Supplementing REDD+ with Biodiversity Payments: The Paradox of Paying for Multiple Ecosystem Services

Jonah Busch

Abstract

An international mechanism to reduce emissions from deforestation using carbon payments (REDD+) can be leveraged to make payments for forests' biodiversity as well. Paradoxically, under conditions consistent with emerging REDD+ programs, money spent on a mixture of carbon payments and biodiversity payments has the potential to incentivize the provision of greater climate benefits than an equal amount of money spent only on carbon payments.

This paradoxical result arises when diversifying payments across multiple services allows a funding agency to spend less on additional rents to existing suppliers of avoided deforestation and more on incentivizing the participation of new suppliers.

JEL Codes: Q23, Q24, Q54, Q57

Keywords: Biodiversity conservation; climate change; deforestation; payments for ecosystem services (PES)

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Contents

Introduction

An emerging international climate policy mechanism called REDD+ (UNFCCC, 2010) is being designed to deliver climate change mitigation benefits from tropical forests. REDD+ would make carbon payments to developing countries that reduce greenhouse gas emissions from deforestation below internationally agreed reference levels. Developing countries would be responsible for designing their own policies to achieve these emission reductions.

But forests not only store carbon. They jointly provide many other ecosystem services, including habitat for over two-thirds of known terrestrial species (Raven, 1988). Thus a REDD+ mechanism that pays for climate mitigation is also expected to benefit forest-dependent biodiversity by conserving forest habitat that would otherwise have been cleared (Busch et al., 2011). However, a REDD+ mechanism whose incentives are focused solely on carbon storage risks undesirable consequences for biodiversity. Such a REDD+ mechanism could favor the conservation of higher-carbon forests over higher-biodiversity forests (Putz and Redford, 2009; Paoli et al, 2010; Siikamaki and Newbold, 2012) or could displace agricultural activity into low-carbon but biologically important landscapes (Miles and Kapos, 2008).

There is substantial interest in policies to increase the biodiversity benefits or ameliorate the biodiversity risks associated with REDD+. This includes more closely linking the objectives of the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biodiversity (CBD) (Secretariat of the CBD 2009), but extends more broadly as well. Harvey et al. (2009) distinguish pro-biodiversity policies between those that contribute to greater climate mitigation and those that present a tradeoff with weakened or delayed climate mitigation. Policies that promote both greater biodiversity conservation and greater carbon storage include increasing finance for REDD+ (Busch et al., 2011, Strassburg et al., 2012), strengthening institutions to handle large financial flows under REDD+ (Ring et al., 2010), minimizing leakage of deforestation to regions with high forest cover and low deforestation rates (da Fonseca et al., 2007; Busch et al., 2011), and ensuring that definitions of forest proclude incentives for the conversion of natural forest to low-carbon, low-biodiversity plantation crops (e.g. oil palm) (Sasaki and Putz, 2008). Policies that present tradeoffs between biodiversity conservation and carbon storage include geographically prioritizing the conservation of forests that are richest in biodiversity (Kapos et al., 2008; Venter et al., 2009; Strassburg et al., 2010; Larsen et al., 2011; Gardner et al., 2012), monitoring the impacts of REDD+ on biodiversity (Gardner et al., 2012), and enacting safeguards to prevent the afforestation of biologically significant grasslands (Stickler et al., 2009).

A commonly suggested policy to increase the biodiversity benefits of REDD+ is supplementing carbon payments with biodiversity payments (Venter et al., 2009; Strassburg et al., 2010; Dinerstein et al., 2010; Busch et al., 2011; Collins et al., 2011). Biodiversity payments could be made to countries or sites based on the biodiversity value of their avoided deforestation, leveraging the institutional infrastructure put in place for carbon payments. This institutional infrastructure includes systems for monitoring, reporting and verifying forest loss, systems of finance, accounting, and governance capable of channeling billions of dollars worth of international demand for climate change mitigation to developing countries, and systems for crediting reductions with the international legitimacy conferred by a United Nations agreement. Biodiversity payments could leverage and guide a potentially vast new source of demand for forest conservation toward high-biodiversity forests, with far lower transaction costs and startup costs than stand-alone biodiversity projects.

Would biodiversity payments contribute to the climate mitigation goals of REDD+ or trade off against them? Clearly if funding for biodiversity payments is entirely additional to funding for carbon payments, then biodiversity payments would incentivize the conservation of additional forests, along with their carbon, benefiting both biodiversity and the climate. But what about the case where the funding for biodiversity payments could otherwise have been spent on carbon payments? This paper will show that money spent on biodiversity payments can in some circumstances result in greater carbon storage than an equal amount of money spent directly on carbon payments.

If total funding for carbon payments and biodiversity payments is constant, why wouldn't there be a strict tradeoff between the provision of carbon and biodiversity? Readers may instinctively envision that any money spent on biodiversity payments rather than carbon payments would result in the provision of less carbon storage, just as any public money spent on national defense ("guns") leaves less public money to spend on domestic welfare ("butter") (Mankiw, 2008). Previous studies of climate and biodiversity considered a conservation planning framework in which a central agency with a constrained budget is able to purchase conservation from landowners based on their opportunity cost, with no producer surplus for suppliers (Nelson et al., 2008; Nelson et al., 2009; Venter et al., 2009; Larsen et al., 2011). These studies indeed found a strict tradeoff between paying to conserve higher-carbon forests and paying to conserve higher-biodiversity forests. In these studies, improved biodiversity outcomes necessarily came at the cost of forgone climate outcomes, even if the improvement to biodiversity was large and the sacrifice to climate was small.

However, the emerging REDD+ mechanism differs from the conservation planning framework used in these previous studies in a fundamental aspect—it is unlikely that an international REDD+ mechanism will differentiate the price of payments to participating forest countries or localities to match suppliers' opportunity costs. In recent global climate decisions (UNFCCC, 2009; UNFCCC, 2010), and in precedent-setting bilateral agreements between Norway and Brazil (Ministry of Environment, 2010) and between Norway and Guyana (Government of Guyana and Government of Norway, 2009), REDD+ payments are not based on opportunity cost, but rather are proportional to the quantity of the climate service provided. Payments are equal to reductions in emissions below an agreed reference level, multiplied by a carbon price. Even within individual forested countries, where national governments will have greater flexibility to structure their own economic incentives for reducing deforestation (e.g. Busch et al., 2012), it appears likely that national payments to local land users or communities would also be based on a carbon price rather than differentiated by opportunity costs. A standard carbon price could result because of competition between multiple buyers, or because equity considerations place a political constraint on price discrimination by a government monopsonist (Gregersen et al., 2010), or because payment programs that include no additional supplier surplus for conserving forest would negate the economic incentive to conserve forest (Borner et al., 2010), or because the benefits of economic mechanisms to reveal private willingness-to-accept such as reverse auctions (Stoneham et al., 2003; Ferraro 2007; Obersteiner et al., 2010) are judged to be outweighed by the costs associated with establishing such mechanisms in a developing country context.

If a REDD+ mechanism compensates ecosystem service suppliers based on a price per unit of climate service supplied rather than on their opportunity cost, then the strict tradeoff between climate outcomes and biodiversity outcomes found by previous literature no longer holds in all cases. The possibility exists that climate outcomes could be helped – not hindered – by shifting some funding from carbon toward biodiversity. Money spent on a mixture of payments for carbon and payments for biodiversity could produce a greater climate benefit than if that money were spent entirely on payments for carbon. This effect is termed here the "paradox of paying for multiple ecosystem services."

Intuitive and graphical exposition of the paradox of paying for multiple ecosystem services

Money spent on a mixture of carbon payments and biodiversity payments can sometimes produce a greater climate benefit than if the same amount of money were spent entirely on carbon payments. This paradoxical outcome is produced by a confluence of four conditions: a joint production function, supplier rent, endogenous prices, and diminishing returns. First, forested land jointly produces both carbon storage and habitat for biodiversity. A forest conserved to obtain carbon payments will provide some habitat for biodiversity, and a forest conserved to obtain biodiversity payments will provide some carbon storage. A joint production function for biodiversity and carbon is necessary for the paradox, but is not in itself sufficient. Second, when suppliers of avoided deforestation are compensated based on a standard price per unit of service supplied, the entire service payment does not simply compensate suppliers for their opportunity costs, as it does under an opportunity cost payment scheme (Figure 1a). Some portion of the payment increases suppliers' surplus, or rent, over and above their opportunity cost. Achieving additional units of carbon storage requires raising the carbon payment price. This increase in the carbon price not only outcompetes the opportunity cost of the marginal carbon supplier, incentivizing their participation, but increases the surplus of every other supplier as well (Figure 1b). This supplier surplus may well be judged as beneficial from a societal perspective, but from the point of view of an implementing agency concerned only with ecosystem services does not contribute directly to the programmatic objective. Third, service prices are endogenous and jointly dependent. If the total funding for carbon and biodiversity payments is constant, a decrease in the payment price for carbon allows for an increase in the payment price for biodiversity, and *vice versa*. Fourth, carbon payments may face diminishing marginal returns over some portion of their range. As the carbon price is raised, an increasing fraction of carbon payments may go to paying rent to suppliers who would have supplied anyway, rather than incentivizing additional reductions from new suppliers. Biodiversity payments, on the other hand, could have a greater fraction of payments incentivize the participation of new suppliers for whom the combined revenue from carbon payments and biodiversity payments is sufficient to outcompete opportunity cost (Figure 1c).

The resulting paradox can be expressed in four related formulations. First, *shifting the mixture of payments made from of a fund of fixed size from carbon payments towards biodiversity payments can in some cases result in increased carbon storage.* A second formulation describes a special case of the first in which initial payments are entirely for carbon: *shifting payments made from a fund of fixed size from carbon payments only to a mixture of carbon payments and biodiversity payments can in some cases result in increased carbon storage.* Third, and equivalently, *an additional biodiversity payment for* (Figure 2b) *can in some cases result in more carbon storage than an equal incremental carbon payment* (Figure 2b). The equivalence of the second and third formulations is illustrated by vector summation ($\overrightarrow{ab} + \overrightarrow{ac} = \overrightarrow{bc}$ in Figure 3). A fourth formulation is slightly less restrictive than the third: *an additional biodiversity payment can in some cases produce carbon storage more cost-efficiently than an incremental carbon payment.* That the fourth formulation of the paradox holds is demonstrated analytically in Section III. That the second formulation of the paradox holds is shown numerically in Sections IV-V.

In any formulation, this paradox occurs when a smaller fraction of biodiversity payments is spent on rent and a greater fraction is spent on incentivizing the provision of carbon from additional suppliers, relative to carbon payments. While the production possibility frontier for biodiversity and carbon is always downward-sloping under an opportunity cost payment framework, the production possibility frontier for biodiversity and carbon bubbles outward under a service price payment framework (Figure 3). As a result REDD+ funding agencies could in some cases achieve both greater climate benefits and greater biodiversity benefits by diversifying payments entirely for carbon to a mixture of payments for carbon and biodiversity.

Carbon, biodiversity, opportunity cost, and forest conservation

Let us consider a landscape of n forested sites. Each site $i \in [1, n]$ contains a forest that jointly produces two public ecosystem services: habitat for biodiversity of value b_i, and carbon storage of value c_i. Each site also has the potential for a net increase in private rental value, or opportunity cost, a_i, if converted from forest to agriculture. Land managers of every site will choose with certainty to completely deforest the land for agricultural use, unless they receive a payment for ecosystem services, p_i, that equals or exceeds their opportunity cost: $i \in I \ iff \ p_i \geq a_i$, where I represents the set of all sites i that are kept forested rather than converted. This commonly employed opportunity cost assumption offers a useful if simplistic starting point for analyzing which sites on a landscape are likely to participate in REDD+ programs (Busch et al., 2009). For richer modeling of REDD+ that includes reference levels, multi-tier participation decisions, leakage, and revealed preferences see Busch et al. (2012).

A central agency such as a government or multilateral facility has funds for forest conservation of fixed size F from which it makes payments to sites in exchange for avoided deforestation. The agency seeks to maximize the total value of carbon across the landscape. The landscape's total biodiversity value, B, and carbon value, C, are assumed to be additive and do not depend upon the configuration of forested and deforested sites: $B = \sum_{i \in I} b_i$ and $C = \sum_{i \in I} c_i$. While REDD+ in its entirety includes deforestation, forest degradation, conservation, sustainable management of forests, and enhancement of forest carbon stocks, in this model only emissions from deforestation are considered.

Payments are made under one of two alternative frameworks. In the "opportunity cost payment framework," the agency is able to differentiate its payments to sites based on sites' opportunity costs. This payment framework relies on the assumption that all sites' opportunity costs are known to the agency and that differentiating prices based on these opportunity costs is politically and programmatically feasible. The agency can choose to prioritize sites based on any mixture of biodiversity and carbon, where $\propto \in [0,1]$ represents the preference of the agency for biodiversity relative to carbon in selecting sites. The agency first ranks sites in order of highest to lowest benefit-to-cost ratio, $\frac{\alpha b_i + (1-\alpha) c_i}{a_i}$. The agency then makes payments to sites equal to the sites' opportunity cost, $p_i = a_i$, until the pool of funding F is fully exhausted, $\sum_{i \in I} p_i \leq F$. All sites to which payments are made avoid deforestation; all other sites are deforested. Thus, the agency's constrained optimization problem is: $\max_{p_i} C \;\; s.t. \sum_{i \in I} p_i \leq F$.

In the "service price payment framework," the agency does not differentiate payments across sites. This could occur because of informational, political, or equity constraints. Instead the agency pays a standard price per unit of ecosystem service supplied across all sites. The agency chooses a single price per unit of carbon, p_c, and a single price per unit of biodiversity, p_b. Forested sites are willing to participate if potential ecosystem service payments exceed their opportunity costs. That is, forested sites choose to avoid deforestation if $p_b b_i + p_c c_i - a_i \geq 0$ and choose to deforest otherwise. Total funding for carbon, F_c, is equal to the sum of carbon payments to sites that avoid deforestation: $F_c = \sum_{i \in I} p_c c_i$. Total funding for biodiversity, F_b, is equal to the sum of biodiversity payments to sites that avoid deforestation: $F_b = \sum_{i \in I} p_b b_i$. The agency is assumed to be able to set prices p_b and p_c such that the pool of funding for forest conservation F is fully exhausted: $F_b + F_c \leq F$. The portion of funding spent on biodiversity is represented as $\beta = \frac{F_b}{F_b + F_c} \in [0,1]$. Thus, the agency's constrained optimization problem is: $\max_{p_c, p_b} C \;\; s.t. \sum_{i \in I} p_b b_i + p_c c_i \leq F$.

The agency requires less information about sites' opportunity costs in the service price payment framework than in the opportunity cost payment framework, and can obtain this information more easily. The opportunity cost payment framework requires the agency to have information on all sites' opportunity costs because payments and site selection are based on this information. Sites have an incentive to overstate their true opportunity costs in order to obtain a higher payment, meaning that carefully designed auction mechanisms are necessary to reveal true opportunity costs (Stoneham et al., 2003; Ferraro 2007; Obersteiner et al., 2010). In contrast, the service price payment framework does not require any information on sites' opportunity cost, because payments are based only on services and sites self-select to participate. Only if the agency seeks to exactly exhaust a fund of fixed size must it have some information on sites' willingness to participate at different prices for at least a partial set of sites whose opportunity costs are near the payment margin. In the service price payment framework sites have no incentive to overstate their true willingness to accept payments if asked, since total payments are not linked to sites' opportunity costs but rather provide surplus revenue to nearly every participating site. This incentive for truthful revelation of opportunity costs in the service price payment framework allows the agency to use a broader set of potential information-revealing mechanisms. Potential mechanisms include simply asking potentially participating sites to state their willingness to accept, or toggling prices until payments to participating sites exactly exhaust the fund.

The possibility that the agency could sell emission reductions onward to developed countries through carbon offset markets or bilateral agreements is implied but not modeled. The agency's motivation to maximize carbon may be thought of as supported by combined external and internal demand for emission reductions, which may fall short of the socially optimal level of emission reductions. Similarly, biodiversity payments could also be supported by some level of external finance.

The paradox of paying for multiple ecosystem services is considered to occur if an additional biodiversity payment can produce carbon storage more cost-efficiently than an incremental carbon payment. That is, if $\frac{\Delta F_C}{\Delta C} > \frac{\Delta F_B}{\Delta C}$, or:

$$\frac{\Delta p_c \sum_{i\in I} c_i + p_b b_j + p_c c_j + \Delta p_c c_j}{c_j} > \frac{\Delta p_b \sum_{i\in I} b_i + p_b b_k + p_c c_k + \Delta p_b b_k}{c_k} \quad (1)$$

Where *j* is the next site to be incentivized to supply avoided deforestation after the original carbon price p_c is increased by increment Δp_c, and *k* is the next site to be incentivized to supply avoided deforestation after the original biodiversity price p_b is increased by increment Δp_b. When the carbon price is increased by Δp_c, $\Delta p_c \sum_{i\in I} c_i$ represents additional rent payments to existing suppliers of avoided deforestation while $p_b b_j + p_c c_j + \Delta p_c c_j$ represents the payment that incentivize the participation of an additional supplier. Likewise, when the biodiversity price is increased by Δp_b, $\Delta p_b \sum_{i\in I} b_i$ represents additional rent payments to existing suppliers of avoided deforestation while $p_b b_k + p_c c_k + \Delta p_b b_k$ represents the payment that incentivizes the participation of an additional supplier. Initial biodiversity payments can be zero, as in Figure 2, in which case $p_b = p_b b_j = p_b b_k = 0$.

In the case with no initial biodiversity payments, $p_b = 0$, $p_c = \frac{a_{\bar{\iota}}}{c_{\bar{\iota}}}$, $\Delta p_c = \frac{a_j}{c_j} - \frac{a_{\bar{\iota}}}{c_{\bar{\iota}}}$, and $\Delta p_b = \frac{a_k - p_c c_k}{b_k}$, where $\bar{\iota}$ is the last site to be incentivized to supply avoided deforestation at the original carbon price p_c. So the conditions under which the paradox occurs in inequality (1) can be expressed entirely in terms of site characteristics rather than service prices:

$$\left(\frac{a_j}{{c_j}^2} - \frac{a_{\bar{\iota}}}{c_{\bar{\iota}} c_j}\right)\left(\sum_{i \in I} c_i + c_j\right) > \left(\frac{a_k}{b_k c_k} - \frac{a_{\bar{\iota}}}{b_k c_{\bar{\iota}}}\right)\left(\sum_{i \in I} b_i + b_k\right)$$
(2)

That this paradox can occur is illustrated numerically in the next section.

The paradox of paying for multiple ecosystem services

First the two payment frameworks described in Section III were applied to a simple six-site landscape (Table 1). The opportunity costs, biodiversity values, and carbon values of the six sites were deliberately selected to simply and clearly illustrate the existence of the paradox of paying for multiple ecosystem services.

Next the payment frameworks described in Section III were applied to randomly generated data representing large landscapes. For illustrative purposes, the number of sites in the base landscape was set to n=10,000 and parameters a_i, b_i, and c_i were independent and randomly drawn from the following continuous distributions: $a_i \in$[0,\$1000], $b_i \in$ [0,100], $c_i \in$ [0,100]. Fund size was set to $= 0.5 \sum_i a_i \ \sim \$2{,}500{,}000$.

The production possibility frontier resulting from shifting funding from carbon to biodiversity in both payment frameworks was traced. In the opportunity cost payment framework, the extent to which the agency includes biodiversity along with carbon in its prioritization of sites, $\propto$, was varied along a continuum from 0 to 1. In the service price payment framework the portion of funding spent on biodiversity payments, β, was varied along a continuum from 0 to 1.

The base landscape was then altered to illustrate the sensitivity of the production possibility frontier for carbon and biodiversity to changes in three variables. First, the correlation between opportunity costs and biodiversity values across sites, r_{ab}, was varied between -0.5,0.0, and 0.5. Second, the correlation between opportunity costs and carbon values across sites, r_{ac}, was varied between -0.5, 0.0, and 0.5. Third, the total fund size, F, was varied from $0.25 \sum_i a_i$ to $0.5 \sum_i a_i$ to $0.75 \sum_i a_i$ to $0.99 \sum_i a_i$.

Finally the payment frameworks described in Section III were applied to data on deforestation, biodiversity values, carbon values, and opportunity costs from three actual landscapes (Table 2). The first data set was from Bolivia (Andersen et al 2012). Deforestation between 2001-2005 (Killeen et al., 2007) occurred within 32,522 of 120,476 forested 900 ha grid cells. Biodiversity values (spp/ha) were an index of species richness created from 17 taxa (Nowicki et al., 2004). Carbon values (tCO_2e/ha) were the carbon dioxide equivalent of the above- and belowground forest carbon per hectare (Ruesch and Gibbs, 2008). Opportunity costs were obtained by converting a global map of maximum potential gross annual agricultural revenues at 1995-2005 prices (Naidoo and Iwamura, 2007) to net present value assuming a 30-year time horizon, a 10% discount rate, and 85% costs, as used in the Stern Review (Grieg-Gran, 2006). For a critical discussion of the strengths and limitations of the Naidoo and Iwamura data set see for example Bode et al. (2008) and Kremen et al. (2008).

The second data set was from Indonesia (Busch et al., 2012). Deforestation between 2000-2005 (Hansen et al., 2008; Hansen et al., 2009) occurred within 86,988 of 195,466 900 ha grid cells. Biodiversity values (spp/ha) were the richness of forest mammal species for which highly suitability forest habitat was modeled to occur in each cell (Boitani et al., 2006). Carbon values (tCO_2e/ha) were the carbon dioxide equivalent of the above- and belowground forest carbon per hectare (Gibbs and Brown, 2007). Opportunity costs were obtained as described above.

The third data set was from Madagascar. Deforestation between 2000-2005 (MEFT 2009) occurred within 7,895 of 65,024 900 ha grid cells. Biodiversity values (spp/ha) were the richness of 92 lemur species from the IUCN Red List database (IUCN, 2012). Carbon values (tCO_2e/ha) were the carbon dioxide equivalent of the above- and belowground forest carbon per hectare (Ruesch and Gibbs, 2008). Opportunity costs were obtained as described above.

For each landscape the production possibility frontiers were traced under each payment framework. In the opportunity cost payment framework the extent to which the agency included biodiversity along with carbon in its prioritization of sites α, was shifted along a continuum from 0 to 1. In the service price payment framework the portion of funding spent on biodiversity payments, β, was shifted along a continuum from 0 to 1. Three funding levels were applied: $F = 0.25 \sum_i a_i$; $F = 0.5 \sum_i a_i$; $F = 0.75 \sum_i a_i$.

Results

In the simple six-site landscape, the greatest carbon value and biodiversity value was achieved under the opportunity cost payment framework (Table 1). Under the opportunity cost payment framework a fund of size F=\$120 obtained forest conservation from sites 1, 2, 3, 4, and 5, producing a landscape with total carbon value C=52 and total biodiversity value B=50. The entire \$120 fund was disbursed on payments incentivizing participation rather than payments providing additional rent to suppliers. Less carbon value and biodiversity was achieved under a service price payment framework. In the service price payment framework with carbon payments only, a \$120 fund with a carbon price of P_c=\$3 and a biodiversity price of P_b=\$0 obtained forest conservation from sites 1, 2, and 3, producing a landscape with total carbon value C=40 and total biodiversity value B=28. From the \$120 fund, \$70 was disbursed on payments incentivizing participation while \$50 was disbursed on payments providing additional rent to supplier. By reallocating some of the fixed funding from carbon payments to biodiversity payments under a service price payment framework, greater carbon value was achieved. A fund of size F=\$120 with a carbon price of P_c=\$2 and a biodiversity price of P_b=\$1 obtained forest conservation from sites 1, 2, and 4, producing a landscape with total carbon value C=41 and total biodiversity value B=38. From the \$120 fund, \$75 was disbursed on payments incentivizing participation, while \$45 was disbursed on payments providing additional rent to suppliers.

In the large landscapes, the opportunity cost payment framework always resulted in greater carbon storage and biodiversity than the service price payment framework at the same level of funding (Figures 4-6). Under the opportunity cost payment framework, production possibility frontiers always slope downward, implying a strict tradeoff between carbon and biodiversity (Figures 4-6). Shifting funding priority toward biodiversity would have always reduced absolute climate value by some amount, even if the increase to biodiversity value was large and the reduction to climate value was small. That is, $\frac{\partial C}{\partial \propto} < 0\ \forall \propto$. An agency concerned only with climate would have maximized climate benefits by prioritizing sites for inclusion based only on carbon value per opportunity cost, with no consideration of biodiversity value. That is, the priority placed on biodiversity in the funding decision at which climate benefits were maximized was always equal to zero: $\alpha^*=0$. This is consistent with intuition and the findings of previous literature.

However, under the service price payment framework, some production possibility frontiers bubble outward, indicating that there may not be a tradeoff between biodiversity and carbon (Figures 4-6). Conditions exist such that reallocating a portion of fixed funding from carbon payments to biodiversity payments would have resulted in both greater carbon benefits and greater biodiversity benefits. That is, $\frac{\partial C}{\partial \beta} > 0 \text{ for some } \beta$. An agency concerned only with climate benefits could have maximized carbon storage by spending across a mixture of carbon payments and biodiversity payments. That is, the portion of funding spent on biodiversity payments at which carbon storage was maximized was greater than zero in some cases: $\beta^*>0$.

The portion of funding spent on biodiversity payments at which carbon storage was maximized, β^*, or at which biodiversity was maximized without decreasing carbon storage relative to the carbon-payments-only scenario, β', was greater if biodiversity values were more correlated with opportunity costs or if carbon values were less correlated with opportunity costs (Table 3; Figure 4). β^* and β' were positive in every case except the case in which carbon values were highly correlated with opportunity costs (r_{ac}=0.5) and biodiversity values were highly negatively correlated with opportunity costs (r_{ab}=-0.5) (Figure 4g). β^* and β' were greatest (β^*=0.47; β'=0.86) when biodiversity values were highly correlated with opportunity costs (r_{ab}=0.5) and carbon values were highly negatively correlated with opportunity costs (r_{ac}=-0.5) (Figure 4c). β* and β' were also greater at greater fund size (Table 3; Figure 5).

The paradox occurred in all three of the landscapes examined (Table 4; Figure 6). In Bolivia, the correlation between biodiversity values and opportunity costs was smaller than the correlation between carbon values and opportunity costs (r_{ab}=0.11 and r_{ac}=0.27). In this landscape the paradox was modest (β*=0.01-0.05 and β'=0.01-0.08, depending on the level of funding).

In Madagascar and Indonesia, the correlation between biodiversity values and opportunity costs was larger than or comparable to the correlation between carbon values and opportunity costs (r_{ab}=-0.02 and r_{ac}=-0.20 in Indonesia; r_{ab}=-0.32 and r_{ac}=-0.24 in Madagascar). In these landscapes the paradox was more pronounced (β*=0.15-0.69 and β'=0.25-0.97 in Indonesia; β*=0.14-0.31 and β'=0.30-0.52 in Madagascar, depending on the level of funding).

When biodiversity values and opportunity costs were correlated, payments for biodiversity improved the overall efficiency of the program by operating more like an opportunity cost-based payment without explicit price discrimination. But the paradox still occurred even when biodiversity values were uncorrelated with opportunity costs, since suppliers of incremental units of carbon were incentivized to participate more cheaply with biodiversity payments than by raising the carbon price for all suppliers. Greater funding had a noisier effect on the paradox in the four real landscapes than in the randomly generated landscapes. As the funding for payments increased, the set of sites participating in avoiding deforestation grew, and the relative correlations of carbon values and biodiversity values with opportunity costs changed as well.

Discussion

This paper has shown using a simple numerical model and data from three landscapes that if REDD+ payments are based on a standard price per unit of service provided, then paradoxically it may be possible to shift a portion of funding away from carbon payments and towards biodiversity payments and obtain both more biodiversity benefits and more climate benefits. Put differently, money spent on biodiversity payments could in some cases indirectly produce greater climate benefits than money spent directly on carbon payments. This paradox is more pronounced when biodiversity values are more correlated with opportunity costs across a landscape, when carbon values are less correlated with opportunity costs across a landscape, or as funding available for payments increases.

Note that the conditions under which this paradox occurs are not an anomalous quirk. Rather, the analytical framework presented here is in fact far more descriptive of emerging REDD+ payment mechanisms than the conservation-planning framework of site selection used in previous literature, which assumed that any land could be acquired at its opportunity cost. Informational, institutional, and equity barriers exist to the implementation of opportunity-cost based payments, while all emerging REDD+ programs use service price based payments.

Further research can show the extent to which this paradox extends beyond deforestation to reforestation or sustainable forest management. With deforestation the land cover decision is binary—maintain existing forest cover along with the more-or-less fixed array of services it provides, or clear the land for agriculture. With reforestation or sustainable forest management, land management decisions such as plantation type, rotation length, and input management can affect the production functions for biodiversity and carbon at the site and landscape scale (van Noordwijk, 2002) in addition to land cover.

This paradox extends beyond REDD+ and tropical deforestation, and beyond carbon and biodiversity, to any payment program for multiple ecosystem services, such as the United States Department of Agriculture's Conservation Reserve Program (Cattaneo et al., 2006), Costa Rica's Payment for Environmental Services program (Sanchez-Azofeifa et al., 2007), or Ecuador's *SocioBosque* program (de Kooning et al., 2011). Extending the scope of payments to include clean water provision (e.g. Munoz-Pina et al., 2007), soil stabilization, or other ecosystem services holds the potential to further increase the values of services provided.

Programs that pay for multiple ecosystem services may not only optimize across a mixture of multiple services, but may even maximize the provision of single services. The implication of this finding is that even governments concerned primarily with the supply of single services should construct their programs to enable supplemental financing of secondary services. Even if biodiversity payments were always less effective than carbon payments in achieving climate goals, policymakers concerned primarily with climate should construct REDD+ programs to enable additional finance from supplemental payments for biodiversity. But since biodiversity payments can in some circumstances achieve climate goals more effectively than equivalent funding spent on carbon payments, even policymakers concerned primarily with climate may want to consider diversifying the services for which a program makes payments across a landscape to include some amount of biodiversity payments. Of course, if policymakers are concerned with both climate and biodiversity, an even larger portion of funding spent on biodiversity payments could be justified.

What is the practical likelihood that REDD+ funding would be diversified from carbon payments to include biodiversity payments? At the international level, it appears improbable that a UNFCCC REDD+ mechanism would explicitly set aside carbon finance for biodiversity payments, even though in theory doing so has the potential to increase climate benefits. UNFCCC rules on REDD+ are quite advanced and focused on carbon, referring to biodiversity only in a safeguard (UNFCCC, 2010). However, the Parties to the UNFCCC should at the very least be interested in enabling supplementary biodiversity payments to the extent that such payments can bring additional funding for the conservation of forests without reducing the funding or price paid for carbon. Individual forest countries will have greater flexibility in designing national REDD+ programs, and will likely be doing so with a set of objectives that are broader than just climate and revenue. So, national REDD+ decision makers should consider the possibility that diversifying the spending of climate funding to include biodiversity payments could possibly advance climate goals in addition to biodiversity goals. Multilateral facilities might also consider supplementing REDD+ carbon payments with biodiversity payments. At the time of writing the Carbon Fund of the Forest Carbon Partnership Facility is planning to negotiate with sellers and buyers of emission reductions on whether and how non-carbon benefits should be taken into consideration in pricing (FCPF, 2012).

Finance for supplemental biodiversity payments could come from either of two sources of willingness-to-pay—premium payments from existing buyers of REDD+ emission reductions, or matching payments from a distinct pool of potential buyers interested primarily in the conservation of biodiversity. Existing buyers of REDD+ emission reductions may be willing to pay a price premium for reductions known to originate from more biodiverse forests. Evidence of increased willingness to pay for high-biodiversity forest carbon has been found by a survey of buyers in the voluntary offset market (Neeff et al., 2009) and by a study of the influence of biodiversity value on the geographic location of REDD+ demonstration activities (Cerbu et al., 2010), although a study of voluntary carbon market transactions did not identify a price premium associated with proxies for forest carbon projects' biodiversity benefits (Conte and Kotchen, 2010). But even if a considerable willingness to pay a price premium does exist on the part of certain public or private buyers of REDD+, the magnitude of finance available through price premiums alone would be fundamentally constrained by the scale of demand for emission reductions. Greater finance could be obtained by enabling the emergence of a pool of biodiversity buyers distinct from carbon buyers to match carbon payments with supplementary biodiversity payments. The World Bank's recently announced Wildlife Premium Market Initiative (Zoellick, 2010) suggests the possible emergence of such a pool.

The incremental costs associated with conserving high-biodiversity forest could be financed by supplemental biodiversity payments in one of three ways. First, additional upfront financial support could be invested in reducing deforestation in especially biodiverse forests, with countries or communities selling the resulting emission reductions through the REDD+ mechanism. Second, biodiversity buyers could purchase emission reductions from especially biodiverse forests above the market price, and then sell these reductions back into the REDD+ mechanism at the market price. Third, biodiversity buyers could pay a biodiversity payment at the time when countries or communities sell emission reductions into the REDD+ mechanism from especially biodiverse forests.

Three additional global institutional investments would be useful to implement any of these supplemental biodiversity finance methods. First, a registry identifying the geographical origin of emission reductions would allow potential buyers to assess which reductions are from forests valuable for biodiversity. Such a registry of geographic origin may be an important feature of international or national REDD+ programs even in the absence of supplemental biodiversity payments. Second, supplemental biodiversity payments would benefit from a facility to consolidate demand for the biodiversity benefit of avoided deforestation from many small and geographically dispersed potential buyers. Third, a standardized, accepted system for geographically differentiating forests' relative biodiversity value would relieve buyers of the private cost of gathering and analyzing this information.

This analysis has considered only one illustrative metric of a forest's value for biodiversity based on the species richness of particular taxa. But other metrics of sites' relative biodiversity value are certainly possible, including those based on threat, habitat connectivity, ratio of indigenous to exotic species (Carswell and Burrows, 2006), range size rarity, or the presence of charismatic flagship species. Importantly, a site's biodiversity value for the purposes of supplemental payments should be independent of whether or not other sites remain forested elsewhere, ruling out metrics of biodiversity value based on complementarity or substitutability commonly employed in systematic conservation planning (Sarkar et al., 2006). Arriving at appropriate and accurate metrics for biodiversity value should result from a transparent and science-based process, but need not be the burden of the UNFCCC. It is worth noting the limitations on the extent to which threatened species could be conserved through supplemental biodiversity payments to the REDD+ mechanism. Supplemental biodiversity payments are unlikely to support the conservation of non-forest species, or forest species for which habitat loss is not a main driver of threat (Collins et al., 2011), or for which management is required in addition to habitat conservation. Furthermore, some species may not be sufficiently charismatic to garner a biodiversity price premium on their own (Collins et al., 2011).

Recoverable international finance for supplemental biodiversity payments is potentially considerable. Historically, payments for biodiversity have comprised a large share of international willingness-to-pay for forest conservation in the form of conservation projects and bilateral and multilateral budget support for national protected area networks. Estimates of current international expenditure on biodiversity conservation have ranged from $1.5 billion annually (Halpern et al., 2006) to $3.5 billion annually (Castro and Hammond, 2009) up to $5 billion annually (Gutman and Davidson, 2008). Funding for forest biodiversity conservation might well increase further if a REDD+ mechanism establishes an efficient forest conservation payment vehicle that could be leveraged by supplemental biodiversity payments.

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Table 1: Carbon and biodiversity values supplied by a six-site landscape under alternative payment frameworks. *a* represents the opportunity cost of forest conservation at a site. *b* represents the biodiversity value provided by a site. *c* represents the carbon value provided by a site. *P*? represents participation. *f* represents the payment made to a site.

Site characteristics						Opportunity cost payment framework				Service price payment framework: carbon payments only				Service price payment framework: mixture of carbon and biodiversity payments			
Carbon payment price						n.a.				$3				$2			
Biodiversity payment price						n.a.				$0				$1			
site	*a*	*b*	*c*	*a/c*	*a/b*	*P*?	*b*	*c*	*f*	*P*?	*b*	*c*	*f*	*P*?	*b*	*c*	*f*
1	$15	10	15	$1	$3	yes	10	15	$15	yes	10	15	$45	yes	10	15	$40
2	$40	16	20	$2	$5	yes	16	20	$40	yes	16	20	$60	yes	16	20	$56
3	$15	2	5	$3	$15	yes	2	5	$15	yes	2	5	$15	no	0	0	-
4	$20	12	6	$3.3	$3.3	yes	12	6	$20	no	0	0	-	yes	12	6	$24
5	$30	10	6	$5	$6	yes	10	6	$30	no	0	0	-	no	0	0	-
6	$36	4	6	$6	$18	no	0	0	-	no	0	0	-	no	0	0	-
Total							50	52	$120		28	40	$120		38	41	$120
Incentive payments									$120				$70				$75
Rent payments									$0				$50				$45

Table 2: Data from three landscapes: summary statistics

Landscape	Data	Source	Minimum	Median	Mean	Maximum
Bolivia	Deforestation (ha/900 ha)	Killeen et al. 2007	0	9	40	870
n=32,522	Opportunity Cost ($/ha)	Naidoo and Iwamura, 2008	0	1,116	1,091	2,311
	Biodiversity (17 taxa spp./ha)	Nowicki et al. 2004	0	487	617	2,825
	Carbon (tCO_2e/ha)	Ruesch and Gibbs, 2008	0	412	399	708
Indonesia	Deforestation (ha/900 ha)	Hansen et al. 2008, 2009	0	6	40	2,230
n=86,988	Opportunity Cost ($/ha)	Naidoo and Iwamura, 2008	0	205	304	18,783
	Biodiversity (mammal spp./ha)	Catullo et al. 2008	0	41	58	131
	Carbon (tCO_2e/ha)	Gibbs and Brown, 2007	0	688	611	826
Madagascar	Deforestation (ha/900ha)	MEFT, 2009	25	50	69	650
n=7,895	Opportunity Cost ($/ha)	Naidoo and Iwamura, 2008	0	331	552	2,565
	Biodiversity (lemur spp./ha)	IUCN, 2011	0	6	6	15
	Carbon (tCO_2e/ha)	Ruesch and Gibbs, 2008	0	264	353	734

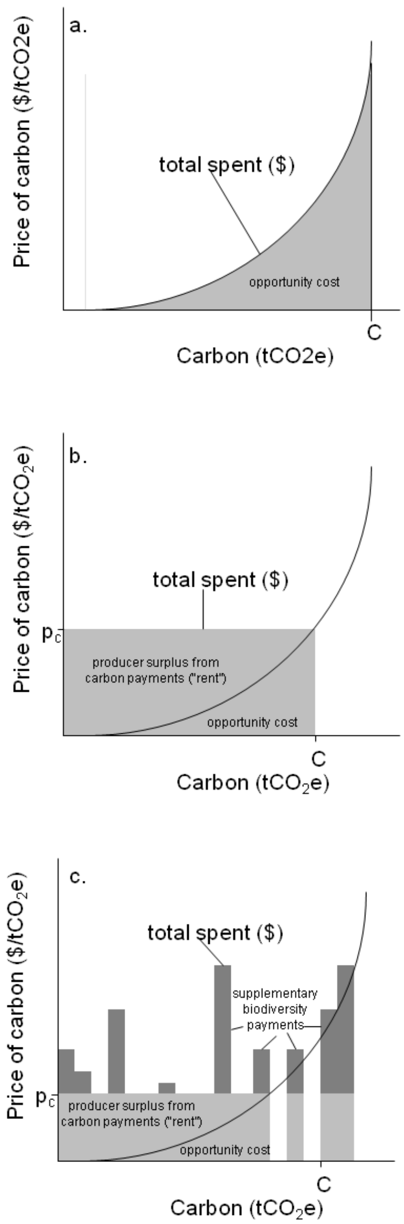
Table 3: Carbon and biodiversity values supplied by randomly generated landscapes under alternative payment frameworks. β represents the portion of total funding spent on biodiversity payments. r_{ab} represents the correlation between opportunity costs and biodiversity values across the landscape. r_{ac} represents the correlation between opportunity costs and carbon values across the landscape. F represents the fund size (thousand USD). B represents the biodiversity value provided by the landscape (thousand spp-ha). C represents the carbon value provided by the landscape (thousand tCO_2e). p_b represents the biodiversity payment price ($/spp-ha). p_c represents the carbon payment price ($/$tCO_2e$). F_b represents the total funding spent on biodiversity payments (thousand USD). F_c represents the total funding spent on carbon payments (thousand USD). β* represents the portion of total funding spent on biodiversity payments at which the supply of carbon is maximized.

Site characteristics			Opportunity cost payment framework		Services price payment framework: carbon payments only (β=0)						Service price payment framework: mixture of carbon and biodiversity payments (β=β*)						
r_{ab}	r_{ac}	F	B	C	p_b	p_c	F_b	F_c	B	C	□□	p_b	p_c	F_b	F_c	B	C
-0.5	-0.5	2516	332	386	0	8.80	0	2516	238	286	0.09	0.89	8.04	218	2299	245	286
0.0	-0.5	2508	316	387	0	8.77	0	2507	216	286	0.19	1.94	7.07	465	2042	240	289
0.5	-0.5	2489	297	386	0	8.71	0	2489	193	286	0.47	5.07	4.37	1174	1314	231	301
-0.5	0.0	2506	301	390	0	8.51	0	2506	228	294	0.03	0.29	8.28	66	2440	231	295
0.0	0.0	2498	311	405	0	8.71	0	2498	215	287	0.01	0.10	8.63	23	2475	216	287
0.5	0.0	2511	271	395	0	8.41	0	2511	183	299	0.29	3.36	5.85	717	1795	214	307
-0.5	0.5	2513	323	333	0	9.49	0	2513	269	265	0.00	0.00	9.49	0	2513	269	265
0.0	0.5	2502	284	331	0	9.49	0	2502	224	264	0.01	0.15	9.35	33	2469	228	264
0.5	0.5	2503	248	331	0	9.54	0	2503	183	262	0.17	2.00	7.80	416	2087	208	268
0.0	0.0	1250	217	289	0	6.12	0	1250	154	204	0.02	0.16	5.99	25	1224	156	204
0.0	0.0	3747	406	474	0	10.68	0	3747	263	351	0.01	0.18	10.54	47	3699	265	351
0.0	0.0	4946	496	496	0	12.61	0	4945	298	392	0.08	1.18	11.63	372	4574	314	393

Table 4: Carbon and biodiversity values supplied by three landscapes under alternative payment frameworks β represents the portion of total funding spent on biodiversity payments. r_{ab} represents the correlation between opportunity costs and biodiversity values across the landscape. r_{ac} represents the correlation between opportunity costs and carbon values across the landscape. F represents the fund size (million USD). B represents the biodiversity value provided by the landscape (million spp-ha). C represents the carbon value provided by the landscape (million tCO_2e). p_b represents the biodiversity payment price ($/spp-ha). p_c represents the carbon payment price ($/$tCO_2e$). F_b represents the total funding spent on biodiversity payments (million USD). F_c represents the total funding spent on carbon payments (million USD). β* represents the portion of total funding spent on biodiversity payments at which the supply of carbon is maximized.

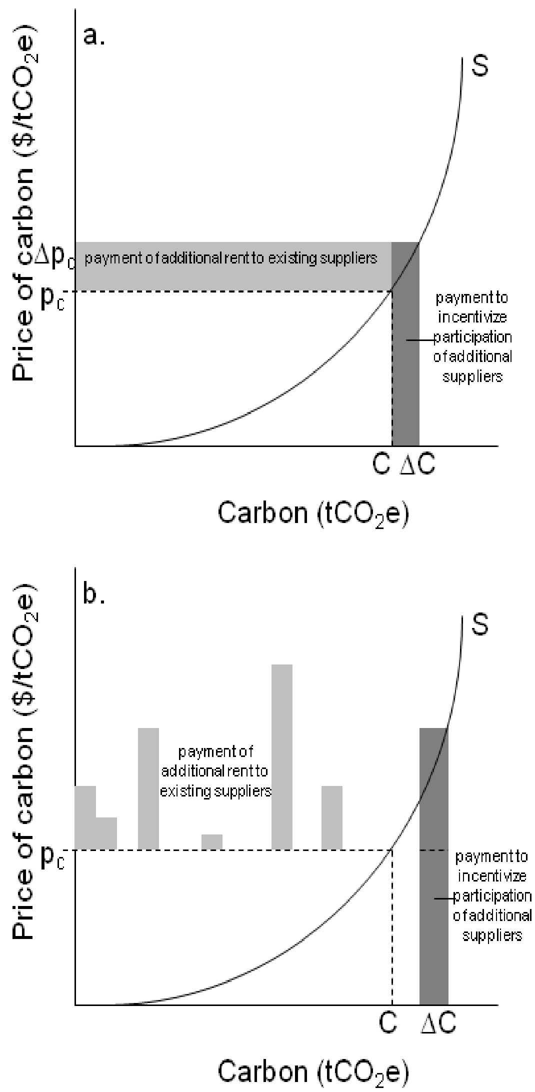
Site characteristics			Opportunity cost payment framework			Services price payment framework: carbon payments only (β=0)						Service price payment framework: mixture of carbon and biodiversity payments (β=β*)							
Country	r_{ab}	r_{ac}	F	B	C	p_b	p_c	F_b	F_c	B	C	□□	p_b	p_c	F_b	F_c	B	C
Bolivia	0.11	0.27	399	311	223	0	2.2	0	399	249	176	0.05	0.07	2.1	20	379	276	178
Bolivia	0.11	0.27	797	512	370	0	2.6	0	797	408	297	0.00	0.00	2.6	0.6	797	408	297
Bolivia	0.11	0.27	1,196	703	478	0	3.1	0	1196	530	379	0.03	0.06	3.0	34	1163	540	380
Indonesia	0.14	-	318	101	1287	0	0.3	0	318	69	823	0.69	2.41	0.1	218	100	90	911
Indonesia	0.14	-	637	161	1793	0	0.4	0	637	108	1363	0.15	0.83	0.3	95	542	115	137
Indonesia	0.14	-	955	197	2061	0	0.6	0	955	129	1554	0.20	1.36	0.4	191	764	141	159
Madagascar	-	-	71	2.55	148	0	0.6	0	71	1.77	112	0.14	5.32	0.5	10	61	1.85	113
Madagascar	-	-	142	3.19	181	0	1.0	0	142	2.24	136	0.31	18.2	0.7	44	98	2.44	137
Madagascar	-	-	214	3.55	197	0	1.4	0	214	2.56	149	0.27	21.2	1.0	57	156	2.69	151

Figure 1: Payments, opportunity cost, and supplier surplus, with fixed fund size



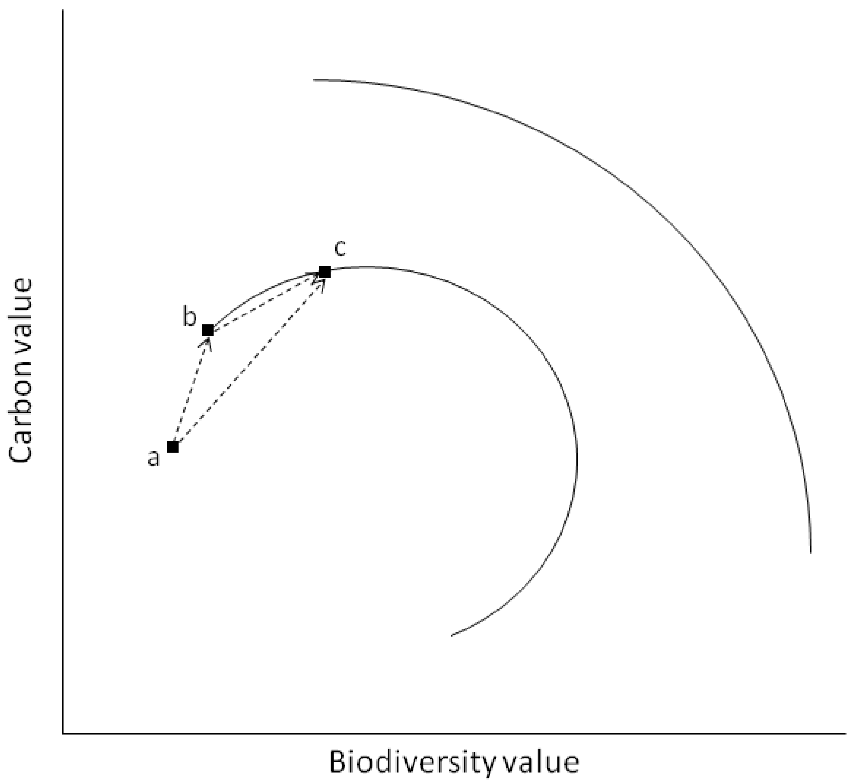
a. opportunity cost framework. b. service price framework with carbon payments only. c. service price framework with a mixture of carbon and biodiversity payments.

Figure 2. Carbon payments, biodiversity payments, and the incremental unit of carbon



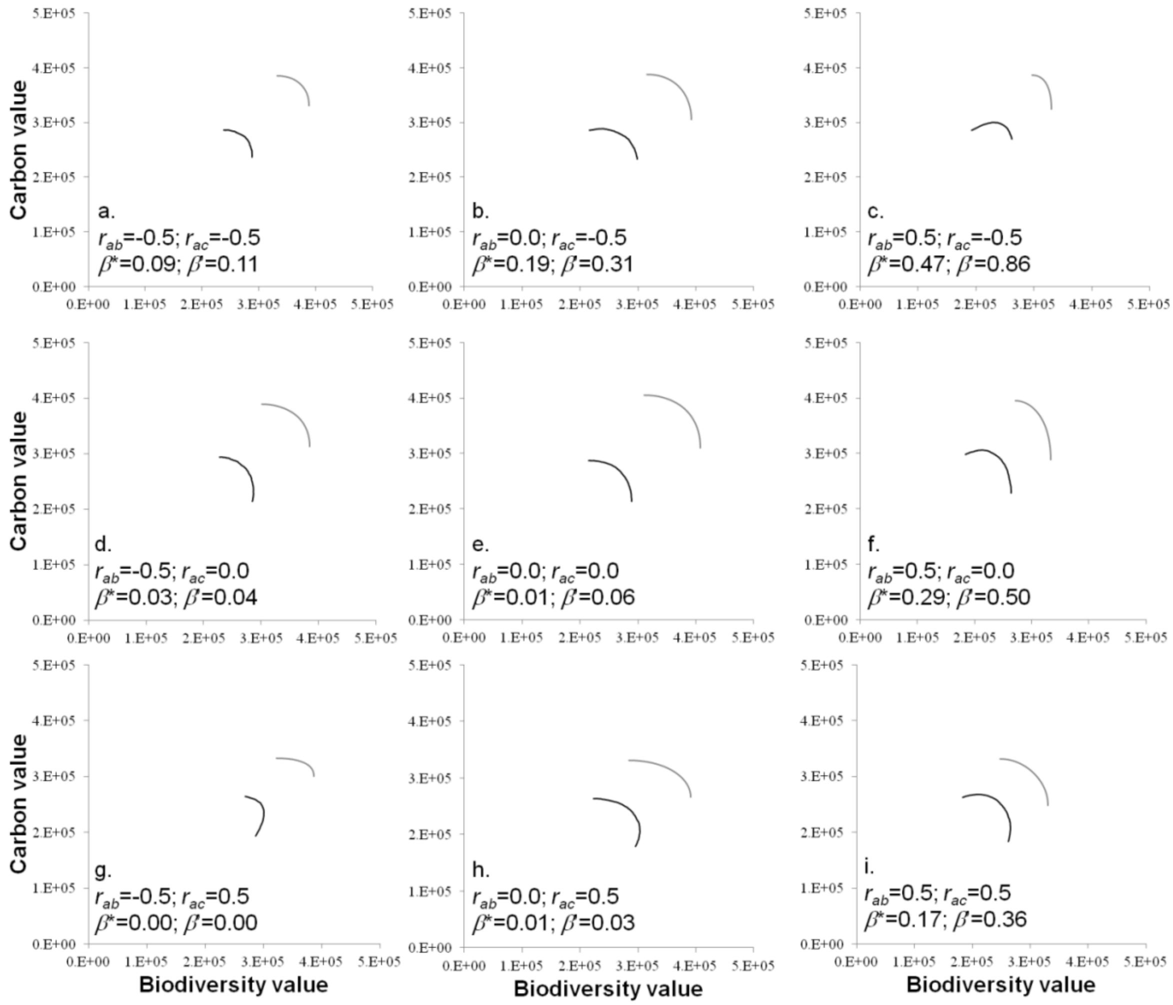
a. service price framework with carbon payments only. b. service price framework with a mixture of carbon and biodiversity payments.

Figure 3. Production possibility frontiers (PPFs) for carbon and biodiversity at a fixed level of finance under two alternative payment frameworks.



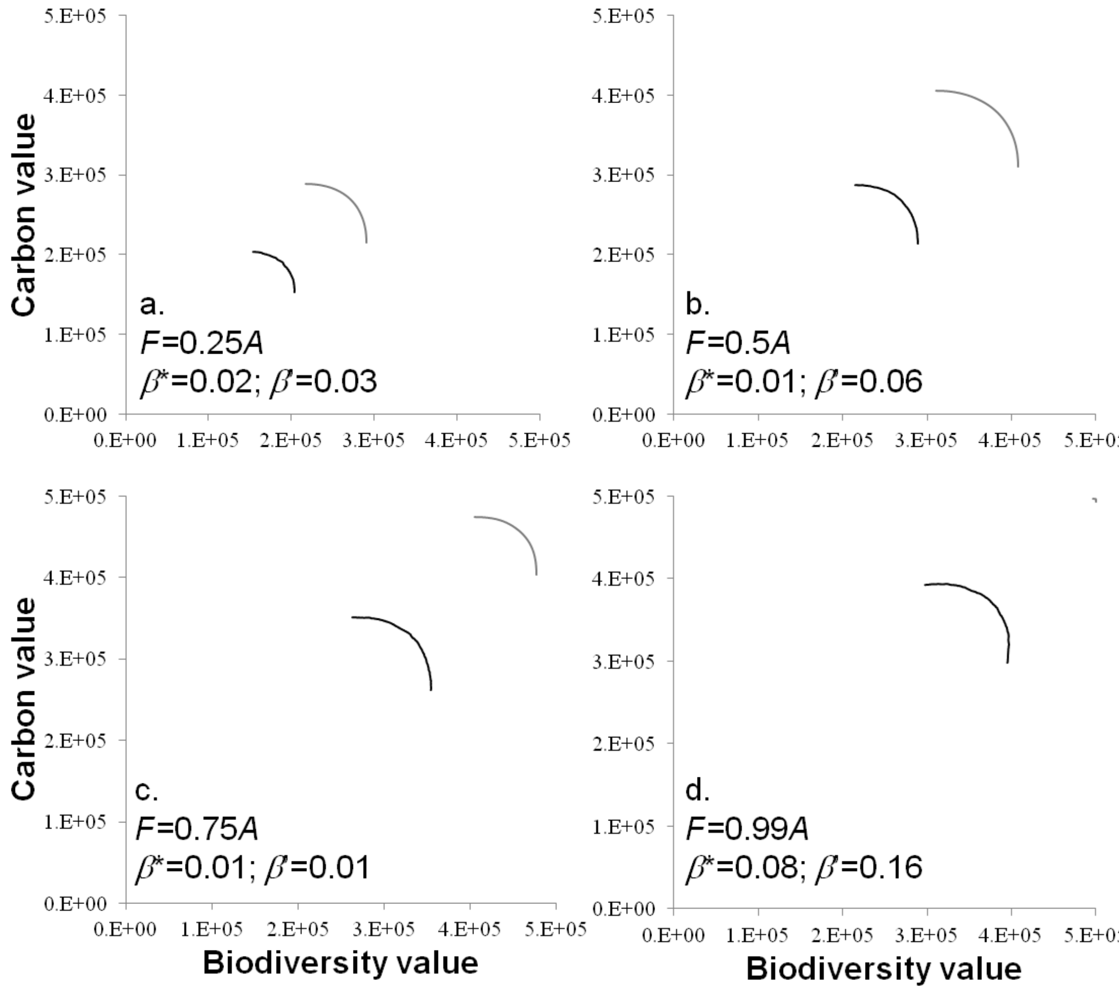
The outer curve represents the PPF under the opportunity cost payment framework. The inner curve represents the PPF under the service price payment framework at the same level of finance. Point a represents a lower level of finance than under the two PPFs. Point b represents a service price payment framework with carbon payments only. Point c represents a service price payment framework with a mixture of carbon payments and biodiversity payments. Vectors represent movements between points.

Figure 4: Production possibility frontiers (PPFs) for carbon and biodiversity under two payment frameworks with alternative correlation of variables



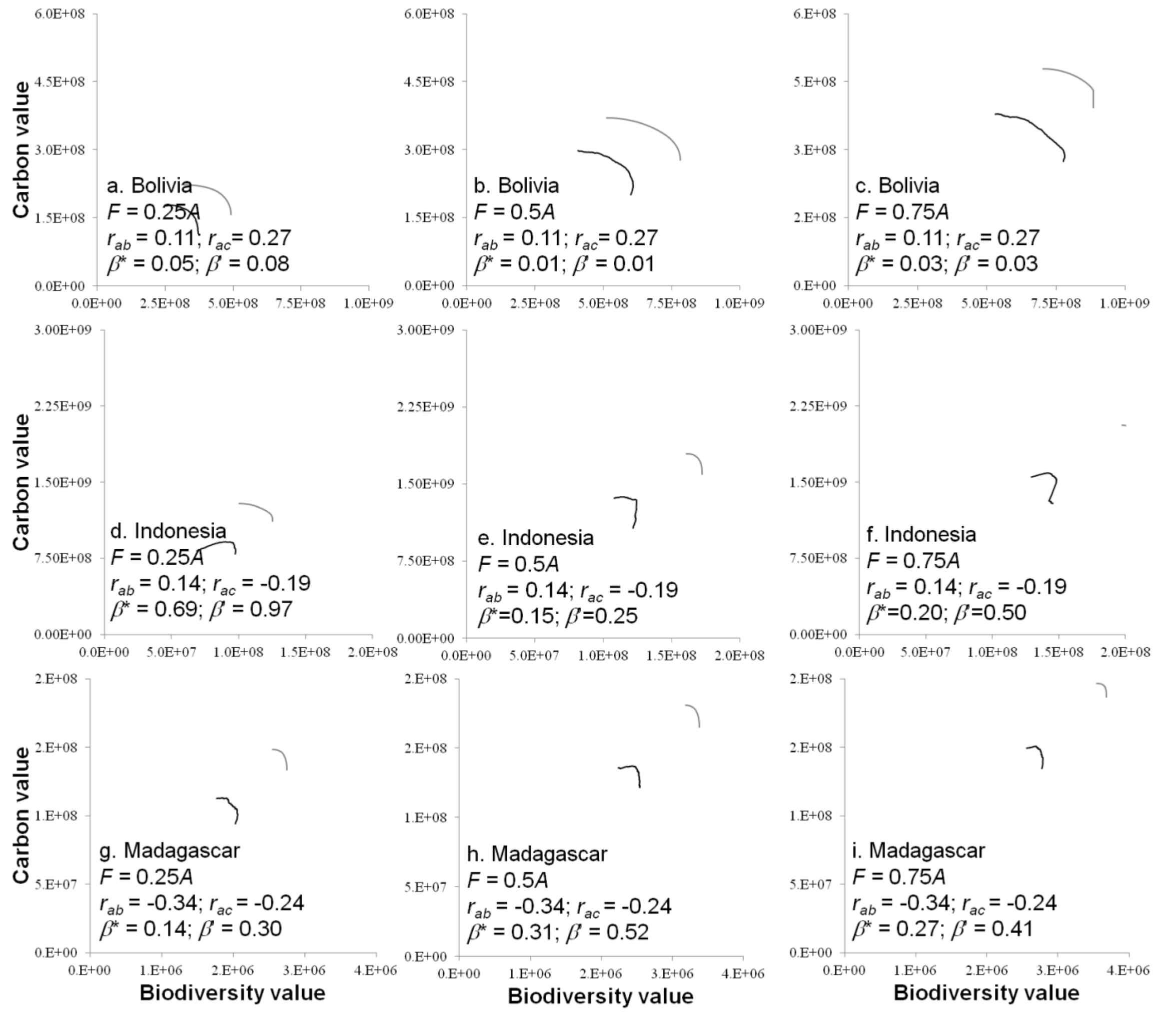
Outer curves represent the PPFs under the opportunity cost payment framework. Inner curves represent the PPFs under the service price payment framework at the same level of finance. r_{ab} represents the correlation between opportunity costs and biodiversity values. r_{ac} represents the correlation between opportunity costs and carbon values. β^* represents the portion of total funding spent on biodiversity at which the supply of carbon is maximized. β' represents the portion of total funding spent on biodiversity at which the supply of biodiversity is maximized while not diminishing the production of carbon.

Figure 5: Production possibility frontiers for carbon and biodiversity under two payment frameworks at alternative levels of finance



Outer curves represent the PPFs under the opportunity cost payment framework. Inner curves represent the PPFs under the service price payment framework at the same level of finance. *F* represents the fund size as a fraction of *A*, the sum of opportunity costs across the entire landscape. β * represents the portion of total funding spent on biodiversity at which the supply of carbon is maximized. β *'* represents the portion of total funding spent on biodiversity at which the supply of biodiversity is maximized while not diminishing the supply of carbon.

Figure 6: Production possibility frontiers for carbon and biodiversity under two payment frameworks across three landscapes



Outer curves represent the PPFs under the opportunity cost payment framework. Inner curves represent the PPFs under the service price payment framework at the same level of finance. *F* represents the fund size as a fraction of *A*, the sum of opportunity costs across the entire landscape. r_{ab} represents the correlation between opportunity costs and biodiversity values. r_{ac} represents the correlation between opportunity costs and carbon values. β^* represents the portion of total funding spent on biodiversity at which the supply of carbon is maximized. β' represents the portion of total funding spent on biodiversity at which the supply of biodiversity is maximized while not diminishing the supply of carbon.