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Carbon outcomes of major land-cover transitions in SE Asia: great uncertainties and REDD+ policy implications

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Abstract

Policy makers across the tropics propose that carbon finance could provide incentives for forest frontier communities to transition away from swidden agriculture (slash-and-burn or shifting cultivation) to other systems that potentially reduce emissions and/or increase carbon sequestration. However, there is little certainty regarding the carbon outcomes of many key land-use transitions at the center of current policy debates. Our meta-analysis of over 250 studies reporting above- and below-ground carbon estimates for different land-use types indicates great uncertainty in the net total ecosystem carbon changes that can be expected from many transitions, including the replacement of various types of swidden agriculture with oil palm, rubber, or some other types of agroforestry systems. These transitions are underway throughout Southeast Asia, and are at the heart of REDD+ debates. Exceptions of unambiguous carbon outcomes are the abandonment of any type of agriculture to allow forest regeneration (a certain positive carbon outcome) and expansion of agriculture into mature forest (a certain negative carbon outcome). With respect to swiddening, our meta-analysis supports a reassessment of policies that encourage land-cover conversion away from these [especially long-fallow] systems to other more cash-crop-oriented systems producing ambiguous carbon stock changes – including oil palm and rubber. In some instances, lengthening fallow periods of an existing swidden system may produce substantial carbon benefits, as would conversion from intensely cultivated lands to high-biomass plantations and some other types of agroforestry. More field studies are needed to provide better data of above- and below-ground carbon stocks before informed recommendations or policy decisions can be made regarding which land-use regimes optimize or increase carbon sequestration. As some transitions may negatively impact other ecosystem services, food security, and local livelihoods, the entire carbon and noncarbon benefit stream should also be taken into account before prescribing transitions with ambiguous carbon benefits.

Keywords: agroforestry, climate change, land-cover change, oil palm, REDD+, rubber plantations, shifting cultivation, slash-and-burn, swidden, tropical deforestation

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Introduction

Parties to the United Nations Framework Convention on Climate Change (UNFCCC) have agreed that efforts to Reduce Emissions from Deforestation and forest

Degradation (REDD+) will play a focal role in future climate change mitigation efforts (UNFCCC, 2010, 2011). Under the proposed UNFCCC REDD+ framework, forest-rich, developing nations would be paid by industrialized nations, if they achieve long-term reductions in carbon emissions by reducing deforestation and forest degradation, protecting and enhancing carbon stocks, and replacing unsustainable forest practices with ones that sequester more carbon (UNFCCC, 2010, 2011). This is expected to produce cost-effective, politically attractive means of reducing greenhouse gas emissions, meanwhile producing cobenefits, such as biodiversity conservation, maintenance of ecosystem services, and sustainable rural development (cf. Gibbs *et al.*, 2007; Mertz, 2009; van Noordwijk *et al.*, 2009; Phelps *et al.*, 2012). However, UNFCCC decisions have yet to specify exactly what land-use reforms and activities will be promoted and rewarded under a future REDD+ mechanism. Given uncertainties about carbon stocks and fluxes under slash-and-burn agriculture and potential alternative land uses, it remains unclear how the mechanism will influence the livelihoods and agricultural practices of rural and forest-dependent communities across the tropics. Herein, we focus on slash-and-burn cultivation, which is more appropriately termed swiddening or shifting cultivation – because it is a land use heavily targeted for transformation under a number of REDD+ policies. Moreover, swiddening has long been criminalized and misunderstood across much of its range (Padoch & Pinedo-Vasquez, 2010; Ziegler *et al.*, 2011). Nevertheless, our analysis provides insight for all major land-cover transitions relevant to REDD+.

Swiddening is a longstanding, widespread, and diverse category of land use (Fox *et al.*, 2009). It is either the main source of livelihood or an important source of supplementary income for millions of people worldwide (Cramb *et al.*, 2009; Mertz *et al.*, 2009). Swiddening typically involves clearing plots of woody vegetation with the aid of fire, then cultivating for a few (<3–10) years before fallowing (cf. Mertz *et al.*, 2009). Although a wide range of land-use systems and management practices fall within this description, a fundamental division exists between partial and integral swidden systems. In partial systems, incipient swiddens involve farmers with little prior knowledge of swidden techniques who devote efforts in clearing and burning swidden fields for other permanent forms of agriculture next to homesteads. In integral systems, pioneer and established, rotational swiddens are predominant (Conklin, 1957). Pioneer swiddens involve farmers customarily clearing portions of primary forest and then cultivating for a few cycles before new plots are established elsewhere, often in another landscape or watershed. Established, rotational systems involve moving from plot to plot within the same landscape, with relatively little primary forest affected (Conklin, 1957).

Both systems are interconnected, however, with the pioneer strategy being used to open up new primary forest areas for rotational swidden, which occurs when populations migrate and expand into previously uncultivated areas; it also occurs without population expansion, in response to market incentives and globalized economies (e.g. Vadya, 1961; Inoue & Lahjie, 1990; Mertz *et al.*, 2009). Rotational swidden systems involve the farmer returning to formerly cultivated plots after short (<5 years), intermediate (5–10 years), or long (10–25+ years) fallow periods and is often part of customary practice (e.g., religion, ritual, and sacrifice). Depending on land-use history, length of fallow, and the degree of disturbance during the cultivation phase, successive regrowth in swidden systems includes vegetation associations ranging from poor-quality grasslands to mature secondary forests high in biomass and species diversity, albeit of various degrees of degradation (cf. Lawrence, 2004, 2005; Cairns, 2007; Bruun *et al.*, 2009; Messerli *et al.*, 2009; Rerkasem *et al.*, 2009; Ziegler *et al.*, 2009b).

Swidden systems vary dramatically in their management of biophysical constraints on plant growth, and thus their impacts on forests and carbon cycling also vary widely. However, these systems are often amalgamated into a single category that is labeled a leading agent of forest degradation, deforestation, and carbon emissions (Dove, 1983; Geist & Lambin, 2002; Mertz, 2009; FCPF, 2010, 2011; Ziegler *et al.*, 2011). Throughout much of South and Southeast Asia, swidden agriculture has largely been replaced by other forms of agriculture (Rasul & Thapa, 2003; Padoch *et al.*, 2007; Schmidt-Vogt *et al.*, 2009). These transitions have often been motivated by government policies restricting swiddening as well as economic factors that promote commercial agriculture (Cramb *et al.*, 2009; Fox *et al.*, 2009; Van Vliet *et al.*, 2012). Nonetheless, REDD+ policymakers across the tropics are proposing that REDD+ carbon finance be used to provide further economic incentives for even more rural communities to transition away from swidden agriculture to other land uses, anticipating the changes will increase carbon sequestration and reduce pressures on existing forests (Indonesia UKP-PPP, 2010; UNREDD, 2010; FCPF, 2011).

Crucially, too little is known about differences in carbon cycling within various types of swidden and replacement agricultural systems to provide convincing evidence as to which land-cover/land-use (LCLU) types would provide the most viable basis for emissions mitigation approaches (Phelps *et al.*, 2010a). Although swidden agricultural techniques result in conspicuous point sources of CO_2 emissions during periodic burning, these carbon losses are offset to varying degrees by sequestration during the fallow phase. At the very least, one must demonstrate, which transitions will result in long-term, verifiable gains in sequestered carbon (cf. ASB, 2011). Herein, we address this issue by analyzing available estimates of above-ground carbon (AGC), below-ground carbon (in root biomass, BGC), and soil organic carbon (SOC) for swidden and major replacement land covers in Southeast Asia.

Methods

We identify several common land-cover/land-use transitions that involve swidden agriculture and are highly relevant to REDD+ and carbon-focused forest management in Southeast Asia: (a) permanent abandonment of a swidden site to allow regeneration of forests; (b) continuation of status-quo rotational swidden systems; (c) replacement of swidden by orchards or monoculture tree plantations, including rubber, oil palm, and timber species; (d) replacement of swidden by non-sequential agroforestry systems (e.g., home gardens, intercropping strategies); (e) intensification of swidden characterized by a lengthened cropping period and a shortened fallow period; (f) replacement of swidden by grassland, pasture, or shrublands; (g) replacement of swidden by permanent cropping of continuous annual field crops and nontree monocultures (e.g., commercial crops); and (h) agricultural expansion into primary forest. We further identify the possibility of (i) extending the swidden fallow periods of existing short- and intermediate-fallow systems, and (j) logging.

For each transition, we assign a range of above- and below-ground carbon stock values from more than 250 published case studies and relevant reviews for the SE Asia region. When necessary, we estimate carbon biomass as one-half the reported vegetative biomass value (cf. Smith *et al.*, 2010). For each prospective transition, we identify a plausible range for total ecosystem carbon (TEC = AGC + BGC + SOC). We classify swidden as short (fallows <5 years), intermediate (5–10 year fallows), and long-fallow agriculture systems (fallows >10 years). The latter category includes pioneer swiddening. Owing to limited data, we do not distinguish among various subtypes of each land cover (e.g., dry vs. moist forest types), but we do separate swiddening from other types of agroforestry. The associated land covers for the transitions mentioned above are the following: forest (F); logged over forest (LOF), orchards and tree-plantations (OTP), long-fallow swidden (LFS); rubber plantations (RP), agroforest (AGF), grassland, pasture, or shrublands (GPS), intermediate-fallow swidden (IFS), oil palm plantations (OP), short-fallow swidden (SFS), and permanent cropland (PC).

Most studies that provide AGC estimates or forest biomass data were conducted in Indonesia and Malaysia (Table S1). A smaller set of reports provide data for Thailand, the Philippines, and southern China (Xishuangbanna and Hainan Island). Data for Brunei, Cambodia, Laos, Myanmar, Papua New Guinea, and Vietnam are rarer still (Table S1). The data represent many forest types, ranging from dry lowland dipterocarp forests to moist mountain forests and wet rainforests, although in many cases, it is difficult to know exactly what species comprise the associations due to the lack of standardization of forest nomenclature (cf. Maxwell, 2004). Fewer data are available for the nine other general land-cover types. From the range of values reported, we exclude outliers to produce minimums and maximums that define the range of plausible values (Fig. 1). This modified range is used for comparing AGC associated with the 11 land covers considered (Fig. 2). In defining the ranges, we excluded many low values that were associated with arguably extreme conditions (e.g., high altitudes for rubber; degraded forest) or young ages (e.g., immature forest stands); similar caution was used in defining the high end (e.g., savanna for the GPS category).

Much less information is available for below-ground carbon biomass (BGC; summary data not shown), largely because of the difficulty in quantifying below-ground phenomena. Thus, in the analysis, we rely on biomass partitioning factors (BPF = BGB/AGC) based on reported root:shoot ratios to estimate plausible BGC values (Table 1). The plausible BPFs for forest (0.10–0.28) are based on several studies conducted throughout SE Asia (Table 1). We also use this same range for logged over forests. The BPF range assigned to agroforestry is slightly larger at 0.10–0.34, for which the maximum is associated with home gardens in Java (Jensen, 1993). Orchards and other tree plantations are assigned a range (0.10–0.33) consistent with data from several SE Asia countries. The range for oil palm (0.15–0.40) and intermediate- and long-fallow swidden (0.12–0.25) are based on limited data from Indonesia and Malaysia. We use the data from China and Cambodia to assign rubber BPFs that range from 0.10 to 0.26. Limited data suggests a very large range (0.30–1.92) for various types of grassland, pasture, and shrublands. The BPFs assigned to short-fallow swidden and permanent agriculture (0.05–0.20) are the lowest of all land covers.

Soil organic carbon (SOC) comprises a percentage of the total ecosystem carbon that is difficult to quantify across most landscapes (cf. Dixon *et al.*, 1994; Don *et al.*, 2011). Direct comparison of SOC values among several sites is problematic because of lack of standardization in determination methods and disregard for depth-specific bulk density (Guo & Gifford, 2002; Don *et al.*, 2011). In addition to initial carbon content, many other factors affect SOC at any location at any given time, including climate, topography, soil type, microbial communities, nitrogen cycling processes, management, and prior land use (Murty *et al.*, 2002; Bruun *et al.*, 2006). Thus, we rely on previous syntheses (Guo & Gifford, 2002; Don *et al.*, 2011) to assign a range of plausible SOC values for each land-cover in this analysis, as follows. We begin by assuming forest is the ideal situation (Table 2). A transition from forest to croplands reduces SOC by 25–30%; and a transition to grasslands brings about a 12% reduction (Don *et al.*, 2011). Furthermore, secondary forests contain 9% less SOC than primary forests (Don

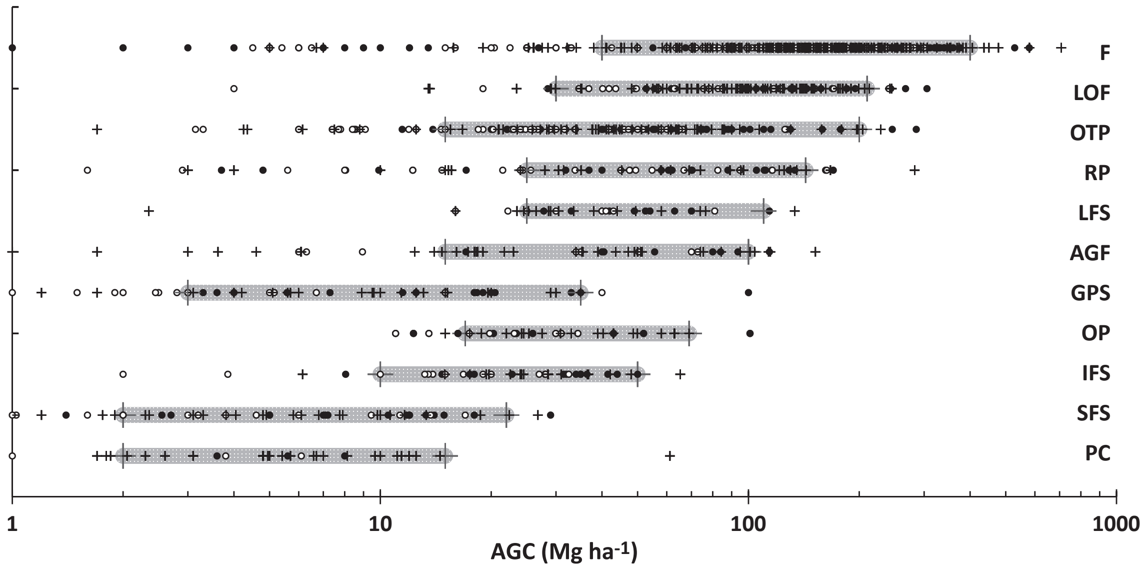

Fig. 1 Summary of above-ground carbon biomass (AGC) values (from Table S1). Open and closed circles are minimum and maximum values of reported ranges, respectively. Crosses refer to reported mean values. The thick line (ending in a cross) is the modified range, defined by the minimum and maximum values reported in Table 3 (also shown in Fig. 2b). Land covers are the following: forest (F); logged over forest (LOF); orchards and tree plantations (OTP); rubber plantations (RP); long-fallow swidden (LFS); nonswidden agroforest (AGF); grassland, pasture, or shrublands (GPS); oil palm plantations (OP); intermediate-fallow swidden (IFS); short-fallow swidden (SFS); and permanent cropland (PC).

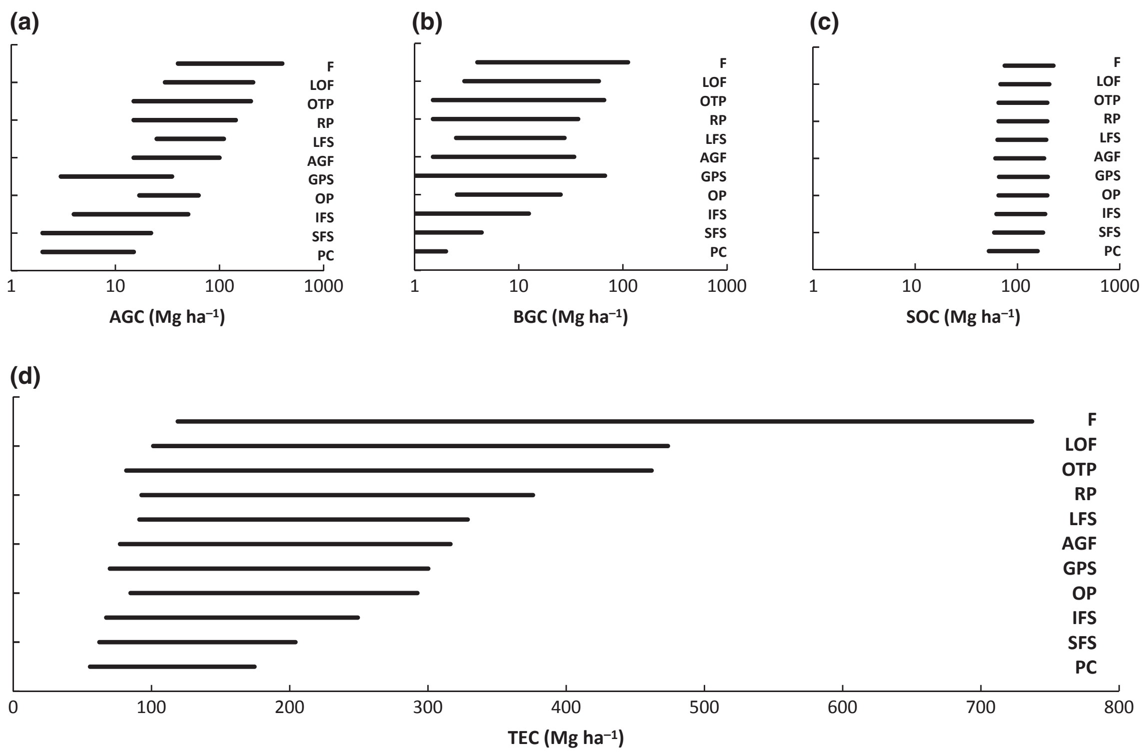

Fig. 2 For the 11 land covers considered in this analysis, plausible ranges of the following: (a) above-ground carbon biomass (AGC); (b) below-ground carbon biomass in vegetation (BGC); (c) soil organic carbon (SOC); and (d) total ecosystem carbon (TEC = AGC + BGC + SOC). Land covers are the following: forest (F); logged over forest (LOF); orchards and tree plantations (OTP); rubber plantations (RP); long-fallow swidden (LFS); nonswidden agroforest (AGF); grassland, pasture, or shrublands (GPS); oil palm plantations (OP); intermediate-fallow swidden (IFS); short-fallow swidden (SFS); and permanent cropland (PC).

Table 1 Biomass partitioning factors (BGB/AGB) used to calculate below-ground carbon biomass values from the estimated range of above-ground carbon biomass values

Land cover*	Min†	Max†	Note
F	0.10	0.28	The range brackets values reported for Brunei (Brown *et al*., 1993), Cambodia (Hozumi *et al*., 1969; Kiyono *et al*., 2010), China (Feng *et al*., 1998; Zheng *et al*., 2000; Shanmughavel *et al*., 2001; Lü *et al*., 2006; Qi & Tang, 2008), Indonesia (Brown *et al*., 1993; Hergoualc'h & Verchot, 2011), Laos (Brown *et al*., 1993), Malaysia (Bandhu, 1973; Koopmans & Andriesse, 1982; Cairns *et al*., 1997; Pinard & Putz, 1996; Hikmat, 2005; Niiyama *et al*., 2010), Papua New Guinea (Edwards & Grubb, 1977); Philippines (Brown *et al*., 1993); Thailand (Ogawa *et al*., 1965; Brown *et al*., 1993; Terakunpisut *et al*., 2007; Pibumrung *et al*., 2008), and the tropics (Germer & Sauerborn, 2008)
LOF	0.10	0.28	Assumed to be the same as forest.
OTP	0.10	0.33	The range brackets values reported for Indonesia (Lasco, 2002; Miyakuni *et al*., 2004; Syahrinudin, 2005), Malaysia (Nykvist *et al*., 1996), Papua New Guinea (Yamada *et al*., 2000a); Thailand (Hiratsuka *et al*., 2005; Gnanavelrajah *et al*., 2008), and Vietnam (Zemek, 2009). Maximum value bound at 0.33.
RP	0.10	0.26	The range brackets values reported for Cambodia and China (Hainan and Xishuangbanna) by Cheng *et al*. (2007), Tang *et al*. (2009), and Mizoue *et al*. (2009).
LFS	0.12	0.25	Values bracket data for swiddening in general in Indonesia and Malaysia (Koopmans & Andriesse, 1982; Kiyono & Hastaniah, 2005; Kenzo *et al*., 2010).
AGF	0.10	0.34	The range brackets values reported for Indonesia (Jensen, 1993; Roshetko *et al*., 2002) and a minimum value of 0.10.
GPS	0.30	1.92	Values correspond with those for tropical/subtropical grassland reported by Mokany *et al*. (2006); the range also brackets that reported by Kamnalrut & Evenson (1992), Syahrinudin (2005), IPCC (2006), and Germer & Sauerborn (2008)
OP	0.15	0.40	Brackets data for Indonesia (Syahrinudin, 2005; Hergoualc'h & Verchot, 2011) and Malaysia (Henson & Chai, 1997; Khalid *et al*., 1999a,b; Henson & Dolmat, 2003).
IFS	0.12	0.25	Values bracket data for swiddening in general in Indonesia and Malaysia (Koopmans & Andriesse, 1982; Kiyono & Hastaniah, 2005; Kenzo *et al*., 2010).
SFS	0.05	0.20	Minimum and maximum values were set slightly lower than those for intermediate- and long-fallow swiddening.
PC	0.05	0.20	Assumed the same as short-fallow swiddening

*Land covers are the following: forest (F); logged over forest (LOF); orchards and tree plantations (OTP); rubber plantations (RP); long-fallow swidden (LFS); agroforest (AGF); grassland, pasture, or shrublands (GPS); oil palm plantations (OP); intermediate-fallow swidden (IFS); short-fallow swidden (SFS); and permanent cropland (PC).
†Min and max root/shoot values are multiplied by the AGC values in Table 3 to produce the BGC estimates.

Table 2 Conversion factors used to calculated differences in SOC among the 11 land-cover types considered in this analysis (based on Guo & Gifford, 2002; Don *et al.*, 2011)

Land cover*	Conversion factor†	Note/explanation
F	1	Equals nominal value of 150 Mg ha^{-1}
LOF	0.91	Conversion for logging
OTP	0.87	Forest conversion to plantation
RP	0.87	Forest conversion to plantation
LFS	0.85	Intermediate of plantations and IFS
AGF	0.81	Conversion factor that is intermediate to plantations and PC
GPS	0.88	Conversion from forest to grasslands or pasture
OP	0.87	Forest conversion factor from forest to plantations
IFS	0.83	Conversion factor that is intermediate of plantations and SFS
SFS	0.79	Conversion factor that is intermediate of GPS and permanent croplands
PC	0.70	Conversion from forest to permanent cropland (low end)

*Land covers are the following: forest (F); logged over forest (LOF); orchards and tree plantations (OTP); rubber plantations (RP); long-fallow swidden (LFS); agroforest (AGF); grassland, pasture, or shrublands (GPS); oil palm plantations (OP); intermediate-fallow swidden (IFS); short-fallow swidden (SFS); and permanent cropland (PC).

†Conversion factors are multiplied by a range of plausible SOC values to produce the minimum and maximum values shown for SOC in Table 3.

et al., 2011). Afforestation of cropland increases SOC 29%; and fallowing and conversion of cropland to grasslands increases SOC 32% and 26%, respectively (Don *et al.*, 2011). Transitions from forest to various types of plantations decrease SOC on average by 13% (Guo & Gifford, 2002).

From these relationships, we estimated SOC values for different land-cover types using an idealized forest soil profile containing 150 Mg ha^{-1} SOC as the reference. The minimum SOC was estimated to be half (75 Mg ha^{-1}) this value; and the maximum is allowed to be 50% higher (225 Mg ha^{-1}). Thus, for example, a change from forest to grassland would yield an estimated SOC range of 66–198 Mg ha^{-1}. This is computed as 0.88×75 and 0.88×225 Mg ha^{-1}, where 0.88 represents a general mean decrease in SOC of 12% associated with a forest-to-grassland transition (Table 2).

Results

Carbon estimates

Forest AGC biomass ranges from 40 to 400 Mg ha^{-1} (Table 3). Many of the highest values are for rainforests and primary forests (e.g., in Indonesia; Table S1). Low values tend to be dry forests or those that are disturbed or potentially stressed by geographical setting (e.g., high altitudes in southern China). Although the range of AGC for most types of forests is large throughout the region, the center value of the range, 220 Mg ha^{-1}, is realistic value for forest in SE Asia (cf. Gibbs *et al.*, 2007). As expected, the AGC range of logged-over forests shifts downward to 30–210 Mg ha^{-1} (Table 3). Rubber plantations (25–143 Mg ha^{-1}), orchards, and other types of tree plantations (15–200 Mg ha^{-1}), oil palm plantations (17–69 Mg ha^{-1}), nonswidden agroforestry (15–100 Mg ha^{-1}), grasslands, pastures and shrubs (3–35 Mg ha^{-1}), and permanent croplands (2–15 Mg ha^{-1}) contain substantially lower AGC biomass than forests. AGC biomass range of short-fallow swidden (2–22 Mg ha^{-1}) was virtually indistinguishable from permanent croplands (Fig. 1). The range of AGC for long-fallow swidden systems (25–110 Mg ha^{-1}) is not greatly different from that of rubber; and the AGC biomass range of intermediate-fallow systems (4–50 Mg ha^{-1}) is most similar to oil palm (Fig. 1).

Patterns of estimated BGC values for the 11 types of land cover largely follow those for AGC, with forests having the highest range (4–112 Mg ha^{-1}); and permanent crops and short-fallow swidden the lowest (1–2 and 1–4 Mg ha^{-1}, respectively). Below-ground carbon in logged-over forests, orchards, and other tree-based plantations ranges from about 2 to 59–66 Mg ha^{-1}. Notably, the maximum oil palm BGC value (28 Mg ha^{-1}) is lower than most other tree-based land covers except long-fallow swidden (28 Mg ha^{-1}) and rubber (37 Mg ha^{-1}). The estimated range of BGC for rubber plantations is 3–37 Mg ha^{-1}; and various types of nonswidden agroforestry have a range of 2–34 Mg ha^{-1} (Table 3).

Estimated SOC for the 11 land-cover transitions are not highly variable, in part because of our generalized way in making categorical calculations from meta-analysis data (Table 3). In general, great uncertainty in SOC changes following conversion is important because the SOC fraction could be large, particularly for deep soils high in organic material. The greatest losses of SOC are expected to occur shortly after the initial forest conversion, and then approach equilib-

Table 3 Range of above-ground carbon biomass (AGCB), below-ground carbon biomass (BGCB), soil organic carbon (SOC), and total ecosystem carbon (TEC) for ten land covers

	AGC		BGC		SOC		TEC	
Land cover*	Min	Max	Min	Max	Min	Max	Min	Max
Forest	40	400	4	112	75	225	119	737
Logged-over forest	30	210	3	59	68	205	101	474
Orchards & tree plantations	15	200	2	66	65	196	82	462
Rubber plantations	25	143	3	37	65	196	93	376
Long-fallow swiddening	25	110	3	28	64	191	91	329
Agroforestry	15	100	2	34	61	182	77	316
Grasslands/pastures/shrublands	3	35	1	67	66	198	70	300
Oil palm	17	69	3	28	65	196	85	292
Intermediate-fallow swiddening	4	50	1	13	62	187	67	249
Short-fallow swiddening	2	22	1	4	59	178	62	204
Permanent croplands	2	15	1	2	53	158	56	175

*Land covers are ranked by maximum TEC (AGC + BGC + SOC components). All values are in units of Mg ha^{-1}. Minimum BGC values were rounded to 1; permanent croplands BGC max was set to 2.

rium – this is likely true for swidden fields and plantations alike (cf. Murty *et al.*, 2002; Bruun *et al.*, 2009). However, immediate, long-lasting reductions in SOC should also result from soil excavations, such as from terracing to allow planting of high-value tree crops on steep slopes (Bruun *et al.*, 2009). Regeneration of SOC is especially relevant to fallowing in swidden systems. Unless sites are severely degraded, lengthy fallowing should increase SOC on the order of 25% (assuming succession leads to grasslands or secondary forests). SOC is also likely to increase in converted plantations as tree stands mature, although increases may be curbed by understory management approaches that remove vegetation and fine/woody organic debris. Herein, we also recognize that carbon stocks after an initial transition may not be recognizable over the course of one rotation of swidden fallow, oil palm, or rubber.

Total ecosystem carbon stock differences

Based on the data and assumptions outlined above, the highest range of TEC values is for forests (119–737 Mg ha^{-1}; Fig. 2; Table 3). The forest TEC range was distinguishable from all other ecosystems by its upper end, which was 55% greater than logged forest (101–474 Mg ha^{-1}) and more than twice that of most other ecosystems (Table 3). In contrast, there was much less variability in the minimum TEC values across ecosystem types (Fig. 2). Orchards/tree plantations TEC range (82–462 Mg ha^{-1}) is similar to logged forests, and rubber plantations are similar to long-fallow swiddening (93–376 Mg ha^{-1} vs. 91–329 Mg ha^{-1}, respectively, Table 3). The range for agroforestry TEC is slightly lower at 77–316 Mg ha^{-1}. Total ecosystem carbon values for grassland/pastures/shrublands (70–300 Mg ha^{-1}) and oil palm plantations (85–292 Mg ha^{-1}) are similar (Table 3). It is important to note that TEC values for grasslands, pastures, and shrublands are elevated because root:shoot ratios associated with some ecosystems are very high compared with other land covers (Table 1). The main difference between short- and intermediate-fallow swidden systems is in the upper range, 204 Mg ha^{-1} vs. 249 Mg ha^{-1}, respectively; the low values are similarly at 62 and 67 Mg ha^{-1}, respectively (Table 3). Finally, permanent croplands have TEC ranges slightly below that of short-fallow swidden (56–175 Mg ha^{-1}).

Based on these data, the following positive and negative outcomes in total ecosystem carbon related to current land-cover/land-use transitions in SE Asia are noteworthy:

1 Certain positive carbon outcomes

- Abandonment of any agricultural system to allow permanent forest regeneration.
- Regeneration of logged forest into high-biomass permanent forest.
- Conversion of permanent croplands and short-fallow swidden systems to other land-uses, including tree-based plantations, orchards, various agroforests, and intermediate- to long-fallow swidden systems.
- Transition from intermediate-fallow swidden to long-fallow swidden systems, other agroforestry systems, rubber, and other tree plantations.

- Regeneration of grasslands – particularly if degraded – or replacement of pastures with orchards, rubber, and timber plantations.
- Conversion from oil palm plantations to rubber plantations, orchards, or other tree-based plantations.

2 Certain negative carbon outcomes

- Logging of high-biomass primary forest.
- Conversion of primary or other high-biomass forest into any type of agriculture or plantation.
- Conversion of any land-cover type, except short-fallow swiddening, to permanent croplands.
- Intensification of any type of swidden agriculture via shortening of fallow period and/or increasing the length of the cropping period. Data presented elsewhere also suggests that the continuation of status-quo rotational swidden systems results in a long-term negative carbon outcome (Eaton & Lawrence, 2009).
- Replacement of long-fallow swiddening by permanent croplands – this may also apply to some intermediate-fallow swiddens.
- Conversion of rubber to oil palm, permanent croplands, or short and intermediate-fallow swidden systems – probably also grasslands.

Based on our analysis, many other land-cover/land-use transitions would produce uncertain or potentially neutral carbon outcomes: e.g. (a) transitions between short-fallow swidden systems and permanent croplands; (b) land-cover changes between/among long-fallow swidden, other agroforestry systems, and possibly rubber; and (c) land-cover changes between/among intermediate-fallow swiddening, grasslands, pastures, shrub lands, and oil palm plantations. This uncertainty is important to stress because many of these transitions are currently at the heart of REDD+ debates (see 'Policy Implications').

Discussion

Caveats

Although the scale of uncertainty as highlighted by this analysis is clear, it remains important to articulate its limitations. First, long-term carbon benefits of any transition depend greatly on the fate of the above-ground vegetation at the end of a land-use rotation. For example, the decision on whether to protect/restore logged-over, previously high-biomass forests (with the explicit intention of increasing biomass) or allow their conversion to another nonforest land cover could lead to significantly different carbon outcomes. Another example, although not currently practised on a wide scale, processing over-mature rubber trees into various wood products would increase the lifetime of carbon gains. In addition, existing data on BGC are only sufficient to conclude that replacement systems that do not increase or maintain root biomass probably store less BGC than most swidden systems.

Data gaps, variable data quality, and lack of methodological standardization created several problems. Notably, most case studies only examined a single land use, and did not determine carbon stock changes over a long period of time, or measure forest biomass before the disturbance/transition. Above-ground C stocks in land-use systems with rotation times of several years (e.g. swidden systems and plantations) should ideally have been calculated as time-averaged values to allow for a comparison of systems with different rotation times. However, due to the limited availability of studies that report the time-averaged above-ground C storage in the land-use systems in question (Bruun *et al.*, 2009), this was not possible.

Information on spatial and temporal heterogeneity was rarely available to allow scaling from sampled plots to the landscape scale. In some cases, plot sizes were arguably too small for estimating biomass with high accuracy (although we still included them); ideally, replicated plots should have areas of at least 2500 m^2 (Brown *et al.*, 1996). In some case studies, inappropriate allometric equations yielded under- or over-estimates of tree biomass. Some studies were also limited by inaccurate land-use change data, such as inaccurate interviewee accounts of land-use history.

Lack of methodological standardization undermines accurate assessments of differences in soil organic carbon among land-cover types. Ideally one would want to determine these differences for profiles of a common depth (e.g., $\geq$1–2 m), accounting for differences in important physical properties, such as bulk density. This was not possible, thus we relied on an SOC conversion factor, relative to an idealized forest (Table 2) based on the most comprehensive meta-analyses to date (Guo & Gifford, 2002; Don *et al.*, 2011). Although having ecosystem-specific conversion values would have been optimal, our solution was the most appropriate given the absence of such data.

Categorizing, each case study was not always straightforward because most provided too little information to ensure unequivocal classifications, and because land-cover is influenced both by land-use history and current management. For example, we occasionally classified some postdisturbance land covers as various stages of fallow rather than grasslands or logged-over forests; and in one other instance we classified a tea plantation within the OTP category, rather than permanent croplands. In making these decisions, we relied on our interpretation of contextual information provided in the case studies, combined with our own experience. However, the few cases of uncertain classifications did not bias the results greatly as they eliminated group outliers that would have been ignored when we selected the subset of values that formed the plausible range for each land cover.

Importantly, our analysis does not assess the relative biodiversity, financial or livelihood benefits associated with each LCLU transition, although these factor heavily in policy discussions (Phelps *et al.*, 2010b, 2012). The carbon-positive transitions that we identify are not equivalent in terms of noncarbon benefits. For example, maintaining land under swidden agriculture can deliver superior biodiversity benefits compared with monoculture plantations and some other agroforestry systems, but crops are likely more lucrative while also exposing landholders to market fluctuations (cf. Lawrence, 2004; Cramb *et al.*, 2009; Fox *et al.*, 2009; Rerkasem *et al.*, 2009; Padoch & Pinedo-Vasquez, 2010). Moreover, not all land-use transitions would necessarily be eligible for REDD+ finance; transitions that negatively affect biodiversity could be disqualified based on UNFCCC safeguards (e.g., some transitions to plantations; UNFCCC, 2011). Finally, these transitions assess the possible carbon outcomes in the context of single-site changes only. When considering the broader landscape, the possibility that one land-cover transition could shift pressures and influence the land-cover transitions – and therefore carbon outcomes – at other sites (so-called leakage) should not be overlooked (Miles & Kapos, 2008).

Policy implications

Our analysis of more than 250 studies reveals that aside from a few exceptions, it is virtually impossible, given the current state of knowledge, to make informed recommendations or policy decisions regarding how many of the land-use changes occurring in Southeast Asia would affect total ecosystem carbon stocks. Notably, there is little evidence to suggest that transitions from swidden agriculture to many other land uses will directly or reliably produce positive carbon gains, e.g., from intermediate- or long-fallow swidden systems to oil palm and rubber plantations. However, many proposals to reduce deforestation and forest degradation target rotational farmers for exactly these types of land-use transitions, placing swidden at the centre of global REDD+ climate change mitigation actions (UNREDD, 2010; FCPF, 2011). This meta-analysis thus supports a reassessment of policies that encourage land-cover conversion away from (especially long-fallow) swidden systems (cf. Ziegler *et al.*, 2011).

Southeast Asia hosts a number of early REDD+ type projects (as of January 2012): Indonesia (44 projects), Cambodia (four projects), Malaysia (one project), Vietnam (seven projects), Thailand (one project), Papua New Guinea (six projects), Philippines (four projects), and Laos (one project); and several countries in the region have also started national-level preparations to engage with a future REDD+ mechanism (CIFOR, 2011; FCPF, 2011). Although most countries have released only initial planning documents, replacement of swidden agriculture with other land uses is a common feature. For example, Indonesia's REDD+ strategy proposes agricultural intensification (permanent cropland) and planting of oil palm and trees for pulp and timber (plantations) as alternatives to unsustainable forest harvest and slash-and-burn agriculture (FCPF, 2011). Similarly, Cambodia's leading REDD+ pilot project at Oddar Meanchey promotes transitions from slash-and-burn farming to intensive, permanent agriculture, and farming of high-value crops, land-use transitions also evident in Vietnam's REDD+ Readiness Proposal (FARGC, 2009; FCPF, 2011). These proposed transitions are representative of the global trend in REDD+ planning promoting shifts away from swidden. The Democratic Republic of Congo intends to 'increase productivity and sedentary lifestyle' of 50% of its subsistence farmers by 2030 with an aim of reducing pressures on forests (FCPF, 2010). Nepal also proposes to replace swidden agriculture with intensive agriculture, and Mozambique intends to 'eradicate slash-and-burn farming' (FCPF, 2010). Similar transitions have already been spurred under the Kyoto Protocol's Clean Development Mechanism (Hung, 2004; Satyanarayana, 2004; LTHRC, 2007).

Despite policy assumptions, our analysis of plot-level carbon outcomes highlights that direct transitions from swidden agriculture to permanent sedentary agriculture – or even rubber or oil palm plantations in some cases – will not necessarily deliver positive carbon outcomes. Such transitions may only result in positive carbon outcomes if a large proportion of the former cultivated land is abandoned, allowed to regenerate to permanent forest, and then protected. However, transitions towards agricultural intensification and high value crops do not necessarily result in land sparing, reduced deforestation or forest degradation elsewhere (Kaimowitz & Smith, 2001; Vandermeer & Perfecto, 2007; Matson & Vitousek, 2006; Rudel *et al.* 2009; Perfecto & Vandermeer, 2010). Previous interventions reveal that agricultural intensification can actually spur in-migration and agricultural expansion (Angelsen & Kaimowitz, 2001; Matson & Vitousek, 2006). This could be aggravated as agricultural intensification is also likely to increase future opportunity costs of implementing REDD+ (Ghazoul *et al.*, 2010). As such, it is foreseeable that REDD+ policies could incentivize transitions away from slash-and-burn agriculture, however, ultimately fail to preserve forest or reduce carbon emissions at larger scales and over time. Such concerns should deter policy simplifications about land-use transitions, although such simplifications are mainstream.

Current policy prescriptions for replacing all types of swidden agriculture may also represent a misreading of the long-term carbon landscape (cf. Dove, 1983; Fairhead & Leach, 1996). From a long-term carbon perspective, intermediate, and long-fallow swidden systems could conceivably represent optimal land-use options in some situations (Fox, 2000; Fox *et al.*, 2000). Although long-fallow swiddening results in a slow, net loss in carbon over time (Eaton & Lawrence, 2009), our analysis suggests that the losses associated with a transition to many other land uses (except tree-based plantations and forest) would be greater – and potentially, faster. In some situations, maximum carbon benefits may accrue by lengthening the fallow periods in existing swidden systems or managing the tree and bush phases of fallows more effectively. While caution is needed in the planning and implementation of such strategies, REDD+ policies should not preclude the option of maintaining or rehabilitating traditional, intermediate and long-fallow swidden, and agroforestry systems within the broader forest landscape. However, there are significant social and economic barriers to reversing the decades-long trend of shortening fallows and transitions away from traditional land management that result in negative carbon outcomes (Mertz *et al.*, 2009). Conflicting conservation, agricultural and land-use policies, human pressures on forest resources, and a range of economic factors fundamentally limit improved resource management. Furthermore, many government agencies fail to recognize the difference between abandoned and fallow lands, and few currently accept swidden agriculture as appropriate under any circumstances (Fox *et al.*, 2009; Ziegler *et al.*, 2009a, 2011).

Increased and appropriately funded research is required to improve estimates of AGC, BGC, and SOC stocks, and would help to identify optimal land uses and transitions, including locations and situations where swidden agriculture can deliver positive carbon outcomes. This will necessitate particular focus on below-ground carbon, which can sway decisions regarding optimal land use. Additional work is also needed to accurately quantify greenhouse gas flux and radiative forcing changes associated with all land-cover transitions.

Current methodologies outlined by the IPCC (2006) and GOFC-GOLD (2009) have the potential to produce reliable estimates (cf. Asner, 2009; Kampe *et al.*, 2010). However, our analyses highlight that extensive, supplementary fieldwork at fine spatial scales is needed because of the great variability among sites–particularly the uncertainty in below-ground carbon stocks. Even sites grouped into the same land-use categories and within the same region and country often have drastic differences because of differences in land-use history and geographical variables that affect vegetative succession. These can only be detected through new site-specific assessments. Importantly, ground-based inventories are essential because most space-borne monitoring techniques, upon which many REDD+ monitoring and reporting strategies plan to rely, are not capable of distinguishing these differences, or of detecting degradation accurately due to the large proportion of biomass in large trees, and because of the nonlinear links between vegetation indices, biomass, and stand characteristics, such as texture, complexity, or degree of fragmentation (Sader *et al.*, 1989; Foody *et al.*, 2001, 2003; Freitas *et al.*, 2005; Lu, 2005). Furthermore, satellite remote sensing (even hyperspectral) has not been shown to accurately determine SOC (Gomez *et al.*, 2008). Thus, improved future assessments, especially for transitions that do not have explicit positive carbon outcomes, could significantly increase the costs of REDD+ implementation and increase demands on participating countries. Extensive local collaboration for data collection (e.g., Danielsen *et al.*, 2011; Alternatives to Slash and Burn benchmark sites, ASB, 2011) are likely to play an important role in gathering such site-specific data and curtailing the costs of REDD+ implementation.

Although the principle REDD+ objectives of reducing deforestation and forest degradation have clear carbon benefits, many of the land-use transitions common across Southeast Asia and promoted by carbon forestry projects involve uncertain carbon outcomes. Considerably, expanded data collection on carbon stocks – particularly those below ground – is needed for all transformed land covers because existing information is inadequate for informed decision-making. This, however, will require a more informed, nuanced, and bottom-up approach to REDD+ policy making.

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Supporting Information

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

Table S1. Summary of above-ground biomass for forest (Mg C ha^{-1}).

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