Global Change Biology

Global Change Biology (2011) **17**, 425–438, doi: 10.1111/j.1365-2486.2010.02220.x

OPINION

The greenhouse gas value of ecosystems

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Abstract

As society faces the urgent need to mitigate climate change, it is critical to understand how various ecosystems contribute to the climate, and to express these contributions in terms that are meaningful to policymakers, economists, land managers, and other nonscience interest holders. Efforts to mitigate climate change call for quantification of the full greenhouse gas (GHG) effects of land use decisions, yet we lack an appropriate metric of the full GHG implications of maintaining a given ecosystem over a multiple year time frame. Here, we propose the concept of greenhouse gas value (*GHGV*) of ecosystems, which accounts for potential GHG release upon clearing of stored organic matter, annual GHG flux, and probable GHG exchanges resulting from disturbance. It treats these ecosystem–atmosphere exchanges in a time-sensitive manner, thereby providing an appropriate framework for computing of the GHG consequences of any land use decision. To illustrate this concept, we provide estimates of the *GHGV* of various biome types (based on data compiled from the literature), disturbance regimes, and decisions on the treatment of time. We show that natural ecosystems generally have high *GHGV*'s, whereas managed ecosystems generally have lower or negative *GHGV*'s; that *GHGV* decreases with increasing probability of disturbance, and that decisions on the treatment of time can be important, affecting some ecosystem types more strongly than others. In addition, we show how *GHGV* may be used to quantify the full GHG effects of land-use or land-cover change in a thorough and rigorous manner. Finally, we provide comparisons of *GHGV* to other major paradigms for valuing the GHG contributions of ecosystems, showing that – for many purposes – *GHGV* is the most appropriate method of quantifying the GHG services of ecosystems.

Keywords: biofuels, biomes, carbon dioxide (CO_2), disturbance, ecosystem-atmosphere exchange, ecosystem services, fire, land-use/land-cover change, methane (CH_4), nitrous oxide (N_2O)

Received 25 January 2010 and accepted 8 March 2010

Introduction

As society faces the urgent need to mitigate climate change, it is increasingly important to understand the contributions of terrestrial ecosystems to climate. This is particularly important in the face of rapid land-use change, which may accelerate because of intensifying pressure from agriculture (e.g., Searchinger *et al*., 2008). Substantial changes in ecosystem structure may also occur as a result of climate change, at times causing complete displacement of biomes (e.g., Scholze *et al*., 2006). Such land cover changes typically entail considerable greenhouse gas (GHG) release upon clearing of the original ecosystem (e.g., Fearnside, 2000; Fargione *et al*., 2008) or long-term changes in annual GHG flux (e.g., Robertson *et al*., 2000). Land management decisions thereby contribute meaningfully to global GHG budgets (Houghton, 2007; Smith *et al*., 2007), and efforts to mitigate GHG emissions from terrestrial ecosystems therefore require rigorous accounting for the GHG services of ecosystems.

There are four main ways in which the GHG services of ecosystems are commonly valued (Table 1). First, ecosystems may be valued for their storage of organic matter that would be released as GHG's upon land clearing (Dixon *et al*., 1994; Miles & Kapos, 2008). This value is the focus of the UN Framework Convention on Climate Change's (UNFCCC) program on Reducing Emissions from Deforestation and Forest Degradation (REDD-plus; Miles & Kapos, 2008; UNFCCC, 2008, 2009), which would provide a financial incentive for developing countries to reduce deforestation. Second, ecosystems may be valued based on measured or projected GHG flux (e.g., CO_2 sequestration or N_2O release) over a time span of interest (e.g., Robertson *et al*., 2000; Lal, 2004; Righelato & Spracklen, 2007; Anderson-Teixeira *et al*., 2009; Smeets *et al*., 2009). In some cases, when a multi-year time frame is

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Table 1 Comparison of metrics of the GHG contributions of ecosystems

Metric	Description	Policy examples	Multiyear	Sensitive to timing of emissions	Accounting for: Organic matter storage	Accounting for: Annual GHG flux	Accounting for: Probable disturbance effects
Organic Matter Storage	Quantification of the GHG's that would be released if the ecosystem were to be cleared. Many applications consider CO_2 only, and do not necessarily quantify all organic matter pools	Original version of REDD (Miles & Kapos, 2008; UNFCCC, 2008)	−/+	−	+	−	−
GHG Flux	Annual ecosystem-atmosphere GHG exchange, sometimes extrapolated over multiple years	Chicago Climate Exchange (CCX, 2009)	−/+	−	−	+	−
Annual GHG Inventory	Annual reporting of GHG emissions – generally from managed ecosystems (e.g., IPCC guidelines for reporting under UNFCCC)	IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006); REDD-plus (UNFCCC, 2009)	−	na	+*	+	+*
Storage and Flux	Combined contributions from storage and flux over a multiyear time span without proper accounting for the treatment of time. Applications (e.g., biofuels life cycle analyses) vary in comprehensiveness.	U.S. Renewable Fuels Standard (US EPA, 2009); California Low Carbon Fuel Standard (California EPA, 2009)	+	−†	+	+	−
GHGV	Accounting for flux, storage, and probable effects of disturbance over a multiyear time period with proper accounting for time		+	+	+	+	+

*Annual GHG inventories count storage and disturbance in the sense that emissions from GHG emissions from clearing or fire would be counted in annual inventories; however, storage and risk of disturbance *per se* do not affect these inventories.

†U.S. EPA's Renewable Fuel Standard proposed rule (US EPA, 2009) applied discounting for their 100-year analyses, but was otherwise insensitive to the timing of emissions.

considered, GHG fluxes are weighed by the year in which they occur. A prominent application of this metric is the Chicago Carbon Exchange, where land management-induced mitigation of annual GHG emissions can be sold as carbon offsets (CCX, 2009). Similar methods of measuring domestic carbon offsets may be implemented under a federal cap-and-trade policy in the United States [e.g., American Clean Energy and Security Act (H.R. 2454); US House of Representatives, 2009]. This method of valuing ecosystems also was used in early biofuel life-cycle analyses that did not consider loss of carbon storage through land use change (e.g., Tilman *et al*., 2006; Adler *et al*., 2007). Third, contributions from both release of stored organic material and annual GHG flux may be counted through inventories of ecosystem-atmosphere exchanges over relatively short (e.g., annual) time periods. This is the approach used for national inventories of GHG emissions from managed land as described by the Intergovernmental Panel on Climate Change (IPCC, 2006). Fourth, the combined contributions of storage and flux may be evaluated over longer time periods, as is commonly done in life cycle analyses that seek to determine the net GHG effects of biofuel production (e.g., Searchinger *et al*., 2008; California EPA, 2009; Melillo *et al*., 2009; US EPA, 2009).

All four of these approaches for estimating the GHG value of ecosystems face certain limitations (Table 1). Those that consider only organic matter storage or only flux – while appropriate for certain applications – miss the value associated with the other. For example, basing a forest's value on the GHG release that would be incurred by its destruction misses the value associated with annual C sequestration, whereas basing an ecosystem's value solely on its GHG flux misses the value of its stored organic material. These approaches typically do not employ an appropriate strategy for quantifying the effects of emissions that occur over multiple years (e.g., annual fluxes, decomposition of wood after forest clearing). Multiyear paradigms often err in their treatment of the timing of emissions, as proper accounting for time over multiyear time frames precludes a simple summation of emissions expressed as CO_2-equivalents (O'Hare *et al*., 2009). Annual inventories of GHG exchange are valid for purposes such as reporting of national GHG emissions under the UN Framework Convention on Climate Change (IPCC, 2006); however, they deal with accomplished emissions on a year-to-year basis and do not provide the structure for understanding the full climate impacts of land-use/land-cover change (henceforth, LULCC) over a multiyear time frame. Finally, these approaches typically do not account for probable effects of natural disturbance (but see IPCC, 2006), which may strongly affect the value of preserving an ecosystem. For example, it may be less beneficial to preserve a forest that is likely to burn, as a burn would result in significant GHG release. Thus, there is a strong need for a method of valuing the GHG contributions of ecosystems that accounts for contributions from organic matter storage, annual flux, and probability of disturbance – and does so over multiple years in a time-appropriate manner.

Here, we propose a quantitative metric for the greenhouse gas value of ecosystems (*GHGV*), which quantifies the value of maintaining an ecosystem over a multiple-year time frame. It incorporates changes in ecosystem-atmosphere GHG exchanges that would occur if the ecosystem were to be cleared, including release of GHG's through oxidation of stored organic material, displaced annual GHG flux, and displaced probable effects of disturbance. Through proper accounting for the timing of emissions, it quantifies the total radiative forcing that would occur upon clearing of the ecosystem. *GHGV* is expressed relative to the effects of a pulse CO_2 emission such that its units are expressed in CO_2-equivalents, which are useful in that they serve as a currency for emissions regulation and offset trading. Following the theoretical development of *GHGV*, we use it to provide illustrative calculations of *GHGV* for various biomes, disturbance regimes, and decisions on the treatment of time. We also discuss how the *GHGV*'s of two ecosystems may be combined to quantify the full GHG effects of LULCC.

Theoretical development of *GHGV*

We define an ecosystem's *GHGV* as the total benefit of avoiding radiative forcing from GHG's through maintenance of 1 ha of the ecosystem. *GHGV* incorporates potential GHG release upon clearing of stored organic matter, the annual flux of GHG's from the ecosystem to the atmosphere, and probable GHG exchange resulting from disturbance. Ecosystem–atmosphere GHG exchanges are counted over an emissions time frame τ_E (years), which is an arbitrarily chosen time span over which the ecosystem's GHG impact is assessed (see below for discussion of τ_E). Because GHG's remain in the atmosphere – and thereby impact the climate – for many years following their release, it is generally desirable to evaluate the climate impact of ecosystem–atmosphere GHG exchange (i.e., cumulative radiative forcing) over a longer analytical time frame, τ_A (years; analogous to 'time horizon' for global warming potentials; Forster *et al*., 2007). Radiative forcing may be weighted over time according to its potential to trigger dangerous climate change; here, we employ a weighting function, $w(t_A)$ (default $w = 1$), which is discussed below. Analogous to the global warming potential

metric (Forster *et al*., 2007) but differing in its treatment of a multiple-year time emissions time span, *GHGV* is expressed in CO_2-equivalents (Mg CO_2-eq ha^{-1}) through comparison with the cumulative radiative forcing that would arise from a pulse emission of CO_2 at time zero. Formally, *GHGV* is as follows:

$$GHGV_{\tau_A}^{\tau_E} = \frac{\int_{t_A=o}^{\tau_A} \left[RF_{GHG}^{\tau_E}(t_A)w(t_A)\right]dt_A}{\int_{t_A=0}^{\tau_A} \left[RF_{pCO_2}(t_A)w(t_A)\right]dt_A}. \quad (1)$$

Here, $RF_{GHG}^{\tau_E}(t_A)$ (nW m^{-2} ha^{-1} ecosystem yr^{-1}) is the additional radiative forcing at time t_A that would arise from GHG's released over time span τ_E following the clearing 1 ha of the ecosystem, and $RF_{pCO_2}(t_A)$ (nW m^{-2} yr^{-1}) is the additional radiative forcing that would arise from a pulse emission of 1 Mg CO_2 at $t_A = 0$. The numerator of this equation gives the ecosystem's value in terms of radiative forcing (nW m^{-2} ha^{-1} ecosystem), while the denominator translates it into CO_2-equivalents.

For each year in the analytical time span, additional radiative forcing ($RF_{GHG}^{\tau_E}$ and RF_{pCO_2}) is calculated by summing the radiative forcings from of all pertinent GHG's (subscripted x):

$$RF(t_A) = \sum_x a_x C_x^{\tau_E}(t_A). \quad (2)$$

Here, a_x is the effective radiative efficiency of GHG species x ($a_{CO_2} = 1.4 \times 10^4$, $a_{CH_4} = 4.9 \times 10^5$, and $a_{N_2O} = 3.03 \times 10^6$nW m^{-2}ppb^{-1}; Forster *et al*., 2007). [For methane, this is calculated by multiplying the radiative efficiency of methane (3.7×10^5 n W m^{-2} ppb^{-1}; Forster *et al*., 2007) by 4/3 to correct for indirect effects from enhancements to ozone and stratospheric water vapour. This gives very similar results to the IPCC's global warming potential estimates for 20-, 100-, and 500-year time horizons (Forster *et al*., 2007)]. This is multiplied by the *additional* atmospheric abundance of GHG x attributable to clearing of 1 ha of the ecosystem, $C_x^{\tau_E}$ (ppb ha^{-1} ecosystem yr^{-1}), which varies as a function of time. In turn, $C_x^{\tau_E}(t_A)$ is determined based on the change in GHG fluxes that would occur as a result of clearing 1 ha of land, integrated over the emissions time period of interest (τ_E):

$$C_x^{\tau_E}(t_A) = \int_{t_E=0}^{\min(\tau_E, t_A)} \left[\frac{I_x(t_E)}{A}\rho_x(t_A - t_E)\right]dt_E. \quad (3)$$

Here, I_x (kmol x ha^{-1} yr^{-1}) is the input of GHG x from the ecosystem to the atmosphere and A is the moles of air in the atmosphere ($A = 1.78 \times 10^8$ billion kmol). The portion of remaining in the atmosphere at time t_A, ρ_x, is calculated based on the decay of a pulse of GHG x over time (see Forster *et al*. (2007) for ρ_x's used here). I_x is as follows:

$$I_x(t_E) = S_x(t_E) - F_x(t_E) - D_x(t_E). \quad (4)$$

Here, S_x, F_x, and D_x (kmol x ha^{-1} yr^{-1}) are the potential release of GHG's from stored organic material upon land clearing, displaced annual flux, and displaced probable emissions from natural disturbance, respectively. These are detailed below.

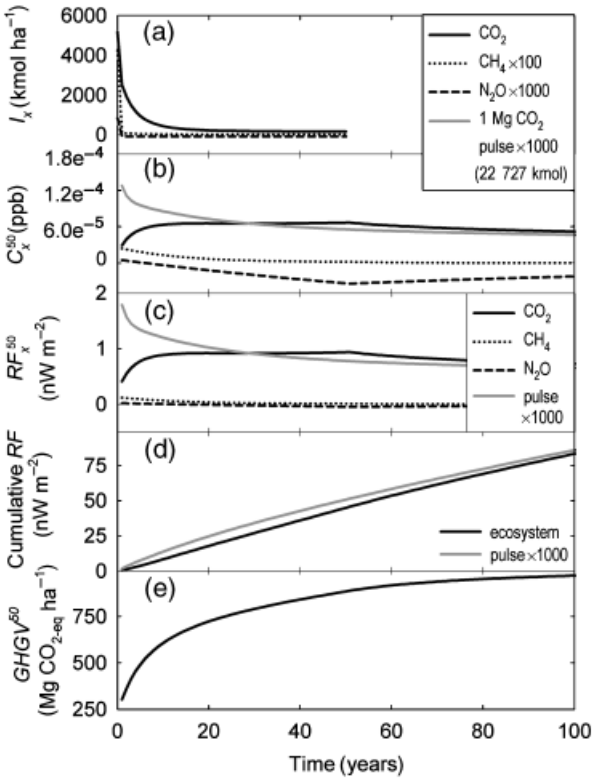

Fig. 1 Example of how a time series of ecosystem-atmosphere GHG exchange translates into *GHGV* for a tropical forest cleared by burning (see SI; Fig. 2). This illustration uses a 50-year emissions time span (τ_E), no discounting [Eqn (1)], and no probability of disturbance [Eqns (4) and (6)]. (a) 50 year time series of ecosystem-atmosphere exchange of for CO_2, CH_4, and N_2O [I_x; Eqns (3) and (4)]; (b) resulting additional atmospheric abundance of these GHG's through time [$C_x^{\tau_E}$; Eqns (2) and (3)]; (c) resulting radiative forcing from each GHG through time [Eqn (2)]; (d) cumulative radiative forcing from all GHG's combined [$RF_{GHG}^{\tau_E}$; Eqns (1) and (2)]; (e) *GHGV* [Eqn (1)] as a function of the analytical time frame. Also illustrated are the effects of a 1 Mg CO_2 pulse at time 0, which is used as a basis of comparison to express *GHGV* in CO_2-equivalents [Eqn (1)]. For display purposes, some values are multiplied by 100 or 1000 (see legends).

Figure 1 illustrates how the time series of ecosystem-atmosphere GHG exchange, $I_x(t_E)$, translates into *GHGV* based on Eqns (1)–(4).

Contributions of stored organic matter (S)

Ecosystems store organic material that is oxidized upon land clearing – either immediately through combustion or over time through decomposition – resulting in GHG release. We define the storage of 'vulnerable' organic material as all aboveground and belowground biomass, dead wood, the organic layer consisting of litter or peat, and any soil organic material (SOM) in the top meter of mineral soil that would be released through long-term annual tillage. Such a SOM baseline is necessary because a large portion of SOM exists in relatively stable forms or deep soil layers and would be minimally affected by LULCC. Annually tilled croplands provide an appropriate baseline, as their soils are the most carbon-depleted among common land use types (Guo & Gifford, 2002), containing on average approximately 30% less SOM in the top meter than their undisturbed counterparts (Davidson & Ackerman, 1993; Guo & Gifford, 2002; Murty *et al*., 2002). Thus, in cases where annually tilled ecosystems are not available for comparison, a reasonable default for native ecosystems is to include 30% of SOM in the top meter of mineral soil. Note that organic matter storage is not necessarily equal to the reduction in organic material that occurs during LULCC; while the baseline serves to allow consistent comparison across sites, LULCC calculations will consider the *change* in organic matter storage (see below for discussion of land-use/land-cover change).

Translation of organic matter storage into GHG release as a function of time [$S_x(t_E)$; kmol x ha^{-1} yr^{-1}; Eqn (4)] is sensitive to the mechanism of land clearing and the distribution of organic material among different pools (e.g., combustible biomass, roots, soil organic matter). For each pool (subscripted p), GHG release is separated into an initial (i.e., $t_E = 0$) release through combustion – if fire is used for land clearing – and a subsequent ($t_E > 0$) release of the remaining material through decomposition – either onsite or off-site as wood products:

$$S_x(t_E) = \sum_p \left(OM_p \begin{cases} f_p^c E_{x,p}^c & t_E = 0 \\ (1 - f_p^c) E_{x,p}^d d_p(t_E) & t_E > 0 \end{cases} \right). \quad (5)$$

Here, OM_p (Mg dry matter ha^{-1}) is the organic matter in each pool, and f_p^c and $(1-f_p^c)$ are the fractions of organic matter in that pool oxidized through combustion and decomposition, respectively. The terms $E_{x,p}^c$ and $E_{x,p}^d$ (kmol x Mg^{-1} dry matter) represent the proportional release of GHG x for each unit of biomass oxidized through combustion or decomposition, respectively, and $d_p(t_E)$ is the proportion of organic matter in pool p that decomposes each year ($0 \leq d_p(t_E) \leq 1$). Typically, $d_p(t_E)$ will be characterized by an exponential decay function. We note that – when fire is used in the process of land clearing – S will be particularly sensitive to burn characteristics (f_p^c and $E_{x,p}^c$), which may be highly variable within a single ecosystem type.

Biomass burning releases a wide range of trace GHG's, including indirect GHG's such as carbon monoxide (CO), non-methane volatile organic compounds, and mono-nitrogen oxides (NO_x) (e.g., Andreae & Merlet, 2001). In theory, the climate impacts of all GHG's should be assessed; however, this is complicated by high uncertainty in both emission factors and global warming potential estimates for some of the less abundant GHG's (e.g., Andreae & Merlet, 2001; Forster *et al*., 2007). For release through decomposition, the most important GHG species is CO_2, although decomposition of stored organic material also sometimes releases significant quantities of CH_4 (i.e., from termite-mediated decomposition) or N_2O (i.e., peatland organic soils).

Contributions of annual flux (F)

The *GHGV* of ecosystems also includes their potential to sequester or release GHG (e.g., Robertson *et al*., 2000; Robertson & Grace, 2004). This flux of GHG's to or from the ecosystem (F; Eqn (4); positive value indicates release to atmosphere) is dominated by the GHG's CO_2, CH_4, and N_2O. The net flux of CO_2, or net ecosystem exchange, is dominated by carbon release through respiration and uptake through photosynthesis, but also includes non-respiratory fluxes (e.g., fire, ultraviolet oxidation of organic matter; Chapin *et al*., 2006). It generally approximates – but is not identical to – the net change in ecosystem carbon storage (Chapin *et al*., 2006). Any carbon removed from the ecosystem through harvest that will be returned to the atmosphere within a short time frame (e.g., through human consumption and respiration, or through biofuels burning) should be subtracted from F_{CO_2}. Fluxes of CH_4 and N_2O are typically small in quantity but potent because of the high global warming potential of these gases, and thereby can strongly affect the *GHGV* of some ecosystems. For managed ecosystems, a complete GHG budget must also include associated anthropogenic emissions, such as those from farm machinery (i.e., fuel combustion emissions) and those associated with agricultural inputs (e.g., emissions from lime and fertilizer production; Robertson *et al*., 2000; West & Marland, 2002; Robertson & Grace, 2004).

Calculating *GHGV* requires values for F_x over the entire time span of interest (Eqn (4)). As a complete time course of GHG fluxes is methodologically difficult to obtain, it will often be necessary to approximate F as a constant through time, or, for aggrading ecosystems, as simple step function (e.g., see Supporting Information). In reality, however, F changes continuously as ecosystems age (Odum, 1969; Robertson *et al*., 2000; Law *et al*., 2003; Litvak *et al*., 2003; Bond-Lamberty *et al*., 2004; Luyssaert *et al*., 2008). We note that the course of GHG fluxes as a function of ecosystem age remains controversial for CO_2 (e.g., Bond-Lamberty *et al*., 2004; Magnani *et al*., 2007; Luyssaert *et al*., 2008) and has not yet been extensively studied for CH_4 and N_2O (but see Priemé *et al*., 1997; Robertson *et al*., 2000; Peichl *et al*., in press). In some cases, resolving these dynamics will be key to producing reliable estimates of *GHGV*.

*Probable contributions from disturbance (*D*)*

Natural disturbances are stochastic, yet stand to strongly impact the *GHGV* of ecosystems. The possibility of multiple types of disturbances of varying severity and frequency, as well as repeat disturbances during the time span of interest makes calculation of the full effects of disturbance complex. In theory, relatively frequent, small natural disturbances (e.g., small-scale mortality, tree-fall events, surface fires) are incorporated into annual GHG flux measurements. The extent to which this actually occurs is mixed. Eddy-covariance flux measurements and biometric estimates of annual carbon flux capture individual mortality, tree-fall events, and – to the extent that researchers do not manipulate their frequency – surface fires. Chamber measurements (e.g., for trace GHG's) are more likely biased to avoid these disturbances.

Here, we incorporate disturbance by adjusting for the probabilistic impacts of large, infrequent disturbances (e.g., stand-clearing forest fires, hurricanes, insect outbreaks). Our approximation assumes that at most one disturbance occurs during the time span of interest, which is reasonable when the disturbance rate (r_D; yr^{-1}) is low and/or the time span of interest (τ_E) is short (i.e., $r_D\ \tau_E \ll 1$), or when repeat disturbance is unlikely to occur shortly following a disturbance. It also assumes that disturbance rate remains constant over the time span of interest.

The expected contribution of GHG x from disturbance, D_x, is as follows:

$$D_x(t_E) = r_D[S_x^D(t_E) - F_x^D(t_E)] \quad (6)$$

Here, ${S_x}^D$ ($kmol\,x\,ha^{-1}\,yr^{-1}$) is the expected GHG release from disturbance, and ${F_x}^D(t_E)$ ($kmol\,x\,ha^{-1}\,yr^{-1}$) is the probable net change in cumulative GHG flux arising from recent disturbance. For each year within the emissions time span, potential GHG release through disturbance, ${S_x}^D$, is calculated by summing the possible GHG exchange from disturbance in that or any previous year [this is then adjusted for the probability of disturbance in Eqn (6)]:

$$S_x^D(t_E) = \sum_p \left(\sum_{t^*=1}^{t_E} \left(OM_p^D \begin{cases} f_p^{cD} E_{x,p}^c & t^* = t_E \\ \left[(1 - f_p^{cD}) E_{x,p}^d d_p^D(t_E - t^*)\right] & t^* < t_E \end{cases} \right) \right) \quad (7)$$

Here, OM_p^D ($Mg\,ha^{-1}$) is the storage of organic matter in pool p subject to release by the disturbance. It will often be less than the organic matter subject to release through anthropogenic clearing [OM_p; Eqn (5)], as mortality may not be complete and the soil will generally remain relatively undisturbed. Likewise, the portion of biomass that burns (f_p^{cD}) and the decay function of remaining biomass (d_p^D) may differ from those that describe land-clearing [Eqn (5)]. The variable t^* refers to the year in which potential disturbance occurs.

During the period of recovery following a disturbance, ecosystems have altered GHG flux rates (e.g., Bormann & Likens, 1994) and this effect is quantified by ${F_x}^D(t_E)$:

$$F_x^D(t_E) = \min(t_E, t^R)(F_x^R - F_x) \quad (8)$$

Here, t^R is the time it takes for the ecosystem to 'recover' to its pre-disturbance state, and the term '$\min(t_E, t^R)$' reflects the fact that any disturbance since the beginning of the time span and within this recovery period would impact the flux at time t_E. The term (${F_x}^R - F_x$) expresses the difference in annual flux between the recovering ecosystem and a counterpart that has not been recently disturbed. This assumes a constant difference between F^D and F over time, which is not realistic, but is a useful first approximation.

Disturbance has an important impact on *GHGV*, and the probability of disturbance should not be ignored in land use decisions. At the same time, challenges remain to accurately quantifying the effects of disturbance. First, disturbance rate is not necessarily easy to predict, as climate change and other anthropogenic impacts are altering disturbance frequencies (e.g., Westerling *et al*., 2006). Additionally, our current treatment of disturbance accounts for only one disturbance type and size, and does not allow repeat disturbances in the time frame of interest. While valuable in quantifying the effects of infrequent stand-clearing disturbances, our model [Eqns (6)–(8)] will require further development

to capture the impacts of a full suite of possible disturbances.

Treatment of time

There are three time-related issues that can impact the value of *GHGV*: the choice of an emissions time frame [τ_E; Eqn (3)], the choice of an analytical time frame [τ_A; Eqn (1)], and the (optional) application of a weighting function, [$w(\tau_A)$; Eqn (1)].

Emissions time frame (τ_E)

GHGV is dependent upon the time frame over which ecosystem-atmosphere GHG exchange is counted (emissions time span; τ_E). In an immediate sense, the *GHGV*'s of ecosystems are strongly dependent on GHG release from stored organic material [*S*; Eqn (5)] – particularly combustion emissions – and, as τ_E increases, contributions from annual flux (*F*) and probable effects of disturbance (*D*) become increasingly important. There is no 'correct' τ_E; rather, the appropriate time frame depends upon the nature of the application. There are, however, a few criteria that should be observed in the selection of τ_E. First, τ_E should not be too short to capture the decomposition fated to occur by the clearing of the ecosystem. The τ_E required to capture this will depend upon organic matter storage and decomposition rates [Eqn (5)]. On the other hand, τ_E could be excessively long if it exceeds the time frame over which conditions can be expected to remain reasonably predictable. In particular, the high likelihood that global change will impact annual GHG exchange (*F*) and disturbance frequency in many ecosystems (e.g., Westerling *et al*., 2006; Field *et al*., 2007) implies increasing uncertainty at longer time scales. Finally, for LULCC applications, one reasonable criterion for τ_E would be the expected duration of the new ecosystem. For example, analyses of the GHG impacts of biofuels-related land use may select τ_E based upon the expected duration of ethanol production (e.g., Righelato & Spracklen, 2007; Searchinger *et al*., 2008; O'Hare *et al*., 2009).

Analytical time frame (τ_A)

GHGV is also dependent upon the time over which the climate impacts are evaluated (analytical time frame; τ_A). Whereas cumulative radiative forcing continues to increase over hundreds of years, normalization by a CO_2 pulse causes *GHGV* of most ecosystems to stabilize within about 150 years after the cessation of emissions. As with τ_E, the choice of τ_A is subjective, but should be guided by several considerations. Logically, τ_A must be greater than or equal to τ_E. To capture the full effects of fluxes over τ_E, it should exceed τ_E by at least 50 years. Many applications select $\tau_A = 100$ years through use of the IPCC's global warming potential values for a 100-year time horizon (Forster *et al*., 2007). Thus, for most applications, use of $\tau_A = 100$ years would be a logical choice that is consistent with other literature.

If the climate change impacts of land use decisions are being evaluated relative to a certain target date, τ_E and τ_A should both equal the number of years until that target date.

Weighting by time of emission [w(t_A)]

It may be desirable to place greater weight on current than on future emissions. Earlier emissions, for example, may be more likely to determine the fate of climate change in that they could trigger feedback mechanisms or push the climate system past critical damage thresholds (e.g., Lenton *et al*., 2008). In addition, because society tends to place more value on near-term than on long-term costs and benefits, economic and policy applications often apply an annual discount rate to future emissions (e.g., Kim *et al*., 2008). Weighting by the timing of emissions can easily be applied to *GHGV* through $w(t_A)$ [Eqn (1); $w = 1$ for no weighting]. One weighting function commonly used in economics is $w(t_A) = (1/(1+r))^{t_A}$, where r is the annual discount rate (typically, $r < 0.1$; e.g., Kim *et al*., 2008; US EPA, 2009). Criteria for selecting $w(t_A)$ are beyond the scope of this analysis.

Illustrative *GHGV* calculations

To provide estimates of *GHGV* of various ecosystem types and to illustrate the effects of disturbance and the treatment of time (Figs 2–4), we compiled published estimates of pertinent variables [Eqns (5)–(8)] for a variety of ecosystem types (see Supporting Information for details and parameter estimates). *GHGV* was calculated according to the above equations using Matlab 7.7.0 (Mathworks Inc.). Matlab code and an accompanying spreadsheet (Microsoft® Excel) are provided as supporting information. Estimates included GHG species CO_2, CH_4, and N_2O, which are the most prominent GHG's exchanged between ecosystems and the atmosphere.

GHGV estimates for various biomes

We calculated *GHGV* for a variety of native, aggrading (recently disturbed or abandoned), and managed ecosystem types (Table S1). For consistency, all examples represent a disturbance of fire followed by on-site decomposition of unburned material. The probability

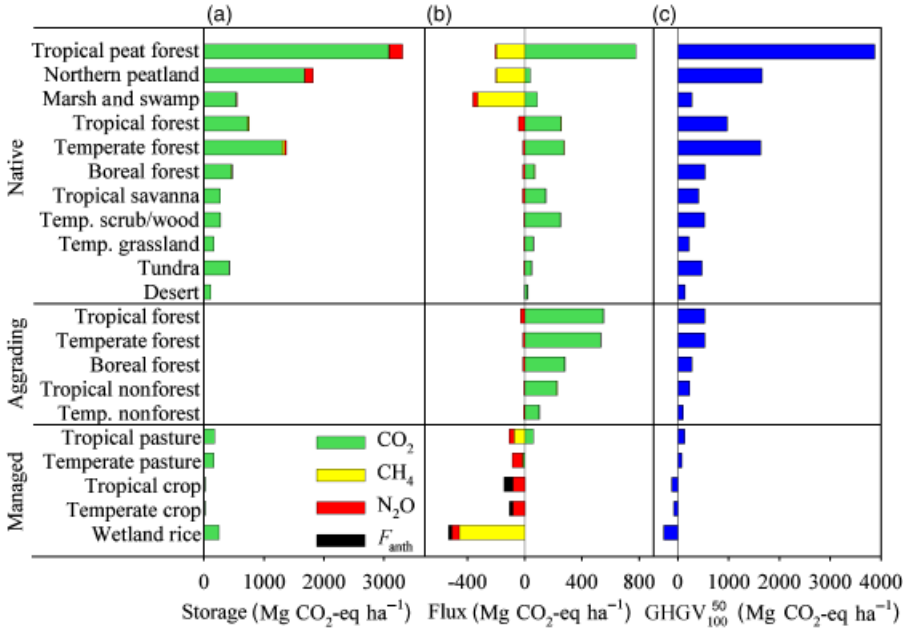

Fig. 2 Contributions to *GHGV* from (a) storage of materials vulnerable to release as GHG's upon land clearing [*S*; Eqn (5)] and (b) displaced flux of CO_2, CH_4, and N_2O, and anthropogenic emissions (F_{anth}). These are combined to yield (c) greenhouse gas value for a 50-year emissions time span and 100 year analytical time span, $GHGV_{100}^{50}$. Here, effects of disturbance are not included, and no discounting is applied. Values are presented in tabular form in Table S9.

of natural disturbance was not incorporated into these estimates (but see below for examples with natural disturbance). In this illustration, we selected an emissions time span of 50 years and an analytical time span of 100 years, and discounting was not applied (see below for examples with variation in time variables).

Contribution to *GHGV* from stored organic material [Eqn (5); Fig. 2a; Table S9] is closely linked to total organic matter storage (Table S2), and also is affected by burn characteristics (f_p^c and $E_{x,p}^c$; Eqn (5); Table S3) and the timing of release of decomposing organic material [$d_p(t_E)$; Eqn (5); Table S4]. The contribution to *GHGV* from *S* ranged from minimal values in current or abandoned cropland (<5 Mg CO_2-eq ha^{-1}) up to 3310 Mg CO_2-eq ha^{-1} in tropical peat forests. This was dominated by potential CO_2 release, which represented at least 90% of the contribution from *S* in all ecosystems. Potential releases of CH_4 and N_2O contributed up to 3% and 8%, respectively.

Contributions to *GHGV* from annual GHG flux tended to be positive in natural ecosystems (with the exception of wetlands) and negative in managed ones, ranging from 524 Mg CO_2-eq ha^{-1} in aggrading tropical forests to −535 Mg CO_2-eq ha^{-1} in wetland rice (Fig. 2b; Table S9). It was not consistently dominated by any one GHG species. In unmanaged ecosystems with unsaturated soils, contributions from CO_2 flux (Table S5) dominated and were consistently positive. That is, native ecosystems tended to be carbon sinks (albeit very high variability in NEE), as has been previously observed (Law *et al*., 2002; Luyssaert *et al*., 2007). Aggrading ecosystems were consistently carbon sinks, with C sequestration rates increasing from cool to warm climates (Fig. 2b; see also Anderson *et al*., 2006). In contrast, CO_2 flux was minimal in managed ecosystems, where flux contributions from CH_4 and N_2O prevailed. Annual methane flux (Table S6) reduced $GHGV_{100}^{50}$'s of all wetlands (up to −460 Mg CO_2-eq ha^{-1} in wetland rice), and also contributed meaningfully to the $GHGV_{100}^{50}$'s of pastures. Its contributions were minimal for ecosystems with unsaturated soils and no livestock (3–5 Mg CO_2-eq ha^{-1}). Because of high uncertainty associated with methane flux estimates from soil (Le Mer & Roger, 2001), uncertain CH_4 contributions from plants (Houweling *et al*., 2006; Keppler *et al*., 2006), and high variability in cattle density, methane flux remains highly uncertain for many ecosystem types. Nitrous oxide flux (Table S7) contributed minimally to the *GHGV* of native and aggrading ecosystems (−3 to −44 Mg CO_2-eq ha^{-1}), but was important in managed ecosystems, where its contributions ranged from −36 Mg CO_2-eq ha^{-1} in tropical pastures to −82 Mg CO_2-eq ha^{-1} in croplands. In croplands, CO_2 emissions associated with fertilizer production, lime, and fuel use by farm machinery (Table S8) contributed −25 to −63 Mg CO_2-eq ha^{-1}, depending strongly on assumed crop management practices.

$GHGV_{100}^{50}$ (Fig. 2c, Table S9) was consistently positive in natural ecosystems, with relatively modest values in dry or cold climates ($<600\,Mg\,CO_2$-eq ha^{-1}) and high values in forests ($>950\,Mg\,CO_2$-eq ha^{-1} in tropical and temperate forests). Despite large methane releases from peatlands (outweighing the benefits of CO_2 sequestration in northern peatlands; see also Roulet, 2000; Laine *et al*., 2006), they had the highest $GHGV_{100}^{50}$'s ($>1600\,Mg\,CO_2$-eq ha^{-1}) because of substantial storage of carbon in peat. Aggrading ecosystems had modest $GHGV_{100}^{50}$'s, ranging up to $529\,Mg\,CO_2$-eq ha^{-1} in aggrading tropical forests. Despite negative contributions from F_{GHG}, moderately grazed pastures had positive $GHGV_{100}^{50}$'s owing to relatively high S_{GHG}'s (Fig. 2). Crop ecosystems, on the other hand, had negative $GHGV_{100}^{50}$'s ($<-80\,Mg\,CO_2$-eq ha^{-1} for both temperate and tropical crops), which were dominated by contributions from N_2O and associated anthropogenic emissions. Thus, in general, unmanaged ecosystems and moderately stocked pastures are acting to mitigate GHG's through storage of organic material and GHG uptake, whereas crop ecosystems are GHG sources.

We caution that the values presented here should not be used as off-the-shelf estimates of *GHGV* for any particular ecosystem. Influential variables such as organic matter storage, burn characteristics, cattle density, and crop management practices can vary by orders of magnitude within some of the biomes considered here. Therefore – particularly for influential and often easily measured variables such as these – *GHGV* should be calculated based on the best available data for the specific ecosystem(s) of interest.

Effect of forest fires on GHGV

To illustrate the effects of natural disturbance on *GHGV*, we calculated the *GHGV* of tropical, temperate, and boreal forests across a range of stand-clearing fire frequencies (Fig. 3). Specifically, we considered fire frequencies [r_D; Eqn (6)] ranging from 0 (no fires) to $0.015\,yr^{-1}$ (66 year fire cycle), which encompasses the range of fire frequencies typically observed for all three forest types (e.g., Turner & Romme, 1994; van der Werf *et al*., 2008). We assumed that any unburned biomass or woody debris would be committed to release through decomposition (all vegetation killed) but that SOC would be unaltered. For all three GHG species, we assumed annual GHG flux during recovery (F_x^R; Eqn (8)) to be equal to GHG flux of the corresponding aggrading ecosystems during the time of recovery [Eqn (8); t^R estimated at 23, 72, and 75 years for tropical, temperate, and boreal forests, respectively; see Supporting Information for details]. As in the above example, we used values of $\tau_E = 50$ and $\tau_A = 100$, and no discounting was applied.

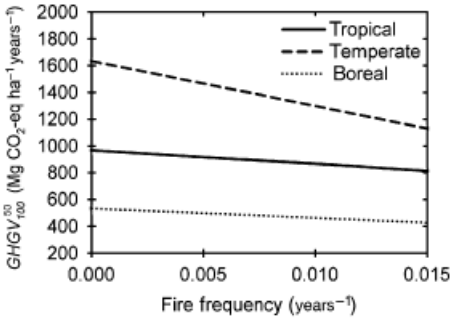

Fig. 3 Influence of the probability of a stand-clearing forest fire on $GHGV_{100}^{50}$ of tropical, temperate, and boreal forests. Storage and flux components are calculated as in Fig. 2. See text for details.

Assuming no repeat disturbances, $GHGV_{100}^{50}$ decreases linearly with fire frequency (Fig. 3). The main effect of disturbance arises from the probability that GHG's will be released through combustion and subsequent decomposition [S^D; Eqns (6) and (7)]. This is partially offset by increased C uptake in the recovering forest [F^D; Eqns (6) and (8)]; however, even the most rapid recovery would not compensate for GHG releases through fire. Differences in the slope of the *GHGV*-fire frequency relationship (Fig. 3) are attributable to ecosystem characteristics. Specifically, probable release from stored organic matter [Eqn (7)] depends upon biomass storage and is larger for high-biomass temperate forests than for tropical or boreal forests (Fig. 2a). Additionally, contributions from disturbance depend strongly upon the difference between rates of carbon uptake in native and re-growing forests. Because forests re-grow most rapidly in the tropics (Anderson *et al*., 2006), there is a large increase in carbon uptake by recovering tropical forests (F^D), thereby lessening the negative impacts of disturbance. It is for these reasons that $GHGV_{100}^{50}$ decreases more rapidly with increasing fire frequency in temperate forests than in tropical or boreal forests.

Effects of time

To illustrate the effects of time on *GHGV*, we consider how *GHGV* changes with the emissions time span (τ_E), the analytical time span (τ_A), and the annual discount rate (r) for several ecosystem types that represent a variety of storage-flux combinations (Fig. 4). Specifically, we consider a tropical peat forest (extremely high organic matter storage released over a long time, large CO_2 sink, large CH_4 source), a tropical forest (high organic matter storage, moderate annual flux), an

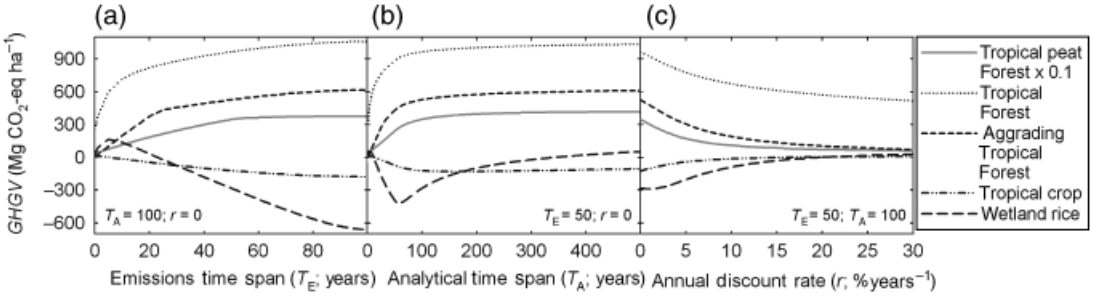

Fig. 4 *GHGV* for five tropical ecosystem types as a function of (a) emissions time span (τ_E), (b) analytical time span (τ_A), and (c) annual discount rate, *r*, in the weighting function $w(t_A) = (1/(1+r))^{t_A}$. For display purposes, values for tropical peat forest are multiplied by 0.1.

aggrading tropical forest (minimal organic matter storage, strong CO_2 sink), a tropical cropland (minimal organic matter storage, large N_2O source), and wetland rice (some organic matter storage, extremely high F_{CH_4}).

Choice of τ_E has the strongest effect on ecosystems with high GHG flux or slow release of stored organic matter (Fig. 4a). The *GHGV* of an aggrading tropical forest becomes more stable around 25 years when their rate of carbon accumulation declines, and that of a tropical peat forest plateaus around 50 years after most of the carbon from peat has been released. The *GHGV* of tropical forests increases continuously with τ_E, but is most sensitive to τ_E at lower τ_E's because of strong contributions of GHG release from stored organic matter. Tropical crops and wetland rice decline continually with τ_E (after an early increase for rice arising from *S*). Thus, *GHGV* is particularly sensitive to τ_E at short time spans and when a large portion of the ecosystems value comes from GHG exchange later in the emissions time frame.

Choice of τ_A has the strongest affect on ecosystems with high GHG exchange later during the time span of interest (Fig. 4b), as the effects of GHG's released later during the emissions time span take longer to stabilize relative to an initial pulse of CO_2. Ecosystems with high methane flux (F_{CH_4}; e.g., wetland rice) are especially sensitive to τ_A, as the global warming potential of CH_4 relative to CO_2 decreases drastically with τ_A because of CH_4's short atmospheric lifetime (Forster *et al.*, 2007). For most ecosystems, however, *GHGV* is not extremely sensitive to τ_A at analytical time spans greater than 100 years (when $\tau_E \leq 50$).

As with τ_E and τ_A, annual discounting has the strongest effect on ecosystems with high GHG flux or slow release of stored organic material (Fig. 4c). The importance of GHG exchange later in the emissions time period declines with increasing annual discount rate such that – at extremely high discount rates – *GHGV* approaches the value of initial GHG releases from land-clearing (i.e., $\tau_E = 0$).

Thus, decisions regarding the treatment of time are moderately influential for ecosystems whose *GHGV*'s are attributed mainly to stored organic material (i.e., most native ecosystems with unsaturated soils; Fig. 2) but are extremely influential for ecosystems with high annual GHG flux or slow release of stored organic material (i.e., wetlands, aggrading ecosystems, managed ecosystems). In cases that involve the latter, careful justification of the choice of the emissions time span (τ_E), the analytical time span (τ_A), and the choice of a discounting function [$w(\tau_A)$] are particularly important.

Calculating the full GHG effects of land-use change

The full GHG effect of land-use or land-cover change (GHG_{LUC}; Mg CO_2-eq ha^{-1}) can be readily calculated as the difference in the *GHGV*'s of the new and old ecosystems:

$$GHG_{LUC} = GHGV_{new} - GHGV_{old} \quad (9)$$

For example, the 50-year cost (with $\tau_A = 100$) of burning and clearing a tropical forest with minimal probability of disturbance ($GHGV_{100}^{50} = 967$ Mg CO_2-eq ha^{-1}) to make way for cropland ($GHGV_{100}^{50} = -121$ Mg CO_2-eq ha^{-1}) would be 1088 Mg CO_2-eq ha^{-1}, with costs arising both from the lost GHG benefits of maintaining the forest (e.g., C storage) and from the GHG costs of maintaining cropland (e.g., N_2O emissions from fertilizer). Conversely, abandoning a tropical cropland such that it becomes an aggrading tropical forest ($GHGV_{100}^{50} = 529$ Mg CO_2-eq ha^{-1}) would result in a GHG benefit of 650 Mg CO_2-eq ha^{-1}, with benefits arising both from the cessation of cropping practices and from the CO_2 sequestration associated with the growing forest.

This approach to calculating GHG_{LUC} captures the full dynamics of land-use or land-cover change (given adequate data) and involves proper weighting for the timing of emissions. When land use change consists of

complete clearing followed by tillage, GHG_{LUC} can be calculated straightforwardly according to Eqn (9). However, for LULCC that does not involve complete clearing followed by tillage (e.g., clearing forest for pasture, conversion of a native forest to a forest plantation, climate change driven land-cover change), it is necessary to adjust the calculation of *GHGV* of the old ecosystem for the fact that organic matter storage will never be reduced to the baseline level. Specifically, any organic matter pools that will not be disturbed should not be included in the calculations, and decomposition rates should be adjusted to reflect the fact that tillage is not occurring.

Legacy effects from the cleared ecosystem are included in its *GHGV*, and are combined with GHG fluxes from the new ecosystem in a time-specific manner. For example, draining and clearing of peatland to make way for cropland is characterized by an initial release of GHG's from vegetation followed by decades of enhanced CO_2 and N_2O release from the organic soil that remains (Kasimir-Klemedtsson *et al*., 1997), all of which are included in the peatland's *GHGV* (see Supporting Information for details). Thus, the *GHGV* approach to quantifying the GHG effects of LULCC allows accurate quantification of the dynamics of LULCC.

Conclusions

Our definition of the GHG value of ecosystems allows straightforward computation of the GHG effect of maintaining ecosystems over a multi-year time frame, expressing their value in terms of storage of organic material that would be released as GHG's upon land clearing [*S*; Eqn (5); Fig. 2a], annual uptake or release of GHG's (*F*; Fig. 2b), and probable contributions from natural disturbance [Eqns (6)–(8); Fig. 3]. *GHGV* treats all of these ecosystem-atmosphere GHG exchanges in a time-sensitive manner, thereby providing an appropriate framework for computation of the GHG consequences of any land use decision. It is expressed in units that make it amenable to a wide range of potential applications, and could readily be used to place monetary value on ecosystems simply by multiplying by the market price of carbon ($\$\,Mg^{-1}$ CO_2-eq).

GHGV improves upon existing metrics for placing a value on the climate services of ecosystems (Table 1) in that it (1) considers contributions from both flux and storage, (2) uses proper accounting for the timing of emissions over a multiple-year time frame, and (3) allows for the quantification of probable GHG exchange arising from disturbance. Comparison of *GHGV* with other metrics reveals that properly accounting for storage, flux, disturbance, and the timing of emissions can alter the perceived value of an ecosystem substantially

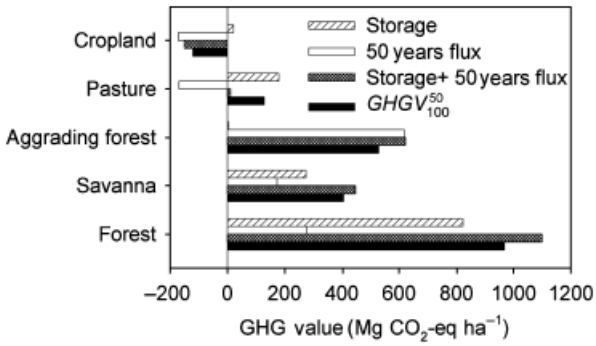

Fig. 5 Comparison of *GHGV* to three commonly used metrics of the GHG contributions of ecosystems (Table 1) for five tropical ecosystem types. Specifically, $GHGV_{100}^{50}$ is compared with (1) 'storage' – the GHG release that would occur upon clearing of the ecosystem, (2) '50 years flux' – the annual flux multiplied by a 50-year time span of interest, and (3) 'Storage + 50 years flux' – the summation of these two. For these three, CO_2-equivalents are calculated based on their global warming potential for a 100-year time horizon (Forster *et al*., 2007), which corresponds to the $\tau_A = 100$ used in computing $GHGV_{100}^{50}$).

(Fig. 5). Valuation of storage alone underestimates the value of ecosystems that also provide climate services through carbon sequestration (e.g., tropical forest, aggrading tropical forest), but overestimates the value of ecosystems that are GHG sources or would release GHG's over long time periods upon clearing (e.g., tropical cropland, wetland rice, tropical peat forest). For example, quantifying only the carbon that would be released from clearing a forest ($724\,Mg\,CO_2$-eq ha^{-1}) to make way for cropland neglects the costs of displaced carbon sequestration by the forest ($252\,Mg\,CO_2$-eq ha^{-1} $50\,yr^{-1}$) and the GHG costs of the cropland ($121\,Mg\,CO_2$-eq ha^{-1} $50\,yr^{-1}$), thereby capturing only 66% of the effect of the land use change. More dramatically, assuming that abandoned land that would succeed to forest is of little value because of its low organic matter storage ($\sim 4\,Mg\,CO_2$-eq ha^{-1}) neglects the high value arising mainly from its potential for a carbon uptake ($529\,Mg\,CO_2$-eq ha^{-1} $50\,yr^{-1}$ for tropical aggrading forests; Fig. 5; see also Righelato & Spracklen, 2007).

Valuation of flux alone neglects the value of stored organic matter and is often done in a time-inappropriate manner. While applications counting the cost of land-use/land-cover change typically do not neglect the value of organic matter in high-biomass ecosystems such as forests, they tend to give inadequate attention to the value of organic matter storage in relatively low-biomass ecosystems (e.g., California EPA, 2009; US EPA, 2009). Conversion of moderately grazed tropical pasture to cropland, for instance, would result in non-negligible GHG emissions from stored organic

matter (177 Mg CO_2-eq ha^{-1}), which is more influential to its *GHGV* than is 50 years of annual GHG exchange (−48 Mg CO_2-eq ha^{-1} 50 yr^{-1}; Fig. 5). Moreover, counting fluxes in a time-insensitive manner overestimates their impact (Fig. 5). For instance, simply summing the carbon that an aggrading successional forest would sequester over time span of interest (~618 Mg CO_2-eq ha^{-1} 50 yr^{-1}) overestimates the *GHGV* of the ecosystem (529 Mg CO_2-eq ha^{-1} 50 yr^{-1}). Arguments that allowing abandoned cropland to regrow to forest or grassland would outweigh the benefits of using the land for biofuels (Righelato & Spracklen, 2007; Piñeiro *et al*., 2009) overestimate the GHG benefits of allowing cropland to revert to natural land. This does not, however, imply that their arguments are fundamentally wrong, as they likewise overestimate the benefits of biofuels.

Evaluation of the combined effects of storage and flux over a multi-year time frame miscalculates the value of ecosystems when the timing of emissions is not treated properly (Fig. 5). This arises largely from overestimation of the flux term. For example, a tropical forest would be valued at 1100 Mg CO_2-eq ha^{-1} 50 yr^{-1} through simple summation of storage and 50 years flux, whereas its $GHGV_{100}^{50}$ is only 967 Mg CO_2-eq ha^{-1}. In contrast, if the ecosystem is a GHG source, its value is underestimated through improper treatment of time (e.g., cropland, pasture; Fig. 5). Applications that use this approach – including many biofuel life cycle analyses – effectively place improper weight on flux relative to storage. Many biofuel life cycle analyses (e.g., Fargione *et al*., 2008; Gibbs *et al*., 2008; Searchinger *et al*., 2008; California EPA, 2009; Melillo *et al*., 2009; Piñeiro *et al*., 2009; US EPA, 2009) may underestimate the GHG costs of biofuels-related land use change relative to the benefits of biofuels production (O'Hare *et al*., 2009).

As a metric whose particular strength is calculating the full effects of maintaining an ecosystem over a multiple-year time period, *GHGV* has many potential applications, and – in many cases – provides a more complete conceptual framework for valuing the GHG services of ecosystems. From a scientific standpoint, *GHGV* may provide a convenient metric for quantifying the impact of land-use or land-cover changes on the atmosphere. *GHGV* also may prove useful in forecasting the impacts of land-use policies – such as those relating to biofuels – on future GHG emissions. While assessments of the potential for biofuel production to help mitigate GHG emissions are increasingly recognizing the most important elements of *GHGV* (e.g., Fargione *et al*., 2008; Searchinger *et al*., 2008; California EPA, 2009; Melillo *et al*., 2009; Smeets *et al*., 2009; US EPA, 2009), we are not aware of any that include all flux and storage components of *GHGV* for both old and new ecosystems involved in both direct and indirect land use changes. Moreover, improper accounting for the timing of emissions may skew their outcomes, as discussed above. The *GHGV* approach to calculating GHG_{LUC} (Eqn (9)) may provide a more comprehensive and transparent framework for such analyses. Finally, *GHGV* could also provide a framework for more accurate representation of ecological and atmospheric dynamics in national and international carbon-offsetting programs that reward avoided GHG emissions or GHG sequestration, which require robust treatment of GHG exchanges over time and the possibility of GHG release through disturbance (i.e., 'volatility'; e.g., Kim *et al*., 2008).

GHGV quantifies the contributions of ecosystems to climate change through ecosystem-atmosphere GHG exchange; however, this does not represent their entire contribution to the climate system, nor does it characterize the full suite of critical ecosystem services. Biogeophysical forcings (e.g., albedo, evapotranspiration, aerosols from biomass burning) affect regional climate and may overshadow the GHG contribution to climate forcing in some biomes (e.g., Betts, 2000; Field *et al*., 2007; Bonan, 2008). These biogeophysical forcings must be combined with *GHGV* to quantify the full climate services of ecosystems. Moreover, a suite of ecosystem services that do not directly affect climate – including regulation of water flow and quality, preservation of habitats and biodiversity, production of food, fuel, or fiber, and utilization by native and marginalized peoples (e.g., Foley *et al*., 2005) – all warrant consideration in the valuation of ecosystems.

Acknowledgements

This work was supported by the Energy Biosciences Institute. We thank J. Anderson, S. Davis, J. Endres, C. Smith, and M. Zeri for helpful discussion; an anonymous reviewer for helpful suggestions; and D. Baldocchi, M. Litvak, S. Luyssaert, and T. Meyers for providing data.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Matlab code for calculating GHGV.
Appendix S2. Microsoft excel spreadsheet to accompany Matlab code (GHGVcalculator.xls).
Appendix S3. Methods for illustrative GHGV calculations.
Table S1. Definitions of ecosystem types.
Table S2. Organic matter storage (Mg dry matter ha^{-1}) in aboveground biomass, root biomass, dead wood, organic layer of litter or peat, soil organic matter (SOM) in the top meter of mineral soil, and SOM vulnerable to loss upon cultivation. Reported are mean values, standard deviations (where applicable), number of sites contributing to the estimate, and references from which the values were obtained.
Table S3. Burn characteristics of combustible biomass for various ecosystem types. Listed are the fraction of combustible biomass consumed in a land clearing fire (f_{CB}^{c}; eq. 5), emission factors for CO_2, CH_4, and N_2O ($E_{x,CB}^{d}$; eq. 5), categories within original references, and references.
Table S4. Decomposition constants for each biomass pool (combustible material, roots, peat, and soil organic matter) across a variety of ecosystem types (eq. 5, 7; also see above).
Table S5. Carbon dioxide flux (F_{CO_2}; kmol CO_2 ha^{-1} yr^{-1}), or net ecosystem production, in various ecosystem types. Sign convention: negative indicates carbon uptake by ecosystems. Listed are F_{CO_2} estimates, standard deviations (where available), number of sites contributing to the estimate, measurement methodology, and references.
Table S6. Methane flux, F_{CH_4}, from soils and enteric fermentation of livestock. Methanogenesis and methanotrophy values (median, n, and category) are as reported in a review by Le Mer & Roger (2001). All units kmol CH_4 ha yr^{-1}.
Table S7. Nitrous oxide flux (F_{N_2O}; kmol ha^{-1} yr^{-1}) for various ecosystem types. Listed are means, standard deviations, number of sites contributing to the estimate, and notes.
Table S8. Anthropogenic CO_2 emissions associated with agriculture: emissions from fertilizer production, lime application, and fuel use of farm equipment. All are expressed in units of kmol CO_2 ha^{-1} yr^{-1}.
Table S9. Contributions of storage and flux components to $GHGV_{100}^{50}$ (50 year emissions time span, 100 years analytical time span) of various ecosystem types. All units Mg CO_2-eq/ha.

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