber design, and emission rate calculations), soil properties, and climatic factors, but we only considered field-based observations from paired studies that lasted for at least one full year.

We calculated the change in soil $CH_4$ and $N_2O$ emissions using the emissions values observed in paired-site studies:

$$\Delta_{m,ij} = E_{m,i} - E_{m,j} \quad \text{(Eq. 9a)}$$

$$\Delta_{n,ij} = E_{n,i} - E_{n,j} \quad \text{(Eq. 9b)}$$

where $\Delta_{m,ij}$ and $\Delta_{n,ij}$ are the differences in net soil $CH_4$ and $N_2O$ emissions between two land uses, respectively, and $E_{m,i,}$ and $E_{m,j,}$ are the net $CH_4$, and $E_{n,i}$ and $E_{n,j}$ are the net $N_2O$ emission rates associated with land uses *i* and *j*, respectively.

Greenhouse gas fluxes related to the change in $CH_4$ ($F_{m,ij}$) and $N_2O$ ($F_{n,ij}$) as a result of specific LUCs were then expressed in units of $CO_2$ equivalents as:

$$F_{m,ij} = 25 \times \Delta_{m,ij} \quad \text{(Eq. 10a)}$$

$$F_{n,ij} = 298 \times (44/28)\, \Delta_{n,ij} \quad \text{(Eq. 10b)}$$

where 25 and 298 are the 100-year greenhouse warming potentials of $CH_4$ and $N_2O$, respectively (Forster et al., 2007). The constant 44/28 converts activity data of $N_2O$ that are usually given in $N_2O{-}N$ units to $N_2O$ units.

**2. 2. 5. Quantifying the combined net greenhouse gas contributions of all greenhouse gases**

After converting the changes in net emissions of all GHGs to the same units and analysing them over the same time interval (100 years), we then calculated the total net GHG contribution from all gases and the different contributing factors as:

$$F_{t,ij} = F_{b,ij} + F_{s,ij} + F_{e,ij} + F_{m,ij} + F_{n,ij} \quad \text{(Eq. 11)}$$

where $F_{t,ij}$ (in t $CO_2$ eq $ha^{-1}$ $y^{-1}$) is the total net GHG contribution of LUC from land use *i* to land use *j*, with the other terms having been defined above.

### 2. 3. Global estimate of total historical net greenhouse gases contribution by land-use change

We estimated the total net GHG contribution of LUC from forest to cropland or grassland from the areas estimated to have undergone different LUCs between 1765 and 2005 (Meiyappan and Jain, 2012) and applying the terms calculated in the present study (Eq. 12). Hence, the historical net GHG contributions from forest to agricultural uses, $F_{h,fa}$, were calculate as:

$$F_{h,fa} = A_{h,\,fc} \times F_{t,fc} + A_{h,\,fp} \times F_{t,fp} \quad \text{(Eq. 12)}$$

where $A_{h,fc}$ and $A_{h,fp}$ are the areas converted between 1765 and 2005 from forest to cropping or pasture, respectively, and $F_{t,fc}$ and $F_{t,fp}$ are the corresponding total net GHG emission rates associated with those respective LUCs as defined above.

### 2. 4. Statistical analysis

The uncertainty of our estimates of GHG emissions changes was assessed by calculating the means and 95% confidence intervals calculated from the values reported in individual studies by treating the numbers reported in different studies as independent observations. All tests were conducted with SAS® ver. 9.2 (SAS Institute, Cary, NC, USA) and SigmaPlot® ver. 11.0 (Systat Software Inc., San Jose, CA, USA).

The goodness of relationships fitted to our compiled observations of changes in soil organic carbon were assessed by calculating model efficiencies, EF, determined as (Nash and Sutcliffe, 1970):

$$EF = 1 - \frac{\sum(y_o - y_m)^2}{\sum(y_o - \bar{y})^2} \quad \text{(Eq. 13)}$$

where $y_o$ are the individual observations, $y_m$ the corresponding model values, and $\bar{y}$ the mean of all observations.

For estimating 95% confidence intervals of our estimates of soil C changes after 100 years, we used the delta method (Weisberg, 2005). It is available as a package (al3) in the R statistical computing environment (version 2.15.2) and was applied to our data of changes in soil organic carbon as a function of time for different LUCs. The delta method allows for the calculation of functions of random variables using Taylor expansions (Seber, 1982; Lyons, 1991; Bolker, 2008).

Biomass C estimates were obtained from IPCC (2001) and FAO (2010), where the underlying data were reported without error estimates which did not allow us to calculate further uncertainty estimates. Similarly, $CH_4$ emissions from enteric fermentation were calculated from data in Kelliher and Clark (2010) and FAOSTAT (2013), which provided no error information of their respective estimates.

## 3. Results and Discussion

### 3. 1. Change in biomass carbon stocks following land-use change

Depending on the type of LUC, perennial vegetation may be removed (i.e. deforestation) and replaced either by different perennial types of vegetation (i.e. tree plantation) or crops or pastures with much lower C stocks. We estimated changes in biomass C stocks following LUC to range from –154.5 t C $ha^{-1}$ to +76 t C $ha^{-1}$ (Table 2). As forests contained much greater biomass than agricultural land, any conversion from forest to other land uses led to large C losses, while the conversion from agricultural land to secondary forest led to large C gains. The average C stocks of

grasslands were also greater than the C stocks of cropland (Table 1), thus also making a small difference in any conversions between grasslands and croplands.

In the present study, we used an estimated global mean natural forest biomass of 156.8 t C $ha^{-1}$, based on the data compilation of FAO (2010) and WBGU (1998), but there is much variation within that global average (Goodale et al., 2002; Houghton, 2005; Aalde et al., 2006; Kindermann et al., 2008), with the largest amount of biomass recorded for tropical forests in South America and *Pinus radiata* plantations in New Zealand, while lowest amounts of biomass have been reported for marginal forests in Russia, China, Canada and Australia (FAO, 2010; Kirschbaum et al., 2012). The numbers estimated here, therefore, constitute estimates of the C-stock changes for LUC involving land parcels with C stocks of the average of their respective categories. It also includes the assumption that LUC would, on average, involve land parcels with these average C stocks. We have no additional information to test that assumption, and whether actual LUC may preferentially involve land units with greater or lesser than average C stocks. For instance, it is possible that more fertile areas are more likely to be chosen for LUC, and that those more fertile areas also have C stocks higher than the average for respective vegetation classes. If that were the case, the C loss associated with such LUCs would be underestimated. This can only be flagged as a possibility here, but it would require more detailed regional analyses to verify whether it would actually constitute a bias, or be able to quantify its extent.

In addition, the loss of C in tree roots is often ignored in carbon counting involving deforestation. Trees usually have root-shoot ratios of 0.2 - 0.5, with higher values under drier or less fertile conditions (Mokany et al., 2006). This contribution is usually included in the assessment of live biomass carbon, but after deforestation, there is no consistent treatment of dead roots. Dead roots usually remain in the soil where they slowly decompose (Ludovici et al., 2002; Boutton et al., 2009). Fine roots may decompose within a year or two (Silver and Miya, 2001), while the decay of coarse roots can range from a few years (Garrett et al., 2012) to decades (Chen et al., 2001; Olajuyigbe et al., 2011). That carbon is either respired as $CO_2$ or incorporated into SOC with the immobilisation

of nitrogen (Kirschbaum et al., 2008). Despite these important roles, and their quantitative significance (Kirschbaum et al., 2008; Dean et al., 2012; Wang et al., 2012), the time course of the decay of roots has not been well quantified for most ecosystems.

Biomass C losses through deforestation can also occur very rapidly, especially if it involves slash burning, while gains in biomass C following re/afforestation tend to be much slower and can take decades to centuries, depending on the climate, nutrient availability and growth properties of specific forests. Losses of biomass C through deforestation therefore cannot simply be reversed because full reversal of the loss of biomass C stocks requires decades to centuries. We used a 100-year time frame in the present study to quantify C-stock changes, but the numeric outcome would have been different if a different time horizon had been used, with shorter time horizons increasing the calculated importance of C-stock changes, and time horizons of more than 100 years reducing it.

### 3. 2. Changes in soil organic carbon stocks following land-use change

Average SOC in soils prior to LUC ranged from 31.3 to 93.9 t C $ha^{-1}$ (Table 4), and SOC changed by between –50.6% and +88.5% over 100 years under different LUCs (Tables 3, 4 and Fig. 1). Combining average SOC stocks in soils before LUC with rates of SOC change following LUC resulted in changes in SOC stocks following LUC to range from –44.5 to +59.0 t C $ha^{-1}$ over 100 years (Table 4). Largest losses were seen in the conversions from natural or secondary forest to cropland (–33.1±11.2 and –44.5±12.3 t C $ha^{-1}$, respectively), while largest gains were possible when cropland was converted to secondary forest (59.0±19.2 t C $ha^{-1}$).

Conversion from primary forest and secondary forest to cropland resulted in SOC loss of 35.3±4.9% and 50.6±3.4%, respectively, and most SOC losses occurred over the initial 10 years after conversion (Fig. 1a and 1h). That pattern is usually considered to be linked to intensive agricultural land management, including soil disturbance so that croplands lose SOC until a new balance between carbon inputs and outputs is re-established. In contrast, when natural forest was converted to grassland, there was no clear pattern of SOC change, but individual sites showed a

wide range of possible changes, ranging from –60% to +80% over 100 years (Fig. 1b). Similar results were reported in previous studies that summarized SOC changes after deforestation to pasture: Murty et al. (2002) found no consistent changes in soil carbon stocks, while Guo and Gifford (2002) reported a small and statistically significant increase in SOC of about 8%. The wide range of changes reported in individual studies suggest that converted grasslands could be either carbon sinks or sources depending on specific local management and environmental conditions (Murty et al., 2002). Previous studies suggested overgrazing may cause soil compaction which may reduce plant productivity and carbon inputs to the soil, which in turn may result in a loss of SOC (e.g., Fearnside and Barbosa, 1998).

Conversion from cropland to secondary forest led to SOC gains of 88.5±21.6% over 100 years (Fig. 1e), reversing the C loss seen when forests were deforested, although the change was much slower than the C loss upon deforestation. It indicates that the factors that cause the decrease in SOC under cropping, probably related to frequent soil disturbance and reduced carbon inputs, can be reversed when that disturbance ceases. The increase in SOC may also be affected by soil type. A meta-analysis of soil C changes after reforesting cropped soils found that soil carbon did not change in low-clay soils, but increased by an average of 26% for sites with higher clay contents (Laganiére et al., 2010).

In the conversion from grassland to secondary forest, there also was an increase of SOC by about 28% after 100 years and 39% after 200 years (Fig. 1g). Previous meta-analyses reported divergent results on SOC change after reforesting pastures. Some studies reported losses of SOC by about 10% (Guo and Gifford, 2002; Paul et al., 2002; Davis and Condron, 2002; Tate et al., 2005), while others reported gains of SOC by up to 28% (Laganiére et al., 2010; Don et al., 2011; Poeplau et al., 2011; Power et al., 2011). The present study combined all the data summarized in these previous studies, and these combined data confirmed the result of the more recent analyses in showing that carbon gains following reforestation of pastures are more common than carbon losses.

Switching between different agricultural land-use types, such as between cropland and grassland, also showed clear patterns in SOC changes (Figs. 1d, 1f). Converting cropland to grassland increased SOC by nearly 50% (Fig. 1d), whereas converting grassland to cropland decreased SOC by about 45% (Fig. 1f) and was largely completed within the first 10 years after conversion. This difference is usually attributed to loss of SOC in cropland due to cultivation and soil disturbance (e.g., Mann, 1986; Lal, 2004).

Land-use change from natural forest to agricultural land can also cause soil erosion by wind and water (e.g., Pimentel et al. 1995; Lal, 2003). Erosion can cause large site C losses (e.g., Lal, 2003; Post et al., 2004; Gaiser et al., 2008). Conversely, converting agricultural lands to secondary forest can reduce soil erosion and prevent further site C losses. However, while erosion clearly depletes local carbon stocks, its effect on the global carbon budget is less clear as carbon lost from a site may not necessarily be lost to the atmosphere but may be buried and stabilised in deep ocean sediments, instead. If most C is lost before deposition or stabilisation, then erosion is likely to constitute a net source of $CO_2$ to the atmosphere (e.g., Lal, 2003; Polyakov and Lal, 2008). However, if most C can be deposited in ocean sediments or stabilized without being oxidised (e.g., Govers et al., 1994; Dunne et al., 1998; Brackley et al., 2010), and if eroded sites can regain their lost soil C stocks, then erosion could even constitute as a net sink of $CO_2$ (e.g., Dymond, 2010; Quinton et al., 2010; Dotterl et al., 2012; Van Oost et al., 2012).

It therefore remains problematic to assess the overall effect of LUC induced soil erosion on global warming (Tate et al., 2005; Kuhn et al., 2009; Kirschbaum et al., 2012; Sanderman and Chappell, 2013). However, while the role of erosion in the global C cycle remains uncertain, it should not detract from the fact that erosion clearly is a massive problem for local food production (e.g., Godfray et al., 2010; Lal, 2010) and siltation of downstream water reservoirs (e.g., Pimentel et al., 1995; Thothong et al., 2011).

### 3. 3. Changes in $CH_4$ and $N_2O$ emissions

Table 5 summarized the effects of LUC from natural forests to croplands, grasslands and secondary forest on rates of enteric fermentation and net soil emissions rates of $CH_4$ and $N_2O$, with more detailed information provided in the Supplementary Information (Tables A–C). Our compiled data showed that conversion of forest to cropland or grassland tended to increase $N_2O$ emissions, which was reversible when cropland or grassland was converted to secondary forest.

$N_2O$ emissions are mainly associated with the turnover of N in the soil (Bouwman, 1996; Kim et al., 2012). These natural processes have been intensified through human interventions, mainly through agricultural activities, and principally through the increased use of N fertilisers (e.g., Del Grosso et al., 2009; Kirschbaum et al., 2012; Kim et al., 2012). Changes in $N_2O$ emissions following LUC can thus be principally related to changes in the amount of N inputs. Cropland and grassland usually receive larger N inputs than forests through applied organic and inorganic N fertilizers and animal excreta. Consequently, nitrification and denitrification processes are intensified, and more $N_2O$ can be produced during N-transformation processes in the soil (e.g., Robertson and Tiedje, 1987; Bouwman, 1996; Kim et al., 2012). In addition, any increase in soil acidity due to excessive synthetic fertilizer use can increase $N_2O$ emissions by decreasing $N_2O$ reductase activity (Barak et al., 1997; Bulluck et al., 2002). Increased soil compaction by intensive soil management can further increase $N_2O$ emissions by increasing the rate of denitrification (e.g., Luo et al., 1999; Bilotta et al., 2007). In contrast, conversion of cropland and grassland to forest is usually associated with reduced N inputs to soils, leading to less $N_2O$ being produced in soils.

$CH_4$ emissions from enteric fermentation associated with any conversions to or from grasslands can be 3 –20 times larger than changes in net soil $CH_4$ emissions resulting from LUC (Table 5). Change in $CH_4$ emissions from enteric fermentation is thus a critical component of altered GHG balances following LUC. In this study, a global average value of 13.0 kg $CH_4$ $ha^{-1}$ $y^{-1}$ was applied to $CH_4$ emission from grazed pastures, but this value is more than an order of magnitude lower than $CH_4$ emissions of 150 (for sheep) and 240 (for cattle) kg $CH_4$ $ha^{-1}$ $y^{-1}$ used as typical values for intensively managed grasslands (Kirschbaum et al., 2013). This highlights the

limitation of using average enteric fermentation values from a global assessment that would have included areas with very low $CH_4$ emissions from enteric fermentation. The contribution of $CH_4$ emissions from enteric fermentation can thus change greatly with the global region where LUC may occur and with the productive potential of those regions.

A final, but generally small, factor is the change in net soil $CH_4$ emissions. Our data showed that the conversion of forest to cropland or grassland tended to increase net $CH_4$ emissions, and conversion of cropland or grassland to secondary forest tended to decrease it. While most well-drained soils can act as either a sink or source of $CH_4$ (e.g., Price et al., 2010), $CH_4$ oxidation generally tends to dominate, and changes in net fluxes tend to be mainly related to changes in a soil's $CH_4$ oxidation potential. Forests create favourable soil conditions for $CH_4$ oxidation that can remove ≈ 1–5 kg $CH_4$ $ha^{-1}$ $y^{-1}$ from the atmosphere (Smith et al., 2000). However, it may take over 100 years to recover maximal $CH_4$ oxidation rates after disturbance by deforestation (Smith et al., 2000; Allen et al., 2009; Singh and Singh, 2012). Changed $CH_4$ fluxes after LUC have been shown to be related to changes in the composition (Singh et al., 2007, 2009) and abundance (Menyailo et al., 2008) of the methanotroph communities, and various studies (e.g., Singh et al., 2007; Dörr et al., 2010; Nazaries et al., 2011) found that increased $CH_4$ oxidation following afforestation was directly linked to a shift towards type-II methanotrophs.

### 3.4. Combined effect on net greenhouse gas emissions

We then combined our best estimates of changes in biomass C, SOC, $CH_4$ emissions from enteric fermentation and soil processes, and soil $N_2O$ emissions following LUC into a combined assessment. It showed that deforestation of primary forests to any other land use increased net GHG emissions (Fig. 2; Table 6). Conversion from natural forest to cropland led to the largest increase in net GHG emissions, followed by conversion from secondary forest to cropland.This was primarily due to the loss of biomass C, but $N_2O$ emissions also tended to increase, and net $CH_4$ emissions increased, especially for any conversions to grazed grasslands (Fig. 2). Increased GHG emissions

were largely, but not completely, reversible over 100 years if agricultural land was further converted to secondary forest. Conversely, conversion from cropland to secondary forest led to the largest reduction of net GHG emissions, followed by conversion from grassland to secondary forest. Conversion from croplands to grasslands also decreased net GHG emissions because of decreased $N_2O$ emissions and slightly increased biomass and soil C stocks (Fig. 2; Table 6).

Since woody biomass consists of 46–51% carbon (Aalde et al., 2006), any loss or gain of woody biomass through LUC corresponds to an equivalent $CO_2$ flux to or from the atmosphere. Considering the large change of woody biomass in most LUCs, it is not surprising that changes in biomass carbon dominated overall net GHG changes (Table 6; Fig. 2). For all LUCs involving forests, the change of biomass C was the major contributor to net GHG emissions (74.1 –90.0% of the net change; Table 6). In the conversion of cropland to grassland, the change of $N_2O$ emissions (77% of the net change; Table 6) was the main contributor to net GHG emissions.

Although net changes in $CH_4$ and $N_2O$ emissions are numerically relatively small compared to the contribution of change in biomass carbon or SOC, they are nonetheless important for global warming because of the high global warming potential of these gases (Forster et al., 2007). This is particularly important for $N_2O$ since it also has a long atmospheric longevity (Forster et al., 2007) so that emissions will still contribute to global warming even centuries after their emission. Emissions therefore cannot be readily reversed even if that were warranted through rising global environmental concerns.

Globally, historical LUC from natural forests to crop and grasslands has contributed a cumulative 1647±521 Gt $CO_2$ eq between 1765 and 2005, equivalent to average emissions of 6.9±2.2 Gt $CO_2$ eq per year (Fig. 3). Conversion to cropping was responsible for about ¾ (74.7%) of that GHG contribution. Regionally, North America (18%), Latin America (27%) and South and South-East Asia (21%) together were responsible for about 2/3 of those net emissions, with conversion to cropping dominating in all regions except Latin America.

There appear to have been few previous studies that attempted to comprehensively assess the total GHG implications of LUC, with the study most closely comparable to our work being that by Kirschbaum et al. (2013). Their work differed from the present study by modelling the GHG effects of LUC in greater detail, including the atmospheric longevity of different GHGs and parameterising their model from agricultural lands with high $CH_4$ and $N_2O$ activities. Consequently, biomass and SOC changes were relatively less important in their study than we found here based on a global data compilation. Kirschbaum et al. (2013) also found that conversion from forest to grazed pastures generally had greater GHG impacts that conversion to cropland. That, too, was related to studying systems with higher grazing activity than the global average. In these high-activity systems, $N_2O$ emissions and $CH_4$ emissions from enteric fermentation had a similar effect on overall GHG balances as C-stock changes. As these emissions are higher for grasslands than croplands (Fig. 2), it increased the GHG impacts of conversion of forests to grasslands even though for cropland, soil C losses added to the GHG impacts of conversion to those systems (Fig. 2).

### 3.5. Draining wetlands

In addition to the issues quantified in the present work, there are a number of land-use transitions that are also of great importance but that could not be covered in the present work. Probably foremost amongst those is the draining of wetlands. It has been estimated that globally, about 50% of wetlands have been converted to agricultural and other land uses (Zedler and Kercher, 2005; Verhoeven and Setter, 2010), with a potentially significant effect on overall global GHG fluxes. Natural wetlands are also typically small C sinks (Hooijer et al., 2010; Mitsch et al., 2012). Any LUC that involves the draining of wetlands is likely to lead to a large reduction in $CH_4$ emissions (Supplementary Information Table E). However, in addition to the usual short-term loss of any woody biomass, draining wetland typically also leads to the release of large amounts of stored SOC into the atmosphere (Hooijer et al., 2010; Crooks et al., 2011) which can lead to an eventual loss of their total soil C stocks (e.g., Glenn et al., 1993; Langeveld et al., 1997; Santín et al., 2009; Crooks

et al., 2011). Carbon losses can be particularly large and rapid if drained wetlands are also subject to fire (van der Werf et al., 2006, 2010). In the overall GHG balance of drained wetlands, C losses from drained wetlands can therefore dominate the overall GHG balance even after accounting for the respective greenhouse warming potentials of $CO_2$ and $CH_4$ (e.g., Janssens et al., 2005).

### 3.6. Implication and suggested future studies

In the present study, biomass C stocks were estimated from reported biomass stocks in different parts of the world, and changes in $CH_4$ and $N_2O$ emissions were estimated from a limited number of available studies. This approach can only provide a first estimate, and the results need to be interpreted cautiously. Further studies could refine activity estimates especially for systems with few current observations. However, the GHG impact of LUC also differs greatly from site to site based on differences in the key determinants at respective sites. This is obviously the case with respect to differences in biomass for any deforestation activity as "forest" biomass can vary greatly based on regional differences (e.g., IPCC, 2006; FAO, 2010) or with the specific local properties of cleared forests.

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### Supplementary materials

Table A to E. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post − pre) in different land use types.

Table F. Greenhouse gas emissions in land use changes from forest to crop and grasslands in 1765 to 2005

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Table 1. Biomass carbon (C) stocks in natural forest, secondary forest, cropland, and grassland. Estimates of C stocks for natural and secondary forest were determined using FAO (2010) and WBGU (1998). Biomass C stocks of cropland and grassland were obtained from IPCC (2001).

| Type | C stocks per unit area ($B_{p,i}$) |
|---|---|
| Natural forest | 156.8 t C $ha^{-1}$ |
| Secondary forest | 78.4 t C $ha^{-1}$ |
| Cropland | 2.5 t C $ha^{-1}$ |
| Grassland | 10.0 t C $ha^{-1}$ |

Table 2. Change in biomass carbon stocks (ΔBiomass C) for various land use changes and their contribution as carbon dioxide ($CO_2$) to the atmosphere (by dividing the carbon-stock change by 100 years). Note that a negative carbon-stock change (i.e. a loss of carbon from a site) leads to a positive change in atmospheric $CO_2$ and makes a warming contribution. Biomass C estimates were taken from Table 1.

| Land use type | | Biomass C (t C $ha^{-1}$) | | ΔBiomass C (t C $ha^{-1}$) | Contribution to the atmosphere (t $CO_2$ $ha^{-1}$ $yr^{-1}$) |
|---|---|---|---|---|---|
| Pre | Post | Pre | Post | | |
| Natural forest | Cropland | 156.8 | 2.5 | -154.3 | 5.7 |
| Natural forest | Grassland | 156.8 | 10 | -146.8 | 5.4 |
| Natural forest | Secondary forest | 156.8 | 78.4 | -78.4 | 2.9 |
| Cropland | Grassland | 2.5 | 10 | 7.5 | -0.3 |
| Cropland | Secondary forest | 2.5 | 78.4 | 75.9 | -2.8 |
| Grassland | Cropland | 10 | 2.5 | -7.5 | 0.3 |
| Grassland | Secondary forest | 10 | 78.4 | 68.4 | -2.5 |
| Secondary forest | Cropland | 78.4 | 2.5 | -75.9 | 2.8 |

Table 3. Parameters of fitted lines for changes in soil organic carbon in land-use changes. Means ± 95% confidence intervals.

| Land use change type | $\Delta_{ij,\,max}$ | $k_{ij}$ | $s_{ij}$ | EF[a] |
|---|---|---|---|---|
| Natural forest to cropland | -35.3±4.9 | 0.3±0.2 | - | 0.10 |
| Natural forest to grassland | - | - | - | |
| Natural forest to secondary forest | - | - | - | |
| Cropland to grassland | 48.7±20.0 | 0.1±0.1 | - | 0.37 |
| Cropland to secondary forest | - | - | 0.89±0.14 | 0.68 |
| Grassland to cropland | -46.2±9.0 | 0.4±0.4 | - | 0.22 |
| Grassland to secondary forest | 36.4±31.0 | 0.01±0.02 | | 0.04 |
| Secondary forest to cropland | -50.6±3.4 | 0.8±0.2 | - | 0.95 |

Most data sets could be well described with single exponential or linear relationships (eqs. 5 and 6). Even for LUCs where the data showed a linear dependence of soil-carbon changes, there are obvious ecophysiological limits to the temporal extent of such linear dependencies. The relationships should therefore not be extrapolated beyond the range of the data.

[a] EF refers to model efficiency (eq. 14).

[b] For conversions for which no parameters are listed, fitted lines provided no significant relationships.

Table 4. Change of soil organic carbon stocks (ΔS) following various land use changes and their contribution to atmospheric carbon dioxide ($CO_2$) averaged over 100 years. The SOC in pre-land use change (LUC) were obtained from the combined global meta data of Murty et al. (2002), Don et al. (2011) and Poeplau et al. (2011). Percentage changes were calculated from the 100-year values of the fitted curves in Figure 1, with the parameters given in Table 3. ΔS after 100 years was calculated by multiplying SOC in pre-LUC by the percentage change after 100 years in the preceding column. For conversions from natural forests to grasslands or seconday forests, no time courses of change could be established (see Fig. 1), and change after 100 years was simply takne as the mean change of all observations. Note that a site-level loss of SOC corresponds to an increase in atmospheric $CO_2$. Means ± 95% confidence intervals.

| Land use type | | SOC in pre-LUC | Change after 100 years | ΔS after 100 years | Contribution to the atmosphere |
|---|---|---|---|---|---|
| Pre | Post | t C $ha^{-1}$ | % | t C $ha^{-1}$, 100 years | t $CO_2$ $ha^{-1}$ $y^{-1}$ |
| Natural forest | Cropland | 93.9±28.8 | -35.3±4.9 | -33.1±11.2 | 1.2±0.4 |
| Natural forest | Grassland | 47.1 ±21.8 | 6.3 ± 5.8 | 3.0 ± 3.1 | -0.1±0.1 |
| Natural forest | Secondary forest | 53.7 | -15.3 ± 12.5 | -8.2 ± 6.7 | 0.3±0.3 |
| Cropland | Grassland | 36.8 ±27.4 | 48.7±20.0 | 17.9±15.2 | -0.7±0.6 |
| Cropland | Secondary forest | 66.7 ±14.3 | 88.5±21.6 | 59.0±19.2 | -2.2±0.7 |
| Grassland | Cropland | 31.1 ±31.9 | -46.2±9.0 | -14.4±15.0 | 0.5±0.6 |
| Grassland | Secondary forest | 69.7 ±16.1 | 23.0±17.1 | 16.0±12.5 | -0.6±0.5 |
| Secondary forest | Cropland | 88.0 ±23.5 | -50.6±3.4 | -44.5±12.3 | 1.6±0.5 |

Table 5. Change of enteric-fermenattion and net soil methane ($CH_4$) emissions and soil nitrous oxide ($N_2O$) emissions following various land use changes and their contribution to atmospheric greenhouse gases as carbon dioxide equivalent ($CO_2$ eq) over 100 years. Numbers in brackets indicate the number of observations. Data show means ± 95% confidence intervals.

| Land use type | | Δ $CH_4$ emissions from enteric fermentation | Δ Net soil $CH_4$ emissions | Total net Δ$CH_4$ emissions | Contribution of total net Δ$CH_4$ emissions[†] | Δ$N_2O$–N emissions | Contribution of Δ$N_2O$ emissions |
|---|---|---|---|---|---|---|---|
| Pre | Post | kg $CH_4$ $ha^{-1}$ $y^{-1}$ | kg $CH_4$ $ha^{-1}$ $y^{-1}$ | kg $CH_4$ $ha^{-1}$ $y^{-1}$ | t $CO_2$ eq $ha^{-1}$ $y^{-1}$ | kg $N_2O$–N $ha^{-1}$ $y^{-1}$ | t $CO_2$ eq $ha^{-1}$ $y^{-1}$ |
| Natural forest | Cropland | 0 | 3.1±3.6 (2) | 3.1±3.6 | 0.08±0.09 | 1.5±1.6 (5) | 0.7±0.8 |
| Natural forest | Grassland | 13.1 | 2.6±2.4 (3) | 15.7±2.4 | 0.38±0.06 | 1.1±1.3 (3) | 0.6±0.6 |
| Natural forest | Secondary forest | 0 | 1.04 (1) | 1.04 | 0.03 | -0.02 (1) | -0.01 |
| Cropland | Grassland | 13.1 | 0.6±1.0 (2) | 13.7±1.0 | 0.34±0.03 | -4.7±9.2 (2) | -2.2±4.3 |
| Cropland | Secondary forest | 0 | -2.3±5.4 (2) | -2.3±5.4 | -0.06±0.14 | -1.5±8.5 (2) | -0.7±3.9 |
| Grassland | Cropland | -13.1 | –[*] | -13.1 | – | – | – |
| Grassland | Secondary forest | -13.1 | -4.9±4.9 (11) | -18.0±4.9 | -0.45±0.12 | -0.05±0.2 (11) | -0.02±0.09 |
| Secondary forest | Cropland | 0 | -0.59 (1) | -0.59 | -0.02 | -1.4 (1) | -0.7 |

* Data not available

Table 6. Contribution of changes in biomass carbon (ΔBiomass C), soil organic carbon (ΔS), total net methane ($\Delta CH_4$) and nitrous oxide ($\Delta N_2O$) emissions following land use change to atmospheric greenhouse gases as carbon dioxide equivalent ($CO_2$ eq) following land use change. Values in parentheses give the contribution of each component to the total net greenhouse gas emission. Means ± 95% confidence intervals.

| Land use type | | Contribution of ΔBiomass C | Contribution of ΔS | Contribution of total net $\Delta CH_4$ emissions[†] | Contribution of $\Delta N_2O$ emissions | Total contribution |
|---|---|---|---|---|---|---|
| Pre | Post | t $CO_2$ $ha^{-1}$ $y^{-1}$ | t $CO_2$ $ha^{-1}$ $y^{-1}$ | t $CO_2$ eq $ha^{-1}$ $y^{-1}$ | t $CO_2$ eq $ha^{-1}$ $y^{-1}$ | t $CO_2$ eq $ha^{-1}$ $y^{-1}$ |
| Natural forest | Cropland | 5.7 (74.1%) | 1.2±0.4 (15.7%) | 0.08±0.09 (1.0%) | 0.7±0.8 (9.2%) | 7.6±1.3 |
| Natural forest | Grassland | 5.4 (85.9%) | -0.1±0.1 (-1.6%) | 0.38±0.06 (6.1%) | 0.6±0.6 (9.6%) | 6.2±0.8 |
| Natural forest | Secondary forest | 2.9 (90.0%) | 0.3±0.3 (9.4%) | 0.03 (0.9%) | -0.01 (-0.3%) | 3.2±0.3 |
| Cropland | Grassland | -0.3 (10.4%) | -0.7±0.6 (24.5%) | 0.34±0.03 (-11.9%) | -2.2±4.3 (77%) | -2.9±4.9 |
| Cropland | Secondary forest | -2.8 (48.5%) | -2.2±0.7 (38.3%) | -0.06±0.14 (1.0%) | -0.7±3.9 (12.2%) | -5.7±4.7 |
| Grassland | Secondary forest | -2.5 (69.9%) | -0.6±0.5 (16.9%) | -0.45±0.12 (12.6%) | -0.02±0.09 (0.6%) | -3.6±0.7 |
| Secondary forest | Cropland | 2.8 (76.0%) | 1.6±0.5 (43.6%) | -0.02 (-0.5%) | -0.7 (-19.1%) | 3.7±0.5 |

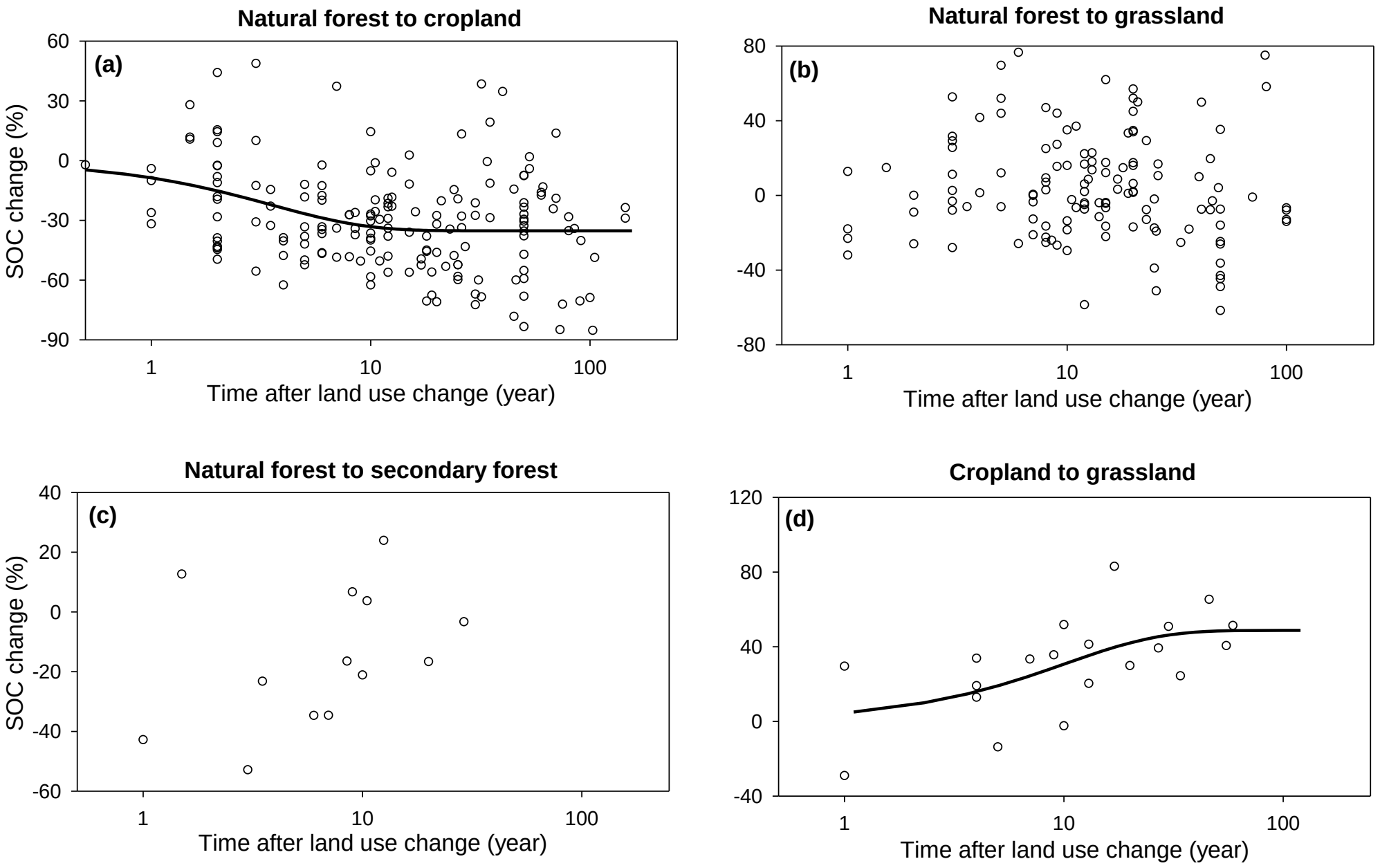
## Figure captions

Figure 1. Changes in soil organic carbon (SOC) plotted against time after land use change from various land use changes as specified in each panel. Lines in different panels are fitted curves, forced through 0 at time 0. We used linear relationships in conversion of cropland to secondary forest (e), and first-order exponential relationship for other panels. Note that the time axis uses a linear scale for (e) and (g) and exponential scale for the other panels. The parameters for each fitted curve are given in Table 3. No relationships were fitted when time courses of change could not be determined. The data shown here combine the information compiled for the earlier reviews of Murty et al. (2002), Don et al. (2011), Poeplau et al. (2011) and Power et al. (2011).

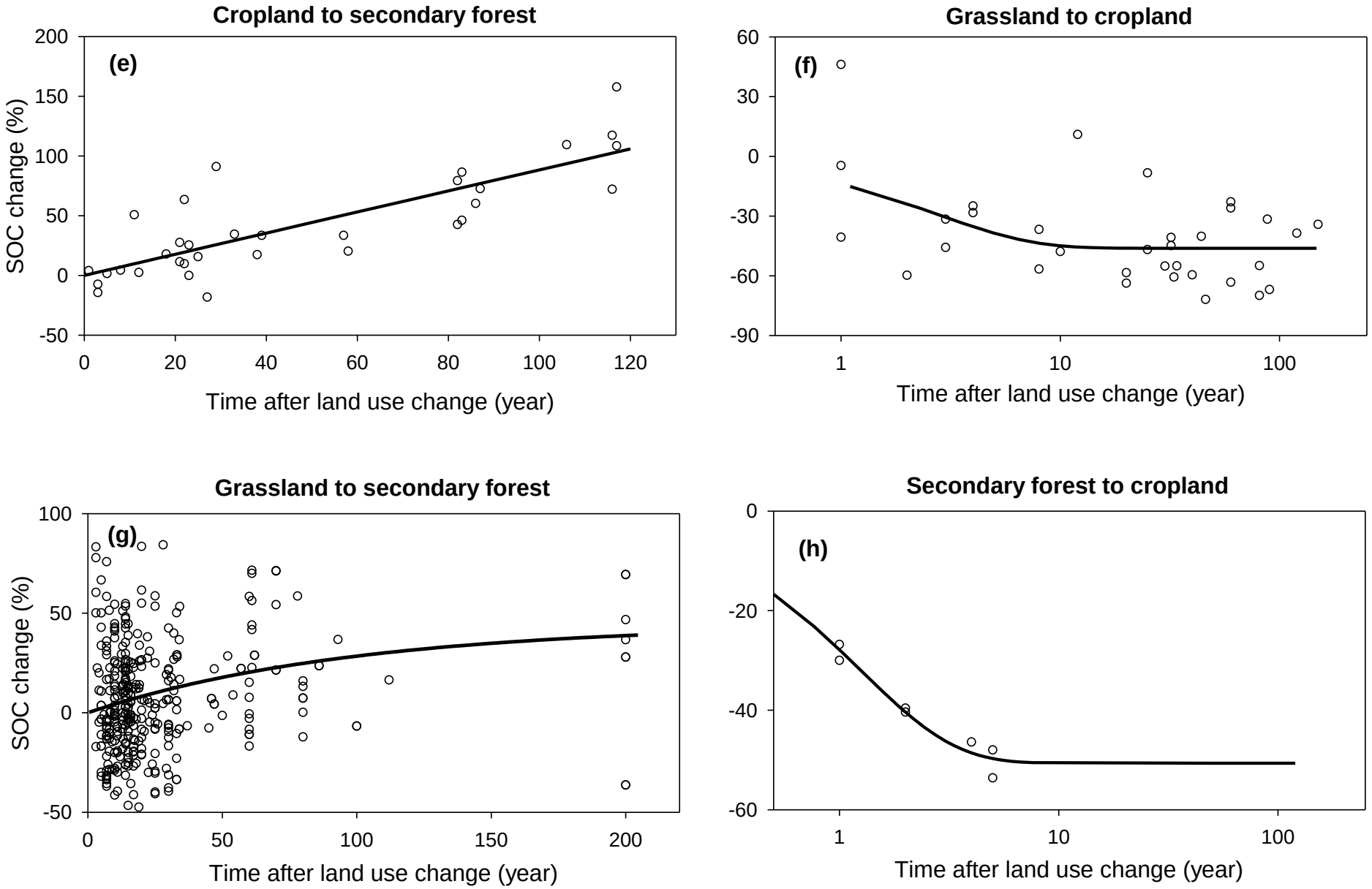
Figure 2. Contribution to global warning by various changes in biomass carbon (ΔBiomass C), soil organic carbon (ΔSOC), total net methane ($\Delta CH_4$), soil nitrous oxide ($\Delta N_2O$) emissions, and the combined effect of all gases following land use changes. Positive numbers indicate a warming contribution.

Figure 3. Greenhouse gas emissions resulting from land use changes from forest to crop or grasslands between 1765 and 2005. Error bars indicate 95% confidence intervals.

# Figure 1


Natural forest to cropland
(a)
SOC change (%)
60
30
0
-30
-60
-90
1
10
100
Time after land use change (year)
Natural forest to grassland
(b)
80
40
0
-40
-80
1
10
100
Time after land use change (year)
Natural forest to secondary forest
(c)
SOC change (%)
40
20
0
-20
-40
-60
1
10
100
Time after land use change (year)
Cropland to grassland
(d)
120
80
40
0
-40
1
10
100
Time after land use change (year)

# Figure 1 cont'd



# Figure 2

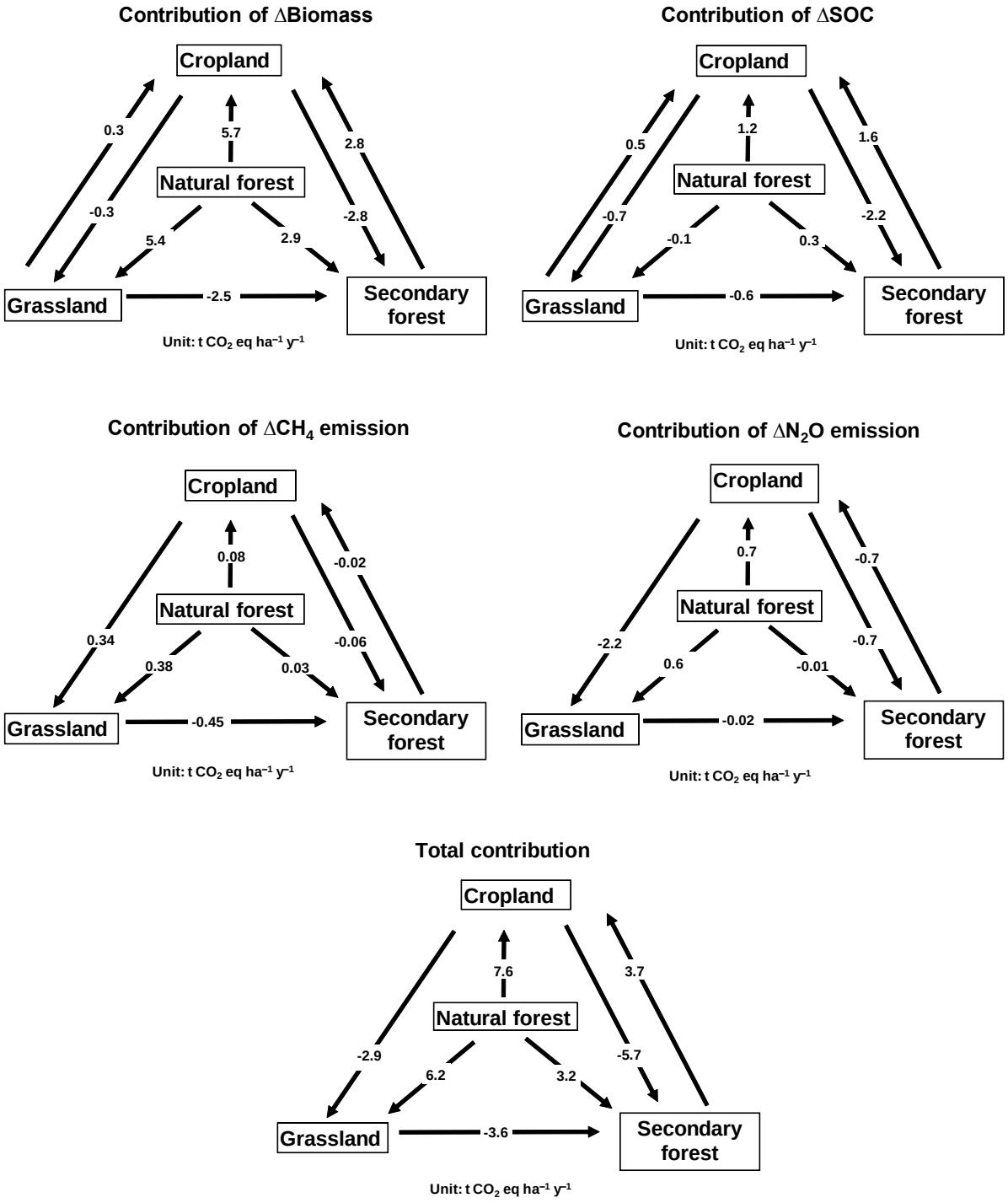
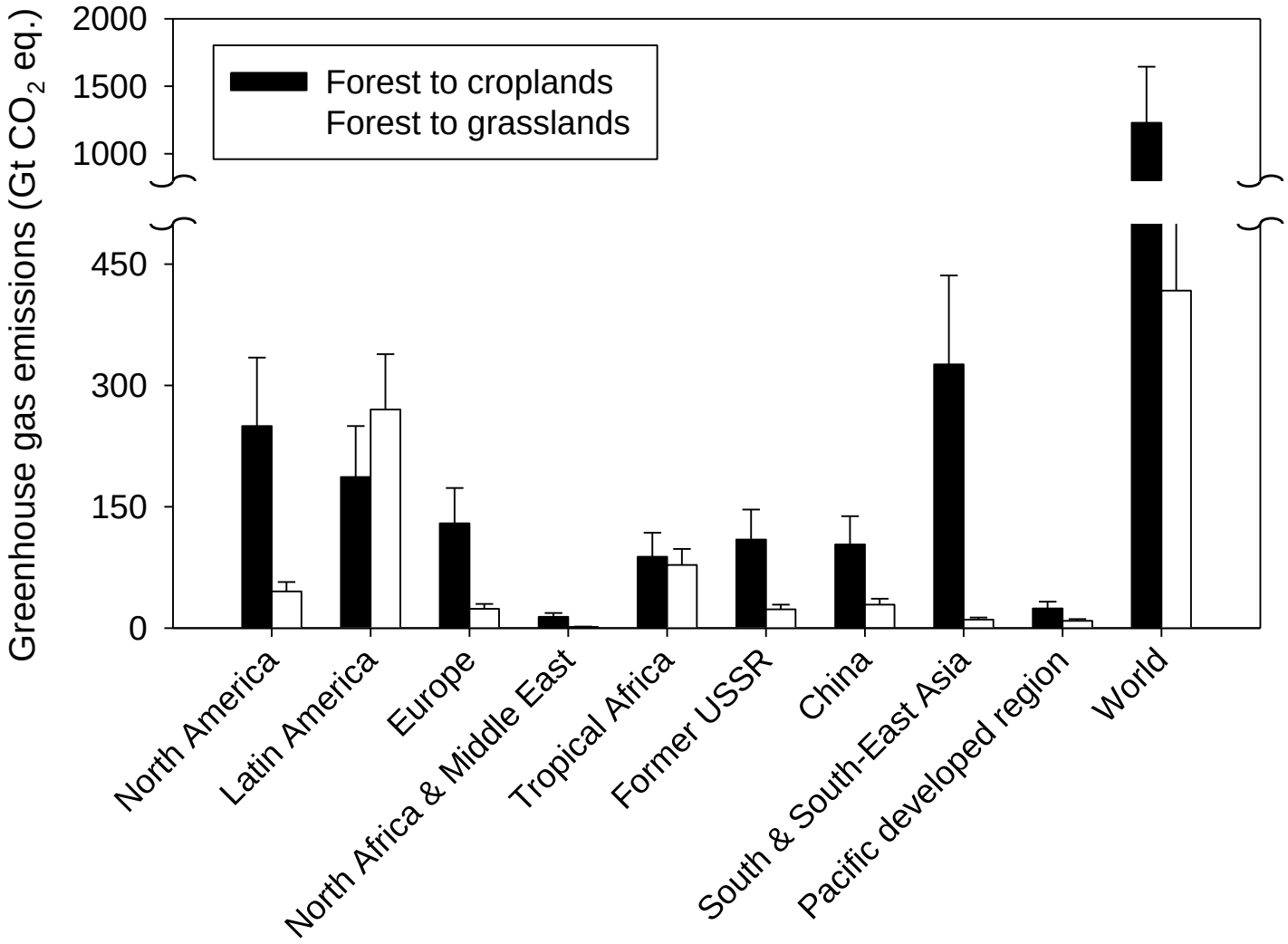

Contribution of ΔBiomass
Cropland
Natural forest
Grassland
Secondary forest
0.3
-0.3
5.7
2.8
-2.8
5.4
2.9
-2.5
Unit: t CO2 eq ha-1 y-1
Contribution of ΔSOC
Cropland
Natural forest
Grassland
Secondary forest
0.5
-0.7
1.2
1.6
-2.2
-0.1
0.3
-0.6
Unit: t CO2 eq ha-1 y-1
Contribution of ΔCH4 emission
Cropland
Natural forest
Grassland
Secondary forest
0.08
-0.02
0.34
-0.06
0.38
0.03
-0.45
Unit: t CO2 eq ha-1 y-1
Contribution of ΔN2O emission
Cropland
Natural forest
Grassland
Secondary forest
0.7
-0.7
-2.2
-0.7
0.6
-0.01
-0.02
Unit: t CO2 eq ha-1 y-1
Total contribution
Cropland
Natural forest
Grassland
Secondary forest
7.6
3.7
-2.9
-5.7
6.2
3.2
-3.6
Unit: t CO2 eq ha-1 y-1

**Figure 3**


Greenhouse gas emissions (Gt $CO_2$ eq.)
2000
1500
1000
450
300
150
0
Forest to croplands
Forest to grasslands
North America
Latin America
Europe
North Africa & Middle East
Tropical Africa
Former USSR
China
South & South-East Asia
Pacific developed region
World

## Supplementary Information

Table A. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post − pre) in natural forest.

| Land use type | | Location (city, country) | $CH_4$ emission (kg $CH_4$ $ha^{-1}$ $y^{-1}$) | | | $N_2O$ emission (kg $N_2O$–N $ha^{-1}$ $y^{-1}$) | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| Pre | Post | | Pre | Post | Absolute change | Pre | Post | Absolute change | |
| Forest | Crop land | Xianning, Hubei Province, Central China | | | | 0.72 | 2.11 | 1.39 | Lin et al. 2012 |
| Forest | Crop land | Xianning, Hubei Province, Central China | | | | 0.72 | 1.37 | 0.65 | Lin et al. 2012 |
| Forest | Crop land | Xianning, Hubei Province, Central China | | | | 0.72 | 1.25 | 0.53 | Lin et al. 2012 |
| Forest | Crop land | Victoria, Australia | 1.27 | 2.52 | 1.25 | 0.16 | 0.44 | 0.28 | Galbally et al. 2010 |
| Forest | Crop land | Mooloolah Valley, Queensland, Australia | -4.96 | -0.08 | 4.88 | 0.52 | 5.21 | 4.70 | Rowlings 2010 |
| Forest | Crop land | Mean ± confidence interval (CI, 95%) | | | 3.1 ± 3.6 | | | 1.5 ± 1.6 | |
| Forest | Grass land | Western Australia | -1.84 | -0.32 | 1.52 | 0.16 | 2.33 | 2.17 | Li Vesley et al 2009 |
| Forest | Grass land | Mooloolah Valley, Queensland, Australia | -4.96 | -0.04 | 4.92 | 0.52 | 1.83 | 1.31 | Rowlings 2010 |
| Forest | Grass land | Western Australia | -1.84 | -0.59 | 1.25 | 0.16 | 0.12 | -0.05 | Li Vesley et al. 2009 |
| Forest | Grass land | Mean ± CI (95%) | | | 2.6 ± 2.3 | | | 1.1± 1.3 | |
| Forest | Secondary forest | Western Australia | -1.84 | -0.80 | 1.04 | 0.16 | 0.15 | -0.02 | Li Vesley et al. 2009 |
| Forest | Secondary Forest | Mean | | | 1.04 | | | -0.02 | |

Table B. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post – pre) in crop lands.

| Land use type | | Location (city, country) | $CH_4$ emission (kg $CH_4$ $ha^{-1}$ $y^{-1}$) | | | $N_2O$ emission (kg $N_2O$–N $ha^{-1}$ $y^{-1}$) | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| pre | post | | Pre | Post | Absolute change | Pre | Post | Absolute change | |
| Crop land | Grass land | Iowa, USA | -1.07 | 0.05 | 1.12 | 12.00 | 2.60 | -9.40 | Kim et al. 2009, 2010 |
| Crop land | Grass land | Spain | -0.04 | 0.09 | 0.13 | 2.85 | 2.81 | -0.04 | Merino et al. 2004 |
| Crop land | Grass land | Mean ± CI (95%) | | | 0.6 ± 1.0 | | | -4.7 ± 9.2 | |
| Crop land | Secondary forest | Iowa, USA | -1.07 | -0.61 | 0.45 | 12.00 | 3.15 | -8.85 | Kim et al. 2009, 2010 |
| Crop land | Secondary forest | western Finland | | | | 5.50 | 11.60 | 6.10 | Maljanen et al. 2012 |
| Crop land | Secondary forest | Spain | -0.04 | -5.09 | -5.05 | 2.85 | 1.05 | -1.80 | Merino et al. 2004 |
| Crop land | Secondary forest | Mean ± CI (95%) | | | -2.3 ± 5.4 | | | -1.5 ± 8.5 | |

Table C. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post – pre) in grass lands.

| Land use type | | Location (city, country) | $CH_4$ emission (kg $CH_4$ $ha^{-1}$ $y^{-1}$) | | | $N_2O$ emission (kg $N_2O$–N $ha^{-1}$ $y^{-1}$) | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| pre | post | | Pre | Post | Absolute change | Pre | Post | Absolute change | |
| Grass land | Secondary forest | Christchurch, New Zealand | -1.52 | -0.65 | 0.87 | 0.03 | 0.09 | 0.06 | Price et al. 2010 |
| Grass land | Secondary forest | Christchurch, New Zealand | -1.52 | -5.09 | -3.57 | 0.03 | 0.16 | 0.13 | Price et al. 2010 |
| Grass land | Secondary forest | Central New South Wales, Australia | -9.37 | -7.01 | 2.36 | 0.28 | 0.28 | 0.00 | Allen et al. 2009 |
| Grass land | Secondary forest | Central New South Wales, Australia | -16.39 | -13.40 | 2.99 | 0.11 | 0.17 | 0.06 | Allen et al. 2009 |
| Grass land | Secondary forest | Central New South Wales, Australia | -10.43 | -22.16 | -11.73 | 0.28 | 0.06 | -0.22 | Allen et al. 2009 |
| Grass land | Secondary forest | South−western Australia | -8.41 | -5.35 | 3.07 | 0.22 | 0.33 | 0.11 | Allen et al. 2009 |
| Grass land | Secondary forest | South−western Australia | -4.56 | -4.64 | -0.08 | 0.17 | 0.33 | 0.17 | Allen et al. 2009 |
| Grass land | Secondary forest | South−western Australia | -4.12 | -8.41 | -4.29 | 0.84 | 0.06 | -0.78 | Allen et al. 2009 |
| Grass land | Secondary forest | South−east Queensland, Australia | -2.11 | -14.19 | -12.08 | 0.22 | 0.11 | -0.11 | Allen et al 2009 |
| Grass land | Secondary forest | South−east Queensland, Australia | -6.21 | -15.07 | -8.85 | 0.17 | 0.28 | 0.11 | Allen et al. 2009 |
| Grass land | Secondary forest | South−east Queensland, Australia | -7.09 | -30.05 | -22.96 | 0.22 | 0.17 | -0.06 | Allen et al. 2009 |
| Grass land | Secondary forest | Mean ± CI (95%) | | | -4.9 ± 4.9 | | | -0.05 ± 0.16 | |

Table D. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post − pre) in secondary forest.

| Land use type | | Location (city, country) | $CH_4$ emission (kg $CH_4$ $ha^{-1}$ $y^{-1}$) | | | $N_2O$ emission (kg $N_2O$–N $ha^{-1}$ $y^{-1}$) | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| pre | post | | Pre | Post | Absolute change | Pre | Post | Absolute change | |
| Secondary forest | Crop land | Central Sulawesi, Indonesia | -2.92 | -3.51 | -0.59 | 2.19 | 0.79 | -1.40 | Veldkamp et al. 2008 |
| Secondary forest | Crop land | Mean | | | -0.59 | | | -1.40 | |

Table E. Summary of soil methane ($CH_4$) and nitrous oxide ($N_2O$) emissions of pre and post land use change and absolute change (post − pre) in natural lands.

| Land use type | | Location (city, country) | $CH_4$ emission (kg $CH_4$ $ha^{-1}$ $y^{-1}$) | | | $N_2O$ emission (kg $N_2O$–N $ha^{-1}$ $y^{-1}$) | | | Reference |
|---|---|---|---|---|---|---|---|---|---|
| pre | post | | Pre | Post | Absolute change | Pre | Post | Absolute change | |
| Wetland | Crop land | Dalat Peat Research Station, Sarawak, Malaysia | 96.36 | 121.76 | 25.40 | | | | Inubushi et al.1998 |
| Wetland | Crop land | Gambut in South Kalimantan, Indonesia | 16.00 | 8.00 | -8.00 | -0.51 | -1.10 | -0.59 | Inubushi et al. 2003 |
| Wetland | Crop land | Gambut in South Kalimantan, Indonesia | 16.00 | 25.33 | 9.33 | -0.51 | -0.37 | 0.14 | Inubushi et al. 2003 |
| Wetland | Crop land | Heilongjiang province, China | 199.12 | 94.83 | -104.29 | 4.07 | 2.09 | -1.98 | Jiang et al. 2009 |
| Wetland | Crop land | Heilongjiang province, China | 199.12 | -1.37 | -200.49 | 4.07 | 4.90 | 0.83 | Jiang et al. 2009 |
| Wetland | Crop land | Mean ± CI (95%) | | | -56 ± 84 | | | -0.4 ± 1.2 | |

Table F. Greenhouse gas emissions in land use changes from forest to crop and grasslands in 1765 to 2005. Mean ± 95% confidence intervals.

| | Forest to Cropland (A) | | Forest to Grassland (B) | | Total (A+B) |
|---|---|---|---|---|---|
| Regions | Converted area* (million ha) | Greenhouse gas emissions (Gt $CO_2$ eq) | Converted area* (million ha) | Greenhouse gas emissions (Gt $CO_2$ eq) | Greenhouse gas emissions (Gt $CO_2$ eq) |
| North America | 137.0 ± 29.4 | 250 ± 84.5 (84.6%) | 30.5 ± 2.4 | 45.4 ± 11.5 (15.4%) | 295.3 ± 95.9 |
| Latin America | 102.3 ± 38.2 | 187 ± 63.1 (40.8%) | 181.7 ± 93.3 | 270 ± 68.4 (59.2%) | 457.0 ± 131.4 |
| Europe | 71.0 ± 32.1 | 130 ± 43.7 (84.5%) | 16.0 ± 4.7 | 23.8 ± 6.0 (15.5%) | 153.3 ± 49.8 |
| North Africa & Middle East | 7.7 ± 1.4 | 14.0 ± 4.7 (90.4%) | 1.0 | 1.49 ± 0.4 (9.6%) | 15.5 ± 5.1 |
| Tropical Africa | 48.3 ± 11.8 | 88.2 ± 29.8 (53.0%) | 52.5 ± 2.4 | 78.1 ± 19.8 (47.0%) | 166.3 ± 49.5 |
| Former USSR | 60.0 ± 40.2 | 109 ± 37.0 (82.6%) | 15.5 ± 5.7 | 23.1 ± 5.8 (17.4%) | 132.5 ± 42.8 |
| China | 56.7 ± 14.7 | 103 ± 34.9 (78.1%) | 19.5 ± 8.8 | 29.0 ± 7.3 (21.9%) | 132.4 ± 42.3 |
| South & South-East Asia | 178.7 ± 26.9 | 326 ± 110.2 (96.9%) | 7.0 ± 3.1 | 10.4 ± 2.6 (3.1%) | 336.3 ± 112.7 |
| Pacific developed region | 13.3 ± 2.5 | 24.3 ± 8.2 (73.1%) | 6.0 | 8.93 ± 2.3 (26.9%) | 33.2 ± 10.5 |
| World | 674.3 ± 16.3 | 1230 ± 415.5 (74.7%) | 280.3 ± 10.4 | 417.1 ± 105 (25.3%) | 1647.1 ± 521.0 |

* Data from Meiyappan and Jain (2012)

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