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Significance of carbon stock uncertainties on emission reductions from deforestation and forest degradation in developing countries

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ABSTRACT

A historical agreement was reached in Bali under the United Nations Framework Convention on Climate Change, encouraging countries to initiate actions to reduce emissions from deforestation and forest degradation in developing countries (REDD). In this context, we use a Panama-based example to show the impacts of the current levels of uncertainty in forest carbon density estimates on GHG baseline estimation and estimations of emission reductions. Using five aboveground tree carbon stock estimates for Moist Tropical forest in a simulation study, we found a difference in terms of annual CO2 emissions of more than 100% between the lowest and the highest estimates. We analyze the economic significance to show that when comparing the income generated for the different forest carbon density estimates to the cost of 10% reduced deforestation, the break-even point differs from US\$6.74 to US\$16.58 per ton of CO2e between the highest and the lowest estimate. We argue that for a country such as Panama, improving the quality of forest carbon stock estimates would make economic sense since the highest forest carbon density estimates were developed nationally while the lowest estimate is the global default value. REDD could result in a huge incentive for forest protection and improved forest management, in consequence, we highlight that progress on the incorporation of uncertainty analysis and on the mitigation of the main sources of error in forest carbon density estimates merit further methodological guidance.

1. Introduction

In December 2007, the highly publicized "Bali Action Plan" was adopted at the conference of the United Nations Framework Convention on Climate Change (UNFCCC). This decision initiated a new era of discussion on the possible role of forests in the post-Kyoto climate regime to the Convention (Ott et al., 2008). The decision *encourages* parties to initiate activities to Reduce Emissions from Deforestation and forest Degradation (REDD) in developing countries (Article 3 in UNFCCC, 2007). Negotiations for the inclusion of tropical forests as a new avenue to climate change mitigation started in December 2005 as the governments of Papua New Guinea and Costa Rica brought the possibility of taking action to reduce emissions from deforestation in developing countries to the attention of the UNFCCC (UNFCCC, 2005). Although deforestation accounts for 10–25% of all greenhouse gas (GHG) emissions (Houghton, 2005a), previous attempts to reach an international agreement on forests under the UNFCCC have failed (LePrestre, 2005). During the negotiations of the Kyoto Protocol (KP), a variety of concerns restricted acceptable land-use mitigation activities to reforestation and afforestation (Streck and Scholz, 2006). The decision reached in Bali is therefore historical.

The program of work on REDD agreed to in Bali "invited Parties to submit their views on how to address outstanding methodological issues" (Article 7 in UNFCCC, 2007). The establishment of baseline that allows the demonstration of reductions in emissions from deforestation is one of the pending issues (DeFries et al., 2007). The notion of a baseline takes its roots in the rules guiding the Clean Development Mechanism (Decision 17/CP. 7, Marrakesh Accords). Carbon trading between developed and developing countries indeed requires project proponents to provide a baseline against which the real carbon removals are estimated (Auckland et al., 2003). It was suggested that a baseline for reducing emissions from deforestation could be based on historical emissions or could use historical emissions as input for business-as-usual projections (Olander et al., 2008) and would serve at calculating emission reductions. One proposal is that evaluation of baselines could rest on: 1) the assessment of changes in land use/land cover (activity data) and 2) the associated carbon stock change (Emission factor) (GOFC-GOLD, 2009).

During the REDD negotiations, several developed countries — EU, USA, Canada, Japan — as well as the Rainforest Coalition, an informal

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group of countries led by Papua New Guinea and Costa Rica, claimed that emission reductions from deforestation must be estimated against a national baseline of GHG emissions (Potvin and Bovarnick, 2008). National baselines are presented by their proponents as the only way to control leakage, or displacement of deforestation activities within a country. Conversely, a loose group of Spanish-speaking Latin American countries led by Columbia argues that national baselines are currently inapplicable because many countries lack the capacity and the necessary information to determine a national baseline for GHGs or do not fully control their territory. In Bali, when discussing the EU's proposed Indicative Guidance, countries agreed that demonstration activities could be done at both the national and the sub-national level. However, the issue remained contentious up to Copenhagen's 15th Conference of the Parties (Potvin, C., personal observation). Regardless of the scale at which baseline emissions are estimated, accuracy and precision are needed to ensure that the reductions compensated for in a hypothetical REDD mechanism are properly quantified (Mollicone et al., 2007a).

In Poznan at the fourteenth Conference of the Parties, the importance given to reference emission levels justified the request for an expert meeting on the topic (Article 6 in UNFCCC, 2008). The report on this meeting identifies outstanding issues and highlights the presence of gaps in data and data quality including *inter alia* standing stocks per hectare, estimates of biomass density, development of biomass expansion factors, and allometric equations and improved estimates at the levels of forest type and forest ecosystem (UNFCCC, 2009a). Furthermore, a technical paper of the UNFCCC on the cost of implementing methodologies and monitoring systems for REDD signals that the majority of non-Annex I countries have limited capacity in providing complete and accurate estimates of GHG emissions and removals from forests (UNFCCC, 2009b).

The SBSTA decision taken in Copenhagen (COP 15) signals that REDD-plus national monitoring systems need to provide estimates that are "transparent, consistent, as far as possible accurate, and that reduce uncertainties, taking into account national capabilities and capacities" (UNFCCC, 2009c).

The purpose of this paper is to assess the impact of uncertainties in forest carbon density on baseline estimation. We present this assessment in the context of the UNFCCC discussions on the current capability of developing countries to estimate emission baselines and other methodological issues to REDD. Using Panama as an example, we illustrate the sensitivity of a land-cover change emission model in regards to estimates of forest carbon density and discuss the different sources of error of these estimates. We examine the effect of uncertainties on possible payments for emission reductions from deforestation. We also highlight research needs for the improvement of forest carbon density estimation.

2. Methods

Panama's Moist Tropical Forest is its most extensive forest ecosystem, covering ~3,000,000 ha (Fig. 1). It is also the forest ecosystem suffering the greatest encroachment from deforestation nationally. To estimate the baseline for the Moist Tropical Forest of Panama we elaborated a modeling approach based on Ramankutty et al. (2007). The model estimated the carbon flux from land-cover change over the entire forest ecosystem. It contains two sections: 1) a land-cover transition model based on a first-order Markov matrix to simulate the land-cover dynamic following deforestation, and 2) a bookkeeping carbon cycle model to estimate the flux resulting from the land-cover dynamics. All model computer simulations were performed using MATLAB, version 7.6. The equations to the model can be found in appendix of Ramankutty et al. (2007).

2.1. Land-cover transition model

To parameterize the land-cover transition model, we compared two land-cover maps (1992 and 2000) to assess annual deforestation and obtain a transition probability matrix for the Moist Tropical Forest. These land-cover maps as well as a life zone map following Holdridge's classification were provided by Panama's Autoridad Nacional del Ambiente (ANAM). They were initially converted from

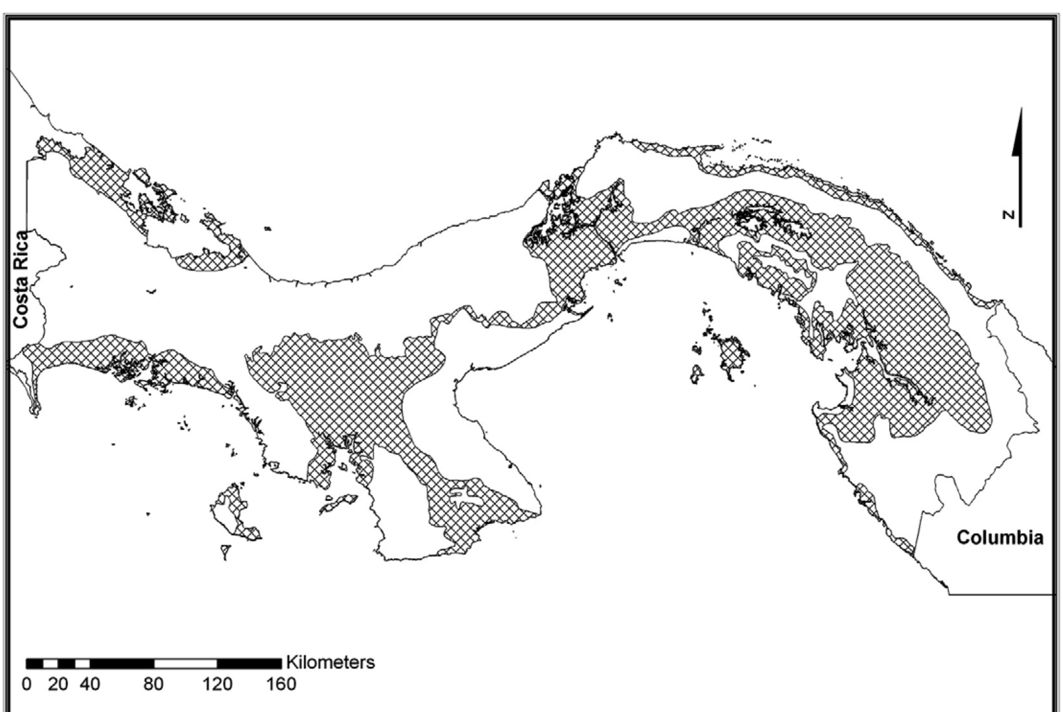

Fig. 1. Map representing the extent of the Moist Tropical Forests in Panama according to the Holdridge's life zone classification and covering approximately 3 million ha.

Table 1
Transition probability matrix for the Markov model of land-use change.

2000	1992				
	Mature forest	Secondary forest	Fallow	Agriculture	Other
Mature forest	0.9709	0.0211	0.0088	0.0006	0.0068
Secondary forest	0.0099	0.9163	0.0480	0.0027	0.0040
Fallow	0.0086	0.0425	0.8791	0.0661	0.0062
Agriculture	0.0099	0.0198	0.0601	0.9295	0.0080
Other	0.0007	0.0002	0.0040	0.0011	0.9751

vector to raster with a pixel size of 100 m by 100 m (area of 1 ha) under Lambert-Azimuthal Equal Area projection, using ArcGIS 9.3 ESRI®. Then, in order to obtain the Markov matrix of annual land-cover transition probabilities, we took the eighth root of the matrix. This matrix included five land-cover classes: Mature forest, Secondary forest, Fallow, Agriculture, and Other (ANAM/ITTO, 2003). Under this ANAM/ITTO classification, the mature forest category includes all forests and plantations with more than 80% tree cover. The secondary forest category covers re-growing, previously cleared, and degraded forest having between 60% and 80% tree cover. The fallow category includes re-growing vegetation following agricultural land abandonment with less than five years of age. The agriculture category was sub-divided into the average percentage area cover with annual crop, permanent crop, and pasture found in Panama's agricultural census (Contraloría, 2001). The “Other” category joined urban areas, inland water (such as lakes or reservoirs), and lowland vegetation liable to flooding (such as salt marshes). For the sake of this modeling exercise, the deforestation was assumed to be zero prior to 1992. The only anterior land-cover map that would be available for the country (Magallon, F., personal communication), was based on the conversion of Garver (1947) verbal descriptions into a land-cover map for 1947 (Heckadon Moreno, 1984; Wright and Samaniego, 2008). We decided not to include this assessment in the present study, but we acknowledge the fact that ignoring past deforestation could underestimate emissions for this time period (Ramankutty et al., 2007). See Table 1 for the transition probabilities among land-cover classes. The land-cover transition model was validated by running the simulation for the base year 1992, and by comparing the model's results with the reality observed on the 2000 map. The results concur to 100% for the year 2000.

2.2. Bookkeeping carbon cycle model

This section of the model served to calculate annual CO_2 fluxes originating from the land-cover change. We modeled the changes in aboveground live biomass only, since it was suggested that in the context of REDD, for monitoring purposes, only the dominant carbon tree pool would be considered as a key category (GOFC-GOLD, 2009). The model tracks the annual emissions and uptake following re-clearing and re-growth of fallow and secondary forest as well as carbon fluxes from permanent cultivation growth and clearing. Only changes in land cover are considered here; neither changes in land-use management nor the effect of natural or human disturbances (e.g. fire and insect outbreak) were considered although they could possibly affect carbon fluxes.

Emissions released following clearing events were partitioned into three pools. Following Gutierrez (1999), 60% of the carbon emissions were considered as immediately lost into the atmosphere due to burning of plant material, 34% were released at slower rate from decay of residues left on site, and 6% were temporarily stored in wood products. We used rates of decay estimates from the Brazilian Amazon for both dead material left on site and harvested woody material (Ramankutty et al., 2007) due to similar forest conditions, especially temperature and precipitation (Food and Agriculture Organization, 2006), and because we are unaware of any national decay data. Non-CO_2 gases (e.g. methane and nitrous oxide) liberated during the burning process and that depending on burning efficiency were not accounted for.

Re-clearing of secondary forest already present in 1992 was assigned a mean value of 80.4 tC/ha emitted (or transferred) from the forest C pool at the time of harvest and carbon re-accumulation was set at a rate of 3.4 tC ha^{-1} year^{-1} (Gutierrez, 2005). The re-growth and re-clearing of secondary forest formed since 1992 followed a logistic function in proportion to the mature forest mean carbon density relative to the age of the forest, where exponential growth in trees is considered in the first year (Potvin and Gotelli, 2008) and where we assumed the carbon stocks to be recovered completely after 75 years (Alves et al., 1997; Brown and Lugo, 1990). Secondary forest growth was simulated starting at the age of 5 years in order to correspond to the land-cover classification, and in particular to distinguish it from the fallow category. Only net changes in annual fallow areas were accounted for at a value of 35.4 tC ha^{-1}. The reverting mature forest class was assigned a plantation growth

Table 2
Summary of data used in the model.

Land-use class		Description	Partitioning of the deforested land	Standing carbon stock (tC ha^{-1})	Rate of C accumulation (tC ha^{-1} year^{-1})	Sources
Mature forest		All forests with more than 80% tree cover and plantations	–	Five estimates (Table 3)	4.3	Gutierrez (2005)
Secondary forest		Previously cleared and degraded forest having between 60% and 80% tree cover	0.312	80.4 fct[a]	3.4 fct[a]	Gutierrez (2005) Alves et al. (1997) Brown and Lugo (1982) Potvin and Gotelli (2008)
Fallow		Vegetation following agricultural land abandonment or slash and burn cultivation with less than five years old	0.307	35.4	–	FRA (2005) Tschakert et al. (2007)
Agriculture	Annual crop	Crops where the vegetation is collected every year	0.353 (0.246)[b]	–	–	
	Pasture	Including managed and unmanaged pasture for cattle	0.353 (0.688)[b]	4.2	–	Kirby and Potvin (2007)
	Permanent crop	Including cocoa, coffee, banana plantations	0.353 (0.066)[b]	50.0	10.0	IPCC (2003) Schroeder (1994)
Other		Urban areas, inland water, and lowland vegetation liable to flooding	0.028	–	–	

[a] The function used to calculate the standing stock of the secondary forest was $Csf = Cveg/(1+e^{1.7-0.105*(t)})$ where t is time in years $Cveg$ is the standing stock in mature forest, and Csf the standing stock in secondary forest. The reverting rate was calculated as $\Delta Csf = f(t) - f(t-1)$.

[b] The fraction of agricultural land in annual crop, pasture, and permanent crop were obtained from the VI Agricultural Census in Contraloría (2001).

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Table 3
Characteristics specific to the five estimates of biomass carbon density for the Moist Tropical Forests of Panama used in the sensitivity analysis to the land-use change emissions model.

Source of the estimate	Site	Measurements for AGB[a]	Plot size (ha)	Number of plots	Description of forest	Estimate (tC/ha)	Model name	Model type/characteristics
Kirby and Potvin (2007)	Ipeti-Embera	All trees ≥ 10 cm DBH[b]	0.07	32	Old-growth, managed by local community	245	Brown (1997)	Allometric model linking DBH to AGB. This estimate was produced using the large-tree correction proposed by Brown (1997), but without the correction for species-specific WD[c] (see Kirby and Potvin, 2007: Appendix A, for further discussion).
Kirby and Potvin (2007)	Ipeti-Embera	All trees ≥ 10 cm DBH[b]	0.07	32	Idem	169.1	Chave et al. (2004)	Allometric model linking DBH to AGB. Model provides conservative estimates of large-tree AGB relative to Brown (1997). This estimate was produced without correcting model for species-specific WD[c] (Chave et al., 2004).
Chave et al. (2004)	Panama Canal Watershed	All trees ≥ 1 cm DBH[b]	50	1	Late-secondary and primary forests	138.5	Chave et al. (2004)	Idem
Gutierrez (2005)	Eastern Panama	All trees with variable minimum DBH[b]	NA[d]	NA	NA	130.2	Brown (1997)	BEF[e] to convert commercial volume estimates to biomass carbon density (t/ha).
IPCC	Global estimate	NA	NA	NA	NA	108.5	NA	NA

[a] AGB = aboveground live biomass.
[b] DBH = diameter at breast height.
[c] WD = wood density.
[d] NA = not available.
[e] BEF = biomass expansion factor.

rate. Pasture land was assumed to store 4.2 tC ha^{-1}, with a three-year burning cycle (Kirby and Potvin, 2007). Permanent crops were considered to sequester carbon at a rate of 10 tC ha^{-1} year^{-1}, while the clearing of permanent crops was assigned a mean value of 50 tC ha^{-1} (IPCC, 2003; Schroeder, 1994). Table 2 summarizes the parameters used in this model.

2.3. Sensitivity to forest carbon density

The model served to test the sensitivity of different aboveground live tree carbon density (hereafter forest carbon density — FCD) estimates on baseline estimation and emission reductions from REDD. For the purpose of this analysis, we kept the above values constant in order to test the effect of different estimates of FCD only. Five published FCD estimates were used in the model described above to calculate annual CO_2 emissions from land-cover change in Panama's Moist Tropical Forest between 1992 and 2000 (IPCC, 2003; Chave, et al., 2004; Gutierrez, 2005; Kirby and Potvin, 2007). With the exception of the IPCC default value (Annex 3A.1, Default tables for Section 3.2 Forest land, Table 3A.1.2.), all estimates were evaluated using ground-based forest measurements. The four Panama-based estimates differ in terms of both the inventory methods used to collect tree dimension data and the allometric equations used to relate tree dimensions to oven-dried biomass (Table 3). To assess the impact of allometric models on FCD uncertainty, we include two FCD values derived from a single set of inventory data (Kirby and Potvin, 2007). In all cases, where biomass rather than carbon stocks were reported in the original studies, we assume carbon to account for half of the biomass value (Houghton, 2003).

2.4. Sensitivity of the economics of REDD

Uncertainties in forest C density have implications for the economics of REDD. To illustrate this point, we carried out a back-of-the-envelope financial analysis for the case study in Panama to compare the potential income generated from REDD with the cost of avoiding deforestation. To look at the effect of forest carbon estimates on potential estimated income from REDD, we applied a 10% reduction of annual deforestation or the equivalent of 2170 ha of mature forest to be conserved yearly, for a period of eight years. This would be a realistic figure according to a government official (Potvin et al., 2008). We evaluated the total emission reductions (TER) for the five FCD estimates by comparing the business-as-usual (BAU) model results to a scenario of 10% annual avoided deforestation (AD) scenario, which can be expressed by:

$$TER_c = \sum f_c(BAU, t) - f_c(AD, t) \tag{1}$$

where TER_c is the total emission reductions in tons of C ha^{-1} per FCD estimate, f_c is the model where the subscript $c = 1$ to 5 for one of the five FCD estimates, BAU stands for business-as-usual deforestation, AD stands for a 10% deforestation reduction and, $t = 1$ to 8 for the eight years of avoided deforestation.

Then, we calculated the potential income by multiplying the total emission reductions to a range of market values of US\$0.50 to US\$30 by ton of CO_{2e}. The potential income generated for emission reductions from avoiding deforestation was calculated as:

$$I_c = TER_c{}^*P \tag{2}$$

where P is the price of carbon where $P = 0.5$ to 30 (\$US t^{-1} CO_2 e) and I_c is the income for FCD estimate c.

This hypothetical income generated through REDD, that depends upon the carbon density of forests, was compared with the cost of avoiding deforestation, a value that is independent of carbon density. Using a discount rate of 5%, Potvin et al. (2008) estimated the overall cost to avoid deforesting 5000 ha per year in Panama for 25 years at US\$114,663,825 with an annual mean of US\$4,586,553, including the land opportunity cost, the cost of protection, transaction, and administration. This value corresponds to a net present value of \$917.31 ha^{-1} year^{-1}. The land-use opportunity cost was estimated in comparison with the income generated by small-scale cattle ranching, a preferred land use in Panama (Coomes et al., 2008). Other available estimates of land-use opportunity costs fall within the same range of values (Louis Berger Group, 2006; Barzev, 2006).

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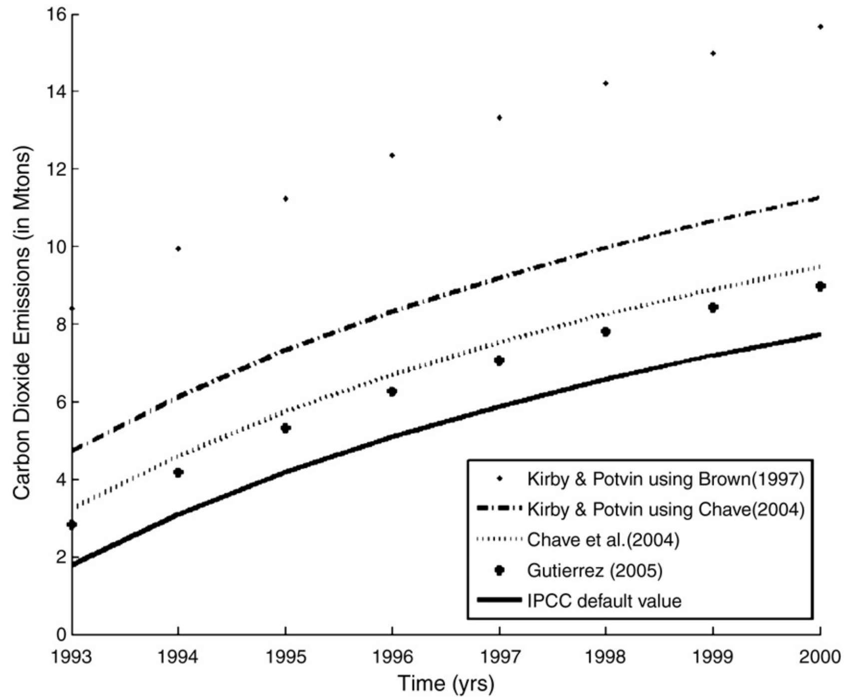

Fig. 2. This figure shows the response of the model to changes in forest carbon stock values on estimated annual CO_2 emissions from land-cover change. Five published estimates of aboveground tree carbon stocks are compared for the Moist Tropical Forests: 130 tC ha^{-1} Gutierrez (2005), 139 tC ha^{-1} (Chave et al., 2004), 109 tC ha^{-1} (IPCC, 2003), and 169 and 245 tC ha^{-1} (Kirby and Potvin, 2007). The last two estimates are based on the same inventory data but use two different allometric models to convert tree measurements to carbon estimates.

The total cost of REDD was estimated as follows:

$$TCD = 917.31 \times \sum (AD^{*}t) \quad (3)$$

where TCD signifies the total cost of avoided deforestation, $917.31 is the overall cost of avoiding deforestation on a per ha basis (ha^{-1} year^{-1}) and, $t = 1$ to 8 for the eight-year avoided deforestation period.

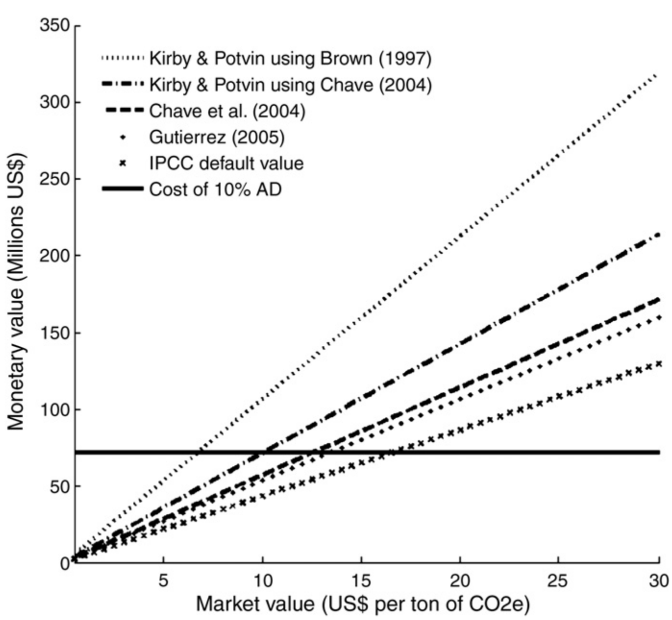

Fig. 3. Comparison of the estimated income received to reduce deforestation by 10% annually for 8 years, with an equivalent of 2170 ha per year, obtained from five different forest carbon density estimates for the Moist Tropical Forest of Panama. The income is estimated in function of the total emission reductions (TER) and the market value per ton of CO_{2e}. The black solid line is the overall cost on a per hectare basis estimated from Potvin et al. (2008). The break-even points are located where the colored lines cross the black line.

Note that the area of deforestation avoidance is cumulative through time, and that protected forest involves an annual cost. Finally, the break-even point of REDD is located where the income from REDD equals the overall cost of avoiding deforestation.

3. Results

Based on the analysis described above, applying the five different estimates of FCD, the sensitivity to changes in this parameter for the Moist Tropical Forest between 1992 and 2000 proves notable. In a single year, the choice of C stock density can result in estimates of annual emissions between the models that differ by 8.0 million t CO_{2e}, with a 103% increase in value between the lowest and the highest estimates (Fig. 2). When we compare the two FCD values obtained from a single set of inventory data but differing in terms of allometric

Table 4
Total emission reductions comparison for five Moist Tropical FCD estimates in Panama, assuming a 10% reduction of deforestation over an eight-year period and break-even points per ton of CO_{2e}. The overall cost for avoiding deforestation was calculated in function of the area protected, using a net present value of $917.31 on a per ha basis (Potvin et al., 2008).

Estimate for the Moist Tropical Forest	Allometric model	Aboveground tree carbon stock (in tons/ha)	Total emission reductions (in Mtons of CO_{2e})	Break-even point (in US$ per ton of CO_{2e})
Kirby and Potvin (2007)	Brown (1997)	245.0	10.6	6.74
Kirby and Potvin (2007)	Chave et al. (2004)	169.1	7.1	10.06
Chave et al. (2004)	Chave et al. (2004)	138.5	5.7	12.55
Gutierrez (2005)	Brown (1997)	130.2	5.3	13.46
IPCC default value	–	108.5	4.3	16.58

equations used (Kirby and Potvin, 2007), the difference between the mean annual emissions for these two estimates is 48%. Our simulation also shows that the IPCC default value yields the lowest estimates of all including the more recent independent scientific values.

In addition, we assessed the impact of the FCD estimates on the evaluation of emission reductions. We calculated emission reductions by comparing the scenario of 10% deforestation reduction with a reference emission level (BAU baseline) for each FCD estimate. When we compared mean annual emission reductions over the eight-year period obtained using these five estimates, the difference between the lowest and the highest estimate is 144%.

Part of the difference between the emission estimates is attributable to the model structure that calculates the carbon density held in re-growing secondary forest as a function of time relative to the proportion of mature FCD (see logistic equation in Table 2). Logically, secondary forest should not have higher carbon density than mature forest unless specific forest carbon management is adopted.

For the economic analysis, we also used a scenario of a 10% reduction in annual deforestation for the calculation. This corresponds to a reduction of 2170 ha per year. Using published overall cost estimate per hectare for Panama (Potvin et al., 2008), our analysis shows the significance of the choice of carbon density estimate on the economics of REDD. Not surprisingly, the results show that the net economic benefit of REDD would be higher, due to greater emission reductions accounted as a result of higher estimated carbon density. Fig. 3 shows that the economic significance of the choice of carbon density estimate increases as the market value per ton of CO_{2e} increases. It may not matter so much which one is chosen when C price is \$1–5, but it becomes much more meaningful at \$15–20 per ton of CO_{2e}.

Yet when comparing the income generated for the different FCD estimates to the cost of 10% reduced deforestation, the break-even point differs from US\$6.74 to US\$16.58 per ton of CO_{2e} for the highest vs. the lowest FCD estimate (Table 4). Thus the economic feasibility of REDD will depend directly on the values of FCD. From a developing country perspective, knowledge of forest carbon stocks is a necessary condition to decide the price at which selling carbon credits become profitable or not.

4. Discussion

4.1. Sources of uncertainty

The contribution of uncertainties in FCD as a source of error in the quantification of emissions from land-cover change in the tropics is receiving a growing body of attention (GOFC-GOLD, 2009; Grassi et al., 2008; Mollicone et al., 2007b; Ramankutty et al., 2007; Houghton, 2005b). FCD is known to vary regionally depending on temperature, elevation, precipitation, tree species composition, disturbance, and soil fertility (Laurance et al., 1999; Clark and Clark, 2000; Malhi et al., 2006; Urquiza-Haas et al., 2007). Beyond this natural variation, FCD uncertainties can also result from estimation methods. Two main constituents can affect FCD estimates: inventory protocol and the method used to convert tree measurement to biomass. A third error factor, which we did not explore in our simulations, stems from uncertainty in accounting for other forest C pools.

Primarily, the error imputable to the inventory protocol includes random sampling error (plot size and number of data points) and, lack of representativeness or systematic error (e.g. possible biases in selecting attractive forests) (IPCC, 2000; Chave et al., 2004; Grassi et al., 2008). The latter is often harder to quantify, but nonetheless important.

Secondly, uncertainties can also stem from methods of biomass estimation whether relying on allometric equations or on biomass expansion factors (BEFs). The error imputable to the choice of allometric model to estimate FCD has also been discussed in the literature where authors have qualified it as being of crucial importance (Clark and Clark, 2000; Keller et al., 2001; Chave et al., 2004; Chave et al., 2005). In temperate regions, allometric models have been developed for individual tree species, whereas in the tropics the high tree diversity renders this approach impractical. For example, in a 50 ha forest plot in Barro Colorado Island, Panama, approximately 300 tree species have been identified (Condit et al., 2004; Hubbell, 2006). As a surrogate, scientists have developed generalized allometric models that use measured forest attributes and relate them statistically to measurements obtained from the destructive sampling of a large number of trees (Brown, 1997; Chave et al., 2005). Results from the literature show that the choice of allometric equations can explain an error of greater than 20% of aboveground tree biomass estimates (Clark and Clark, 2000; Keller et al., 2001; Chave et al., 2004) and can be amplified when large trees are numerous (Kirby and Potvin, 2007).

Besides this, another method to convert forest inventory data to FCD estimates is the use of biomass expansion factors (BEFs), which employ ratios to convert wood volume to biomass (e.g. Table 3, Gutierrez (2005) estimate). BEFs require the estimation of wood volumes, followed by application of expansion factors to account for non-inventoried tree components, then propagating sources of error along the way (Brown, 1997; Nogueira et al., 2008). The uncertainty in conversion from tree volume to carbon content is one of the major gaps in carbon accounting at regional and national levels, but also the scant presence of quantitative uncertainty analysis (apart from expert knowledge) is obvious (Fehrmann and Kleinn, 2006; Lehtonen et al., 2007). With both methods, the relative accuracy and precision depend on the underlying data used to derive the allometric model or the ratio of volume to biomass. In this study, we illustrated the point by using two allometric equations on the same inventory data; the different results prove that allometric models are another important source of uncertainty in the quantification of emissions from land-cover change.

Ultimately, as noted in other studies, another source of uncertainty roots from the inclusion of distinct field measurements or adjustments for other C pools, explaining also the discrepancy in total FCD estimates (Houghton et al., 2000; Keller et al., 2001). It is important to emphasize that in this exercise we focused our attention on uncertainty in aboveground live FCD only. While countries willing to engage in REDD could participate by only tracking changes in aboveground live biomass, an obliged key category (GOFC-GOLD, 2009), including a range of estimates for the other C pools (roots, woody debris, litter, and soil organic carbon (SOC)) would further increase the overall uncertainty of the reference emission levels. For the case of the SOC, studies signal that land-cover change could result in changes in soil carbon density by 13 to 59% (Guo and Gifford, 2002) and that conversion of forest to cropland generally leads to a loss of soil carbon (Murty et al., 2002). The studies of SOC in Panama that we reviewed showed stability in this carbon pool across land-use types (with no difference between forest, pasture, young fallow, old fallow, subsistence agriculture plots and native tree plantations) (Kirby and Potvin, 2007; Potvin et al., 2004; Tschakert et al., 2007; Schwendenmann and Pendall, 2006). However, Kirby and Potvin (2007) note that none of these studies tracked changes in SOC at the same site through time, which would provide more reliable estimates of changes in SOC with land-use change. In conclusion, while the inclusion of other pools in a REDD national (or sub-national) monitoring system will most probably depend on the financial resources available; efforts to improve and standardize methodologies for monitoring carbon in these pools are also needed.

4.2. IPCC default value

A set of guidelines produced by the Intergovernmental Panel on Climate Change (IPCC), opens three methodological avenues to countries for estimating national greenhouse gases (GHGs), according to different levels of quality from a very coarse to a highly detailed assessment. Emission categories that are considered key because of their significant influence on a country's total inventory of direct GHGs should be estimated using sophisticated calculations and nationally developed models, and have been termed Tier 2 and 3 methodologies depending on the level of details provided. For less important emission categories, or when data is not available, default values and simpler approaches (for example, Tier 1 methodologies (IPCC, 2003)) could be used.

The UNFCCC's technical paper on the cost of implementing methodologies and monitoring systems for REDD (UNFCCC, 2009b) suggests that many developing countries do not have the financial and/or human capital necessary to produce national estimates to comply with Tier 2 or 3 methodologies. As a result, IPCC default values are likely to be used to evaluate REDD. However, in countries such as Panama where the forests are tall and dense, using the IPCC default values would be disadvantageous for the country, although the estimate would be conservative as emission reductions would not be overestimated. The REDD negotiation hinges around the notion that developing countries would be paid to reduce emissions on the basis of tradable emissions reduction units expressed as t CO_{2e}. Using default values that underestimate carbon density would allow developing countries willing to engage in the fight against deforestation to be able to claim less than they could if they improved their inventories. Thus, improving the quality of FCD estimates in tropical forests can be justified economically compared to the use of a global default value. While accurate and precise estimates of FCD in tropical areas will likely translate into higher REDD estimates (Grassi et al., 2008), our simple economic calculation indicates that this might result in a lower break-even price when a nation sells hypothetical REDD credits. We argue here, that in turn this would enhance the likelihood of successful REDD implementation since countries with tall and dense tropical forest would have to successfully halt forest cover loss over a smaller surface area to reach a given emission reduction.

4.3. Tier 2 or 3 methodologies: overcoming technical challenges

In the absence of a well-designed, regional-scale sampling effort, the choice of the "right" estimate for the carbon density of Panama's Moist Tropical Forests is quite subjective. The inventories described in Chave et al. (2004) and Kirby and Potvin (2007) are most probably not representative of Panama's Moist Tropical Forests as a whole because they are based on relatively small-scale samples that did not cover the entire region. The estimate calculated for the Forest Resource Assessment (Gutierrez, 2005) comes from different scientific and commercial forest inventories from the 1970s forward that did not follow a single, standard methodological protocol. For instance, some inventories only measured trees ≥60 cm diameter at breast height (dbh) while others started at ≥40 cm. Even if adjusted *a posteriori* to produce a single estimate of carbon density per hectare, the result has a low confidence level. Therefore, technical guidance from forest scientists is needed if developing countries such as Panama want to improve the quality of FCD estimates and tackle a Tier 2–3 methodology to REDD.

Moreover, the case of forest degradation, explicitly included in the Decision 2/CP13 (UNFCCC, 2007), is an example where the carbon density changes might be hard to estimate but those estimates will affect the income generated. Forest degradation is a land-cover modification rather than conversion which results from human activities that partially reduce FCD without regeneration in a reasonable time frame (in the order of a decade) (Lambin, 1999; Defries et al., 2007). In the context of current UNFCCC discussions, forest degradation is essentially a non-temporary reduction of FCD. According to the UNFCCC definition of forest set at a minimum area of 0.05–1.0 ha with 10–30% tree crown cover, a substantial decrease in the carbon density can occur without any change in classification (Sasaki and Putz, 2009). The potential for selective logging, *inter alia*, to lead to an important reduction of FCD has been highlighted in both empirical and theoretical studies (Gaston et al., 1998; Asner et al., 2005a; Asner et al., 2005b; Bunker et al., 2005; Souza et al., 2005; Laporte et al., 2007; Putz et al., 2008). The variability of FCD in the landscape is expected to increase due to the impacts of varying intensities of selective logging or other agents such as fires on forest structure and composition (Gerwing, 2002). The comparison with uncertain estimates of mature forest might result in quite small conservative emission reductions.

One way in which scientists could contribute to the REDD agenda is by ensuring that countries have access to the most recent data and methods on carbon density estimates. In this context, the recently published research by Gibbs et al. (2007) presenting an updated global map of national-level carbon density estimates deserves mention. Also, the effort of the Center for Tropical Forest Science to undertake a full assessment of carbon density changes in their ten large (16 to 52 ha each) forest plots world-wide should be applauded (Chave et al., 2005; Chave et al., 2008). In a recent study for the Amazon basin, allometric equations from directly weighed trees in small-scale samples in specific forest types were used to assess uncertainties and improve models for biomass estimates based on wood-volume data from large-scale inventories (Nogueira et al., 2008). Further efforts to improve our knowledge of tropical FCD should be encouraged.

In conclusion, REDD is surging forward as a historical incentive for forest protection and improved forest management in the tropics. Our results suggest that the impact of uncertainties in FCD is an outstanding methodological issue that could affect the quantification of emission reductions and potential payments to developing countries for avoiding deforestation. The model applied for this study concentrated on the effects of changes in land cover and did not consider changes in land-use management or the effect of natural or human disturbances (e.g. fire and insect outbreak) possibly affecting carbon flux. Our study highlights that it may be worthwhile for national governments to recognize the potential value of improving/developing good national forest carbon monitoring systems in the context of REDD, under the UNFCCC. Finally, REDD methodological guidance should include the means to stimulate continuous progress on the incorporation of uncertainty analyses and on the mitigation of the main sources of error in the quantification of emissions from land-cover change, particularly on forest carbon density estimates.

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