Maximizing the Environmental Benefits of Carbon Farming through Ecosystem Service Delivery

BRENDA B. LIN, SARINA MACFADYEN, ANNA R. RENWICK, SAUL A. CUNNINGHAM, AND NANCY A. SCHELLHORN

The international carbon market provides a unique opportunity to increase ecosystem services and biodiversity through the revegetation of agricultural landscapes. Although the primary motivation for revegetation is to increase carbon sequestration, revegetated areas can provide additional financial, social, and environmental cobenefits that provide different levels of private and public net benefit. Conversely, carbon farming, if it is not implemented carefully, can create disbenefits, such as increased land clearing, monoculture plantations replacing diverse remnants, and unintended impacts across national borders. Economic models of carbon revegetation show that policies aimed at maximizing carbon sequestration alone will not necessarily lead to high uptake or maximize cobenefits. Careful consideration of policy incentives that encourage carbon plantings to deliver both public and private cobenefits is required, and solutions will need to balance both objectives in order to incentivize the sustainable, long-term management of carbon plantings across the landscape.

Keywords: biodiversity, cobenefits, disbenefits, land-use change, revegetation, tree planting

The series of carbon markets and offset schemes introduced internationally represent an opportunity to improve biodiversity outcomes through the revegetation of previously cleared agricultural land. Agricultural land in many parts of the world are being pressed to provide more environmental and economic services, while at the same time, their capacity to provide even the most basic provisioning services under potential climate change is in question (Schoeneberger et al. 2012). Land-use change and forestry policies have taken on an increasingly important role as a result of the negotiations of the United Nations (UN) Reducing Emissions from Deforestation and Forest Degradation (REDD+) Programme, part of the Clean Development Mechanism included in the Kyoto Protocol. Although avoiding deforestation and forest degradation is one of the main focuses of REDD+, the UN emphasizes that REDD+ goes beyond deforestation and forest degradation and includes the role of conservation, the sustainable management of forests, and the enhancement of forest carbon stocks (UN-REDD 2009). In this program, a carbon market was created in which corporations can buy carbon from farmers and land managers who sequester and store carbon above and below ground and thereby offset their emissions. The carbon-emitting corporations make payments to reward land managers for taking action to protect and plant trees or otherwise increase the carbon content of soils and vegetation on the land they manage (UN-REDD 2009). European countries have established a cap-and-trade system to create a carbon price, and a variety of voluntary state or regional strategies for emissions reductions have been developed elsewhere, including the Regional Greenhouse Gas Initiative in the 10 Northeastern states of the United States (Fahey et al. 2010). Many countries are now considering policies to allow landholders to access the carbon market through voluntary changes to land use. Whether these programs will lead to significant land-use changes will depend on the carbon price relative to the cost of other land-use or agricultural-production possibilities, as well as on the recognition and reward of the benefits that come from land-use change associated with carbon sequestration. Although carbon-accounting methodologies for carbon sequestration projects have been developed, there is often uncertainty about the likely rates of carbon sequestration on a specific site (Ritson and Sochacki 2003) and about the institutional mechanisms that are required to monitor and broker carbon stocks at the landscape level (Perez et al. 2007). In short, much research is still needed in order to understand how to successfully increase carbon sequestration across the landscape (Harper et al. 2007).

In Australia, the Carbon Farming Initiative (CFI) has been developed as a carbon-offset scheme established by the Australian government to provide farmers and other

BioScience 63: 793–803. ISSN 0006-3568, electronic ISSN 1525-3244. doi:10.1525/bio.2013.63.10.6

land managers access to international carbon markets (for a policy description, see Bradshaw et al. 2013). Carbon credits (measured in metric tons of carbon dioxide equivalents [CO_2^{-e}], with each credit representing 1 ton of CO_2^{-e}) can be earned by storing carbon (e.g., tree planting) or reducing greenhouse gas emissions (e.g., methane capture from landfill or livestock manure) on their land and can then be sold to people and businesses wanting to offset their own carbon emissions. Carbon is removed from the atmosphere through sequestering carbon in plants and increasing organic matter in soil or through avoiding losses of vegetation or organic matter in soils. All of the projects have a permanence commitment of 100 years (based on the estimated life of 1 ton of carbon pollution in the atmosphere) to discourage the rapid return of carbon back into the atmosphere, which would reverse the benefits of sequestration. A price on carbon is expected to generate demand for carbon-offset schemes, but the economic returns on carbon plantings are highly variable (Crossman et al. 2011). Nevertheless, the CFI presents an opportunity to sequester carbon across millions of hectares (ha) of cleared farmland and rangelands (Harper et al. 2007), and incentives to create additional benefits to meet conservation and restoration objectives should be considered in the development of carbon plantings (Bekessy and Wintle 2008).

Carbon farming and the international increase in voluntary carbon markets have been criticized for their lack of contribution to sustainable development—in particular, in terms of providing cobenefits to local communities (Chhatre and Agrawal 2009). Here, we define *cobenefit* as a benefit that is in addition to the intended benefits and a *benefit* as a measurable improvement that is seen by stakeholders to be positive and worthwhile (London Councils 2011). We suggest that the implementation of carbon farming as an offset to emissions is more likely to succeed if these schemes embrace the broader goal of providing sustainability outcomes that benefit the local community in multiple ways. These cobenefits include increased farm income diversity, along with less economically tangible benefits such as reduced agricultural externalities, improved ecosystem services to agriculture, and improved biodiversity outcomes, each of which would increase the overall value of carbon farming (Corbera and Brown 2008).

Therefore, carbon-farming initiatives present an opportunity to improve environmental quality through the careful consideration of methods to revegetate agricultural landscapes and to gain private and public benefits associated with biodiversity conservation and ecosystem service delivery. Although a range of potential carbon-offset methodologies are currently being considered around the world (e.g., soil management techniques such as no-till farming, changes to pasture and crop rotations, livestock management), we focus here on the revegetation of agricultural land using woody tree species, because most carbon sequestration offsets are expected to be generated through tree plantings (Crossman et al. 2011). We present a short review of the potential changes to land use in agricultural landscapes under voluntary carbon-farming markets, the potential benefits of this land-use change for biodiversity conservation and ecosystem service delivery for both public and private benefit, and the barriers and challenges to implementing revegetation strategies that increase the production of cobenefits across the agricultural landscape. The examples discussed in this article are not the only ways in which land use can change the carbon cycle, but the only pathway we consider here is carbon sequestration through tree planting. We also consider the effects that new policy incentives may have on overall landscape carbon sequestration and the other noncarbon benefits that can be gained from tree planting. Most carbon investments have been in monocultural plantations of trees that offer a rapid return on investments but relatively little compositional and structural diversity (Bekessy and Wintle 2008). Our goal in this review is to consider alternative methods in which increased biodiversity and structural complexity can be achieved in carbon tree plantings in order to provide environmental cobenefits across the landscape, in addition to the carbon sequestration benefits that are the fundamental drivers for change. This approach highlights how these various land-use trade-offs come together in the context of carbon farming. We focus on the biophysical aspects of this issue because the specific policy instruments required for implementation are understandably broad and outside the scope of this article. Highlighting a range of revegetation strategies, we present a framework that could be adopted in carbon-farming revegetation schemes to generate cobenefits across the landscape. The land-use changes that one would adopt to sequester carbon (as the only benefit) are not the same as what one would adopt to gain cobenefits in addition to carbon sequestration. We must examine the range of land uses that allow cobenefits to be gained while carbon sequestration remains the main goal. We suggest that, if cobenefits are not pursued in the carbon sequestration landscape, the outcomes will be adverse. In addition, by taking into account the increased cobenefits derived from revegetation with diverse composition and age structures, carbon-farming initiatives will have greater rates of adoption and higher levels of success.

Potential land-use change impacts under the new initiatives

Historically, the conversion of natural ecosystems to agricultural systems and the intensification of management have been linked to the loss of biodiversity-rich habitats, resulting in highly disturbed and fragmented ecosystems (Tilman et al. 2001) and a reduction in biodiversity and ecological function. Simplified production landscapes disrupt many natural processes, such as pest management, carbon sequestration, and water and soil conservation (Tscharntke et al. 2005). By enhancing provisioning services from landscapes, such as food and fodder, we typically reduce regulating and cultural ecosystem services, such as nutrient cycling, flood protection, and tourism (Tscharntke et al. 2005).

Agricultural landscapes are dynamic, and farmers constantly adapt in incremental ways to environmental changes. Although, in the past, the trend was toward tree clearing and deforestation in many parts of the world, carbon farming creates an incentive to reverse this trend and to increase tree planting and revegetation activities across agricultural landscapes (Crossman et al. 2011).

Economic scenarios of land-use change in response to the introduction of a mandatory price on carbon have been focused on the basic economic incentives (carbon sequestration for maximum offset payments), and it has been assumed that the most profitable system will prevail. Planting a monoculture of trees offers a rapid return on investments (Bekessy and Wintle 2008, Bradshaw et al. 2013); therefore, tree-based monocultures have often been favored over diverse plantings of native species, which would provide a greater variation in vegetation structure and composition and more cobenefits (Nelson et al. 2008). In addition, tree-monoculture plantations, on which there is a single species of uniform age, planted in a regular pattern of large contiguous blocks, are easier to manage and monitor for carbon sequestration than are ecologically diverse plantings composed of many species of different ages and forms, with a complex distribution pattern across the landscape (Harper et al. 2007). Because the prevailing direction in carbon sequestration projects is to move toward tree plantations, it is important to present and incentivize other revegetation methods that allow for considerable carbon sequestration gains while providing the ecological structure and function necessary to provide cobenefits.

In order to better understand the outcomes of carbon farming for stakeholders, we have adopted a commonly used benefits-based framework (Besculides et al. 2002) to assess the co- and disbenefits of carbon-farming initiatives (figure 1). Benefits can be classified in a number of ways (e.g., personal, sociocultural, economic, environmental), with both financial and nonfinancial outcomes (Besculides et al. 2002). Financial benefits are usually characterized monetarily through direct payments (e.g., offset payments for carbon sequestered) or gained efficiency that leads to reduced resource expenditure (e.g., integrated pest management that results in less time and money spent on pesticide application). Nonfinancial benefits include improvements across services or functions and can be measured by performance indicators (e.g., improved filtration of hydrological outflows from a farm; London Councils 2011). *Disbenefits* are outcomes that are perceived by stakeholders as negative, although the same outcome can be seen by different stakeholders as either a benefit or a disbenefit (e.g., increased efficiency can save money but can also cost jobs; London Councils 2011). Because of these incongruities, it is important to understand which stakeholder will be disadvantaged, so that disbenefits can be managed for the affected social group. The trade-offs between cobenefits and disbenefits to landholders will significantly influence the decisions that they make about voluntarily adopting carbon farming and about which land-use strategies are chosen. The goal is to develop carbon-farming methods that provide benefits to private landholders and that encourage

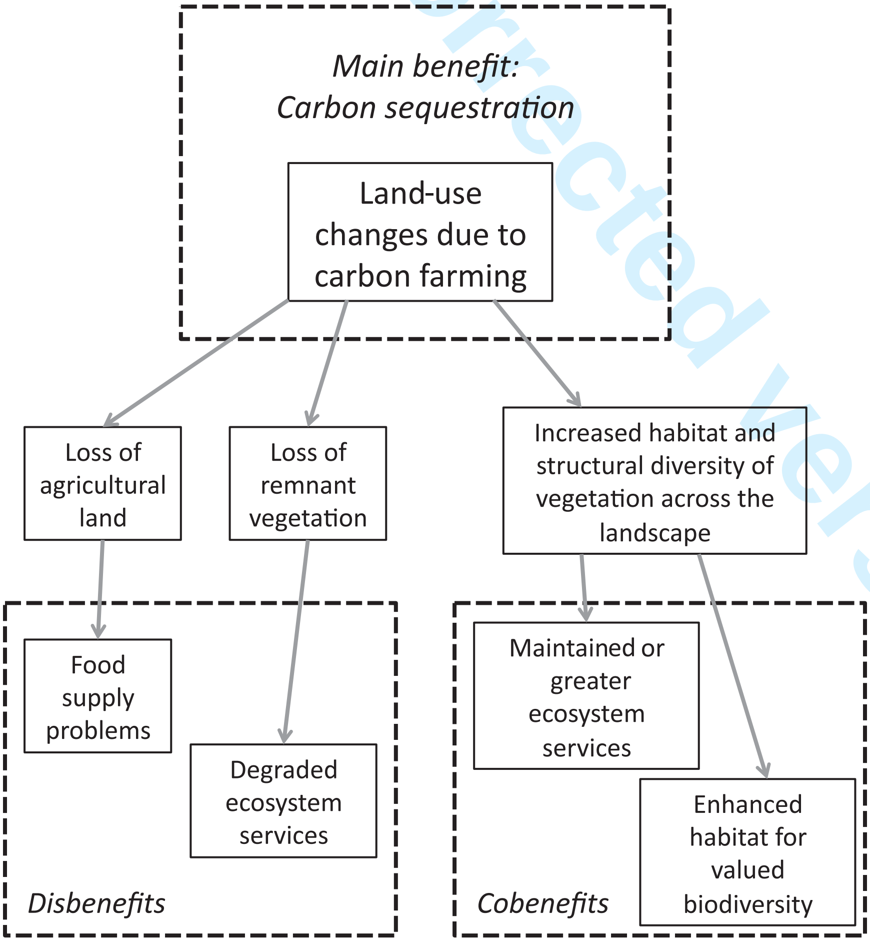

Figure 1. Potential cobenefits and disbenefits of land-use changes associated with carbon farming. This flow diagram provides a general framework for the consideration of the impacts of carbon sequestration across the landscape. It does not categorize all possible impacts but gives a couple examples of land-use changes and the potential cobenefits and disbenefits from those changes. For more specific examples of benefits and disbenefits, please refer to table 1.

wide adoption but also provide cobenefits for the wider public, so that the net benefits can be maximized.

Certain management systems have been shown to be successful in establishing and developing cobenefits with revegetation, and such methodologies could be useful for developing best-practice carbon-farming systems that allow for high carbon sequestration levels while providing cobenefits (figure 2). Raudsepp-Hearne and colleagues (2010) showed that carbon-regulating services can be positively and significantly correlated with several other regulating services, such as soil organic matter, soil phosphorous retention, and drinking water quality; if they are implemented to provide additional services, there is the potential for many environmental public and private cobenefits. Some agricultural systems have been identified as especially successful for providing carbon storage and biodiversity benefits. Multifunctional systems, such as mixtures of multiple species, tree cropping on cropland, and the use of perennial plant species, have been shown to create complex ecosystem structures that sequester carbon, support biodiversity, and deliver ecosystem services to the surrounding agricultural land (see Boody et al. 2005, Jordan et al. 2007). Perennial crops have the potential to capture and hold large quantities of soil organic carbon. For example, in Minnesota, up to 0.9 megagrams (Mg) of carbon per ha per year can be accumulated (Paustian et al. 1997), and a modeled scenario of increased vegetation cover in a single Minnesota catchment showed that soil organic content increased by 86% as more grasslands and riparian buffers were established (Boody et al. 2005). Agroforestry is another example of a diversified and structurally complex system that can provide high rates of carbon sequestration above and below ground and that can increase ecosystem structure for biodiversity through increased multilayered vegetation and tree coverage (Jose 2009). However, estimates of carbon sequestration potential vary, depending on climatic gradients. Ramachandran Nair and colleagues (2009) showed that the carbon sequestration potential of the vegetation component of agroforestry varied from 0.29 Mg per ha per year in a fodder-bank agroforestry system in West African Sahel to 15.21 Mg per ha per year in mixed species stands of Puerto Rico. In alley-cropping systems in Canada, soil carbon estimates were around 1.25 Mg per ha per year, and in silvopastoral systems in Costa Rica, soil carbon measurements were around 173 Mg per ha per year. Many other cobenefits can potentially arise from agroforestry systems, such as hydrological protection (Jose 2009), temperature regulation (Lin 2007), pest and disease control, and enhanced soil fertility (Jose 2009).

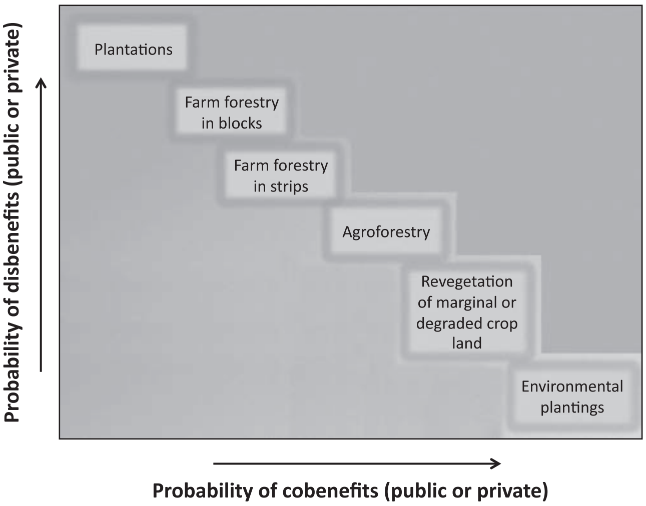

Figure 2. This continuum of carbon revegetation methodologies represents a range of strategies that could be implemented on agricultural land. Their position within the diagram illustrates the likelihood that there will be cobenefits or disbenefits that occur along with carbon sequestration. Plantation styles of revegetation are often favored, because they can provide high levels of carbon sequestration, although there are other circumstances in which other types of planting can still provide higher levels of carbon sequestration, with additional cobenefits.

Certain management systems have proved to be unsuccessful for carbon farming, and information on failed systems can be equally important in developing a list of best-practice carbon-farming methodologies (figure 2). In one example, tree plantations suffered a high rate of failure if few tree species were planted and if the trees were not well suited to the site conditions (Wuethrich 2007). Of 98 publicly funded reforested areas in Brazil, only 2 were considered successful. In most areas, after initially flourishing, many of the trees died and weeds took over. It was found that reforested areas with 30 tree species or more generally performed better than areas with only 3 or 4 species. The appropriateness of the species to the local conditions was an important consideration in successful implementation. Revegetated systems with appropriately and sufficiently diverse species were able to recruit other native flora and fauna to sustain the system (Wuethrich 2007). In addition, monocultural tree plantations can facilitate the establishment of invasive species and can increase susceptibility to species-specific pathogens by creating novel ecosystems with new species combinations and relationships (Hobbs et al. 2006).

Trade-offs of land allocation between revegetation and agriculture will also affect the successful implementation of carbon-farming initiatives. In a study in which the attributes of 42 programs involving carbon-offset payments to fund tree-planting activities across sub-Saharan Africa were examined, Reynolds (2012) found that sites with soil quality and rainfall favorable for tree growth were generally not selected for carbon farming by landholders because of the high opportunity cost of forgone agricultural production, whereas projects on degraded sites were much less disputed and often successfully generated and sold offsets. Contrary to expectations, payments to groups (as opposed to individuals) were common among the successful projects, and benefit sharing (allowing local people access to nontimber products) was another determinant of project success, as was local involvement in project design and implementation (Reynolds 2012). Projects specializing in carbon sequestration alone were far more likely to fail than were projects emphasizing multiple benefits, and most of the successful projects advertised environmental benefits, such as biodiversity, and social outcomes, including employment; training; and access to food, fuel, and clean water (Shames and Scherr 2010). Collective action and benefit sharing in many reforestation projects are important because success is partially contingent on selecting project sites that are environmentally and socially conducive to collective action (Ostrom 1990). Local communities must develop formal institutions consistent with and complementary to the local norms surrounding forest use in order to achieve success (Corbera and Brown 2008). In another example of community-based carbon sequestration leading to multiple social and environmental benefits, the Humbo Assisted Natural Regeneration Project in Ethiopia planted 2728 ha of degraded native forests to access the carbon market while reducing poverty and restoring the local agroecosystem (Brown et al. 2011). The local Humbo communities were able to harvest fodder and firewood within a year of the project's initiation and wild fruits and other nontimber forest products within 3 years. The establishment of user rights and local cooperatives to generate community ownership was important in generating enthusiasm for the project and in empowering the community to sustainably manage communal lands.

In some studies, issues relating to the cobenefits or disbenefits associated with revegetation have begun to be incorporated into the analysis of carbon-farming implementation. For example, Rittenhouse and Rissman (2012) explicitly incorporated cobenefits related to habitat area for species conservation. Increases in habitat area and habitat connectivity resulted from afforestation across Wisconsin for forest-associated species of conservation concern. However, grassland species experienced a reduction in suitable habitat. Crossman and colleagues (2011) modeled the spatial distribution of monoculture carbon plantings and ecological carbon plantings on agricultural land in South Australia. They concluded that an additional but relatively small incentive would be required to establish ecological carbon plantings in areas of high priority for restoration. Less optimistically, in an economic analysis of carbon tree-planting opportunities in the wet tropics of Australia, Hunt (2008) concluded that it is likely that land that is important for endangered or threatened ecosystems will be converted to monocultures, not restored rainforest. These analyses incorporated differences between tree-planting methodologies and spatial configurations (table 1) that can greatly affect the potential for cobenefits and disbenefits (Paul et al. 2013). In table 1, we distinguish between different ways of integrating trees into the agricultural landscape—from systems that have significant levels of integration with the agricultural production system to those for which they are very separate. For example, a large block monoculture is not well integrated into the farm processes of an agricultural landscape but is able to store significant amounts of carbon. Linear planting strips within the farm landscape will have greater levels of integration with farm-level processes and will store some carbon. Environmental blocks might have carbon and biodiversity outcomes but not a lot of integration. The various tree-planting configurations can have different levels of coverage and can satisfy different goals for the landholder.

In many production landscapes, there is land that provides little to no direct farm income and that can be revegetated, including areas along fences, field margins, roadsides, waterways, degraded land, rocky areas, and areas with steep slopes. The opportunity cost of revegetation on these parts of the landscape, where agricultural production is marginal, may be very low. Carefully conceived carbon farming on these areas represents a true win–win opportunity, because the overall production losses due to revegetating these areas will be very low (Macfadyen et al. 2012, Paul et al. 2013). However, well-designed economic and policy instruments focused on both carbon sequestration and biodiversity conservation are required in order to ensure that these areas are revegetated and maintained in the long term (Bradshaw et al. 2013).

Cobenefits of carbon farming for landholders

Multiple cobenefits can be gained from carbon revegetation on agricultural land (figure 1), but these benefits accrue for a range of different beneficiaries. *Private benefits* apply to the private landholder and are a result of changes in land management, whereas *public benefits* are those that affect everyone (Pannell 2008). The two direct benefits are carbon sequestration from revegetation (a public benefit) and the financial gain from carbon-offset payments (a private benefit). However, cobenefits of carbon farming can provide additional positive outcomes such as saved time, money, and resources as a result of increased ecosystem service delivery (e.g., a decreased need for pesticide application) and nonfinancial public environmental benefits (e.g., better air and water quality from reduced chemical inputs). Certain strategies can provide both public and private benefits (e.g., reduced pesticide use can lead to financial benefits to farmers and well-being for the environment and society), and these strategies present an important area of focus for the consideration of carbon-farming methodologies that maximize the beneficial outcomes. Of course, as was stated above, benefits are very context specific, and there is great potential for beneficial land-use change if the carbon price is high enough and cobenefits are possible.

With this perspective, carbon-farming initiatives present an opportunity to revegetate a portion of the agricultural landscape and restore ecosystem structures that maximize both the main benefit and environmental cobenefits (figure 1). Forest restoration studies, such as Chazdon (2008), have shown that many ecosystem functions and

Table 1. Revegetation options for a landholder considering carbon-farming initiatives.

Revegetation methodology	Characteristics	Environmental cobenefits: Private	Environmental cobenefits: Public	Disbenefits	References
Plantations	Monoculture, intensively managed, not permanent	Reduced soil erosion, wind breaks		Negative hydrological flows, altered fire regimes, biodiversity loss, invasive taxa, reduced cropping land, low chance of success	Hobbes et al. 2006, Wuethrich 2007
Farm forestry in blocks (nonindustrial private forests)	Monoculture or a few easily managed species, not permanent	Pest control, pollination, livestock shelter	Reduced pollutant runoff into waterways	The same as for plantations but a higher chance of success	Atela et al. 2012, Rittenhouse and Rissman 2012, Paul et al. 2013
Farm forestry in strips (on nonproductive land)	Monoculture or a few easily managed species, semi-permanent	Pest control, pollination, wind breaks, reduced soil erosion	Improved connectivity, reduced pollution outflow, increased habitat area	Potential biodiversity loss, invasive taxa	Rittenhouse and Rissman 2012, Paul et al. 2013
Agroforestry	Trees and shrubs (can be multiple species, usually native) integrated into the farm, semi-permanent	Pest control, pollination, windbreaks, reduced soil erosion	Improved hydrological connectivity, reduced pollution outflow, increased habitat area	Reduced land available for cropping, competition for water resources	Gurr et al. 2003, Montagnini and Nair 2004, Jose 2009, Rittenhouse and Rissman 2012, Munro et al. 2012
Revegetation of marginal or degraded crop land	A few species that are chosen for their ability to improve a land-degradation issue, permanent	Windbreaks, soil erosion and fertility, salinity control	Improved connectivity, reduced pollution outflow		Atela et al. 2012, Rittenhouse and Rissman 2012, Paul et al. 2013, Reynolds 2012
Environmental plantings in strips (on nonproductive land)	A few to many native species, permanent	Pest control, pollination, wind breaks, reduced soil erosion	Improved connectivity, reduced pollution outflow		Brandle et al. 1992, Smith and Jarvis 1998, Landis et al. 2000, Paul et al. 2013
Environmental plantings in blocks	A few to many native species that replicates a natural community, not permanent	Pest control, pollination, livestock shelter	Species conservation, increased habitat area, improved connectivity	Reduced cropping land	Landis et al. 2000, Munro et al. 2012, Paul et al. 2013

Note: The type of revegetation methodology chosen alters the potential supply of cobenefits and disbenefits. Trade-offs between cobenefits and disbenefits will influence the decisionmaking process for participating in carbon-farming initiatives and for which strategies to implement. The methods toward the lower half of the table have greater cobenefits and fewer disbenefits than those in the upper half. We have made no distinction about which of these strategies will provide more or less carbon sequestration (because they will vary in different contexts).

many components of the original biodiversity can recover from postagricultural transformation. For example, after 30–40 years, natural regeneration following the abandonment of pasture and coffee plantations produced secondary forests in Puerto Rico, with biomass, stem density, and species richness similar to those of the island's mature forests (Zimmerman et al. 2007). Munro and colleagues (2012) found that revegetation on agricultural land in Australia improved the ecosystem functions of soil stability, water infiltration, and nutrient cycling. Interestingly, Munro and colleagues (2012) did not find better function in ecological plantings with multiple trees and shrub species than with monoculture plantings of *Eucalyptus* trees. In general, mechanisms that are able to enhance ecosystem structure will provide opportunities to restore lost ecosystem functions and to potentially strengthen the delivery of ecosystem services (Chazdon 2008). This may include public cobenefits important for environmental well-being, such as soil health and water quality, and those important for human health, such as reduced air pollutants.

On-farm environmental cobenefits can incentivize landholders to engage in carbon-farming initiatives. For example, about 75% of the crops that are grown around the world benefit in some way from insect pollination (Klein et al. 2007). Studies of many crops show that pollinator visitation rates and fruit set are greater for crops that are near patches of noncrop vegetation that can provide pollinator habitat (Ricketts et al. 2008). The rate of yield increase achieved over recent decades is lower for crops that depend most on insect pollination, and it has been hypothesized that this constraint is a result of a trend toward the loss of pollinator habitat from agricultural landscapes (Garibaldi et al. 2011). Therefore, increased tree plantings and revegetation have the potential to increase pollination and crop yields. Pest suppression is a perennial challenge to farmers and, in certain seasons, can be an essential ecosystem service for preventing widespread crop losses. Revegetation and habitat management from carbon farming could be used to create ecological infrastructure that provides favorable resources to natural enemies (invertebrate predators and parasitoids) and that is also practical for producers to implement (Landis et al. 2000). Economic calculations of revegetation for natural enemy habitat have shown monetary benefits to landholders, even when the land was taken out of production for this purpose (Thomas et al. 1991). The results from a meta-analysis study of the density response of natural enemies showed that experimental increases in structural complexity led to a large and significant increase in natural enemy abundance, both at the habitat and the within-plant scales (Langellotto and Denno 2004). In addition, many types of ecological complexity could promote the diversity of natural enemies while increasing the carbon sequestration potential of, for example, silviculture, agroforestry, and the use of perennial noncrop plants (Jose 2009).

Another potential on-farm environmental cobenefit is the ability of increased tree cover to provide windbreaks and other microclimate benefits for crop production. Trees can reduce wind speeds at ground level, especially when they are planted as windbreaks or shelterbelts (Smith and Jarvis 1998). Tree shelter has also been shown to protect crops from extreme storm events in which intense rainfall and high-velocity winds cause landslides, flooding, and premature fruit drop from crop plants (Philpott et al. 2008). There are many additional benefits from windbreaks and shelterbelts, including reduced soil erosion, improved microclimate for crops, shelter for livestock, and increased pasture production (Hodges et al. 2004). Windbreaks are also likely to provide many other benefits that are shared more broadly, rather than concentrated with the landholder, such as greater native biodiversity. The economic benefits to the farmer would increase once one includes the subsidies for revegetation (Brandle et al. 1992) that could be associated with carbon farming.

Barriers and challenges to carbon-farming implementation with maximum cobenefits

There are numerous scientific challenges associated with identifying and developing carbon-farming strategies that maximize cobenefits, including a lack of understanding about which tree-planting strategies allow for increased cobenefits to be developed in addition to carbon sequestration benefits. There are uncertainties that can prove to be challenging in developing initiatives, and many projects may result in outcomes different from those predicted. Furthermore, the broad range of potential beneficiaries means that trade-offs associated with private, public, and shared cobenefits must be thoroughly considered (Pannell 2008).

First, there is still much research needed to understand the value of the various ecosystem services that could arise as cobenefits and to better understand the biological mechanisms that underpin them so that we can design good management strategies. Differences in private and shared benefits and the trade-offs associated with different revegetation strategies will affect which methodologies will be implemented on agricultural land. Biodiversity is often neglected in policy formulation outside of environmental portfolios, despite its support of many ecosystem services (Thompson et al. 2011), and a cobenefits focus will require that biodiversity is considered in revegetation schemes. In addition, ecosystem service benefits (e.g., climate regulation, soil formation, nutrient cycling) are commonly not taken into account in environmental-planning decisions, which leads to a discounting of their value (Balmford et al. 2002). This is largely due to a shortage of information regarding which ecosystem services are affected by changes in land use and the trade-offs associated with ecosystem service loss and land-use conversion. It is important to remember that, although carbon can be traded on the international market as relative equivalents, the environmental cobenefits associated with specific carbon-planting schemes cannot; consequently, the value of cobenefits is usually ignored in trading schemes (McKenzie et al. 2011). Therefore, research on the gains and losses of ecosystem services under alternative carbon-farming strategies is needed in order to determine the level of benefit that can be achieved by different strategies.

Second, private landowners lack incentives to manage land with ecosystem services and biodiversity conservation benefits in mind, because many of the benefits produced on their land benefit others (Nelson et al. 2008). Therefore, many landowners will logically choose revegetation that offers the most carbon sequestration benefits for the least amount of work and cost and that will provide few cobenefits. Incentives to support ecosystem services and biodiversity conservation are sometimes provided by government programs such as the US Department of Agriculture's Conservation Reserve Program and the Common Agricultural Policy in the European Union (Donald and Evans 2006). Crossman and colleagues (2011) investigated the incentives necessary to drive investment away from tree-based monocultures that provide higher carbon yields and toward more-diverse plantings of native tree and shrub species that provide greater vegetation diversity and more structure for biodiversity. They quantified the economic returns from carbon plantings (monoculture and mixed species) under six carbon-price scenarios for landholders in southern Australia. Monoculture plantings sequestered more carbon than mixed plantings did, which therefore required an extra incentive payment for ecological plantings in order for them to be competitive with monoculture plantings. This extra payment increased with higher carbon prices, although at a smaller proportion of the monoculture profit. For example, at a low carbon price of AU$10 per ton of CO_2^{-e}, an average annual payment of AU$7 per ha to the landowner would be needed to make ecological plantings competitive with monoculture plantings. This payment would increase to AU$50 per ha at a medium carbon price of AU$20 per ton of CO_2^{-e} and AU$125 per ha at a high carbon price of AU$45 per ton of CO_2^{-e}. Part of this difference in pricing is related to the higher cost of planting diverse native species rather than monocultures, because a significant effort is required to collect and germinate seeds from diverse sources and to plant in a strategic manner (Pannell et al. 2006).

Third, in complex and diverse agricultural landscapes, there is a poor understanding of the best strategies to revegetate landscapes to regain some of the lost biodiversity, ecosystem functions, and ecosystem services (Munro et al. 2012). For example, in forest ecosystems, restoring functionality depends strongly on the initial state of land degradation and the desired outcome, time frame, and financial constraints of the land manager (Chazdon 2008). In addition, land-use models have shown that policies aimed at increasing only the provision of carbon sequestration across the agricultural landscape do not necessarily also increase species conservation (Nelson et al. 2008). In a study modeling private land-use decisions in response to policy incentives that increase carbon sequestration and species conservation in the United States, Nelson and colleagues (2008) compared land-use changes under five conservation incentive schemes. They found that when conservation was maximized (e.g., through the restoration of rare natural habitats), carbon sequestration benefits were minimal. When carbon sequestration was maximized (e.g., through restored forests), the increased habitat helped a few but not the majority of species. Therefore, there are great challenges to simultaneously increasing carbon sequestration and species conservation, and trade-offs must be considered carefully (Macfadyen et al. 2012). Local knowledge of tree characteristics, plantings of diverse species of ecological and economic importance, and the integration of rehabilitation programs with regional development strategies are essential elements of restoration success (Chokkalingam et al. 2005) and are relevant to producing best-practice tree plantings for carbon farming. Most economic studies on land-use change under carbon farming have not been designed at the farm scale; however, more studies based on a typical small farm in a given area could be used by farmers to assess the costs and benefits of revegetation (e.g., Atela et al. 2012, Mello and Hildebrand 2012).

Fourth, reconciling biodiversity conservation with changing demands on land use requires that the ecological, sociocultural, and economic values of a landscape be fully taken into account in planning and decisionmaking (de Groot 2006). These values may vary across the local community and will certainly change, depending on the levels of private and public benefits gained or lost for the land-use transformation. For some farmers, reforestation on agricultural land may represent a major change in land-use and farming traditions (Polglase et al. 2011). Furthermore, land-use change decisions are often influenced by cultural preferences and other noneconomic motivators. Revegetation activities with broad community acceptance and participation will be more likely to succeed and to be maintained and, therefore, increase the possibilities of achieving long-term environmental cobenefits. In a study of 80 forest commons in 10 countries across Asia, Africa, and Latin America, Chhatre and Agrawal (2009) showed that both larger forest size and greater rulemaking autonomy at the local level were associated with high levels of carbon storage and livelihood benefits. Therefore, the transfer of forest ownership to local communities, coupled with payments for improved carbon storage, can contribute to large-scale sequestration and to benefits for local livelihoods.

Fifth, although there is significant environmental cobenefit potential from carbon farming, land-use change due to carbon farming can also create disbenefits, which may lead to a loss of biodiversity and ecosystem service delivery (figure 1). Perverse outcomes of land-use change could occur as a result of carbon farming, especially if the carbon market increases pressure on native forests and food production land, which could potentially exacerbate food security problems (Niesten et al. 2002). This issue is especially relevant when the international context and the potential for transcountry leakage are considered. *Leakage*, or the displacement of greenhouse gas emissions from one area to another (Atmadja and Verchot 2012), can result in increased deforestation and a loss of remnant vegetation elsewhere, either domestically or internationally. Within the REDD+ scheme, the issue of leakage is beginning to be addressed; however, cross-boundary and international leakage is still a considerable problem that is difficult to quantify but that is indispensable for assessing global efforts to reduce atmospheric carbon (Atmadja and Verchot 2012). The possibility of cobenefits and disbenefits also exists for land-use changes incentivized by biofuels revegetation, and similar analyses can also be adopted to examine such land-use changes in order to maximize cobenefits. However, a further discussion of biofuels-induced land-use change is beyond the scope of this article.

There is also the possibility that remnants of vegetation presently in the landscape, if they are not used for carbon planting, will be removed to make up for productive land lost to carbon plantings (figure 1). Although plantations and replanted vegetation for carbon farming can improve ecosystem services and enhance biodiversity conservation, they are unlikely to match the composition and structure of the original vegetative cover (Hobbs et al. 2006). An equivalent problem occurs in wetland mitigation banking, in which restored wetlands are unable to maintain the quality and function of the original wetlands lost to development (Zedler 2000). Fast-growing, short-lived species with low-density wood are often favored by projects designed to provide carbon offsets, but long-term carbon sequestration is promoted by the growth of long-lived, slow-growing tree species with dense wood and a slow turnover of woody tissues (Chazdon 2008), which also provides more robust ecosystem functions and supports greater biodiversity.

Finally, we must manage our expectations and be realistic about what outcomes are possible in already degraded and marginal lands. Developing a range of methodologies to bring best-practice carbon farming to the landscape will likely come with a number of failures in implementation, as has already been seen in other revegetation projects. Landholders will discover that some ecosystems can be only partially restored, whereas others may never be able to be restored, even under intensive practices. Some systems will require many years to reach something resembling the desired community, and some approaches may restore a specific ecosystem service desired by the landholder but may not achieve significant ecosystem function or biodiversity outcomes. In some areas, the people using the land will have very little control over land-use decisions because of a lack of land tenure and will be unable to implement systems that can be maintained for the time periods required under carbon-farming initiatives (Tallis et al. 2011). Therefore, some level of failure and partial ecosystem service restoration should be expected in the implementation of projects as strategies are tested across many different types of landscapes.

Conclusions

Carbon-farming schemes provide the possibility of supporting practices that not only increase carbon sequestration but also provide environmental cobenefits for society. However, efforts to develop and promote revegetation strategies that go beyond tree monocultures to maximize carbon sequestration are necessary in order to capture additional environmental cobenefits. Benefits come only from certain tree-planting approaches, and it is necessary to understand which planting designs allow cobenefits to develop. Otherwise, tree plantings that are focused solely on carbon sequestration can lead to poor outcomes. Many cobenefits can come with strategic revegetation methods to provide improved ecosystem structure across the landscape, to support biodiversity conservation, and to improve ecosystem service delivery. This outcome should be pursued in the interests of private landholders and society in general. However, developing policies that encourage carbon-farming best practices to maximize cobenefits and to minimize disbenefits will be challenging. Planning for carbon farming that explicitly includes cobenefits will be necessary in order to make carbon-farming policy a beneficial endeavor. Strategies aimed at balancing the tension between a carbon-only focus and a cobenefit focus will lead to different—and, we think, better—land-use decisions. More research and more monitoring of methodologies are required, as are incentives that promote the adoption of structurally complex vegetation. The profit from carbon credits alone may not be enough to drive landowners into the carbon market, but the multiplicity of cobenefits that can be derived—if they are made explicit—may. Continued research into land management strategies and their subsequent cobenefits and disbenefits will be necessary to help landholders decide which strategies to pursue in order to support long-term management of carbon plantings and to ensure that desired cobenefits are sustained across the landscape. Carbon-farming initiatives, therefore, provide an opportunity for those involved in agricultural industries to address their commitment to the environment and land stewardship.

Acknowledgments

We thank Andrew Reeson, Rosalind Bark, and the reviewers for their comments on earlier versions of the manuscript. Funding was provided by the Australian Commonwealth Scientific and Industrial Research Organisation's Climate Adaptation and Sustainable Agriculture Flagships. There are no competing financial interests associated with the publication of this article.

References cited

Atela JO, Denich M, Kaguamba R, Kibwage J. 2012. Agricultural land allocation in small farms around Maasai Mau forest, Kenya and the implications on carbon stocks. Journal of Ecology and the Natural Environment 4: 98–108.

Atmadja S, Verchot L. 2012. A review of the state of research, policies and strategies in addressing leakage from reducing emissions from deforestation and forest degradation (REDD+). Mitigation and Adaptation Strategies for Global Change 17: 311–336.

Balmford A, et al. 2002. Economic reasons for conserving wild nature. Science 297: 950–953.

Bekessy SA, Wintle BA. 2008. Using carbon investment to grow the biodiversity bank. Conservation Biology 22: 510–513.

Besculides A, Lee ME, McCormick PJ. 2002. Residents' perceptions of the cultural benefits of tourism. Annals of Tourism Research 29: 303–319.

Boody G, Vondracek B, Andow DA, Krinke M, Westra J, Zimmerman J, Welle P. 2005. Multifunctional agriculture in the United States. BioScience 55: 27–38.

Bradshaw CJA, et al. 2013. Brave new green world: Consequences of a carbon economy for the conservation of Australian biodiversity. Biological Conservation 161: 71–90.

Brandle JR, Johnson BB, Akeson T. 1992. Field windbreaks: Are they economical? Journal of Production Agriculture 5: 393–398.

Brown DR, Dettmann P, Rinaudo T, Tefera H, Tofu A. 2011. Poverty alleviation and environmental restoration using the clean development mechanism: A case study from Humbo, Ethiopia. Environmental Management 48: 322–333.

Chazdon RL. 2008. Beyond deforestation: Restoring forests and ecosystem services on degraded lands. Science 320: 1458–1460.

Chhatre A, Agrawal A. 2009. Trade-offs and synergies between carbon storage and livelihood benefits from forest commons. Proceedings of the National Academy of Sciences 106: 17667–17670.

Chokkalingam U, et al. 2005. Local participation, livelihood needs, and institutional arrangements: Three keys to sustainable rehabilitation of degraded tropical forest lands. Pages 405–414 in Mansourian S, Vallauri D, Dudley N, eds. Forest Restoration in Landscapes: Beyond Planting Trees. Springer.

Corbera E, Brown K. 2008. Building institutions to trade ecosystem services: Marketing forest carbon in Mexico. World Development 36: 1956–1979.

Crossman ND, Bryan BA, Summers DM. 2011. Carbon payments and low-cost conservation. Conservation Biology 25: 835–845.

De Groot R. 2006. Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. Landscape and Urban Planning 75: 175–186.

Donald PF, Evans AD. 2006. Habitat connectivity and matrix restoration: The wider implications of agri-environment schemes. Journal of Applied Ecology 43: 209–218.

Fahey TJ, Woodbury PB, Battles JJ, Goodale CL, Hamburg SP, Ollinger SV, Woodall CW. 2010. Forest carbon storage: Ecology, management, and policy. Frontiers in Ecology and the Environment 8: 245–252.

Garibaldi LA, Aizen MA, Klein AM, Cunningham SA, Harder LD. 2011. Global growth and stability of agricultural yield decrease with pollinator dependence. Proceedings of the National Academy of Sciences 108: 5909–5914.

Harper RJ, Beck AC, Ritson P, Hill MJ, Mitchell CD, Barrett DJ, Smettem KRJ, Mann SS. 2007. The potential of greenhouse sinks to underwrite improved land management. Ecological Engineering 29: 329–341.

Hobbs RJ, et al. 2006. Novel ecosystems: Theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15: 1–7.

Hodges L, Suratman MN, Brandle JR, Hubbard KG. 2004. Growth and yield of snap beans as affected by wind protection and microclimate changes due to shelterbelts and planting dates. Hortscience 39: 996–1004.

Hunt C. 2008. Economy and ecology of emerging markets and credits for bio-sequestered carbon on private land in tropical Australia. Ecological Economics 66: 309–318.

Jordan N, et al. 2007. Sustainable development of the agricultural bioeconomy. Science 316: 1570–1571.

Jose S. 2009. Agroforestry for ecosystem services and environmental benefits: An overview. Agroforestry Systems 76: 1–10.

Klein A-M, Vaissière BE, Cane JH, Steffan-Dewenter I, Cunningham SA, Kremen C, Tscharntke T. 2007. Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B 274: 303–313.

Landis DA, Wratten SD, Gurr GM. 2000. Habitat management to conserve natural enemies of arthropod pests in agriculture. Annual Review of Entomology 45: 175–201.

Langellotto GA, Denno RF. 2004. Responses of invertebrate natural enemies to complex-structured habitats: A meta-analytical synthesis. Oecologia 139: 1–10.

Lin BB. 2007. Agroforestry management as an adaptive strategy against potential microclimate extremes in coffee agriculture. Agricultural and Forest Meteorology 144: 85–94.

London Councils. 2011. Definition of benefits and disbenefits. London Councils. (24 June 2013; *http://pspmawiki.londoncouncils.gov.uk/index.php/Definition_of_Benefits_And_Disbenefits*)

Macfadyen S, Cunningham SA, Costamagna AC, Schellhorn NA. 2012. Managing ecosystem services and biodiversity conservation in agricultural landscapes: Are the solutions the same? Journal of Applied Ecology 49: 690–694.

McKenzie E, Irwin F, Ranganthan J, Hanson C, Kousky C, Bennerr K, Ruffo S, Conte M, Salzman J, Paavola J. 2011. Incorporating ecosystem services in decisions. Pages 339–354 in Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S, eds. Natural Capital, Theory and Practice of Mapping Ecosystem Services. Oxford University Press.

Mello R, Hildebrand P. 2012. Modeling effects of climate change policies on small farmer households in the Amazon Basin, Brazil. Journal of Sustainable Forestry 31: 59–79.

Munro NT, Fischer J, Wood J, Lindenmayer DB. 2012. Assessing ecosystem function of restoration plantings in south-eastern Australia. Forest Ecology and Management 282: 36–45.

Nelson E, Polasky S, Lewis DJ, Plantinga AJ, Lonsdorf E, White D, Bael D, Lawler JJ. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. Proceedings of the National Academy of Sciences 105: 9471–9476.

Niesten E, Frumhoff PC, Manion M, Hardner JJ. 2002. Designing a carbon market that protects forests in developing countries. Philosophical Transactions of the Royal Society A 360: 1875–1888.

Ostrom E. 1990. Governing the Commons: The Evolution of Institutions for Collective Action. Cambridge University Press.

Pannell DJ. 2008. Public benefits, private benefits, and policy mechanism choice for land-use change for environmental benefits. Land Economics 84: 225–240.

Pannell DJ, Marshall GR, Barr N, Curtis A, Vanclay F, Wilkinson R. 2006. Understanding and promoting adoption of conservation practices by rural landholders. Australian Journal of Experimental Agriculture 46: 1407–1424.

Paul KI, Reeson A, Polglase P, Crossman N, Freudenberger D, Hawkins C. 2013. Economic and employment implications of a carbon market for integrated farm forestry and biodiverse environmental plantings. Land Use Policy 30: 496–506.

Paustian K, Andrén O, Janzen HH, Lal R, Smith P, Tian G, Tiessen H, Van Noordwijk M, Woomer PL. 1997. Agricultural soils as a sink to mitigate CO_2 emissions. Soil Use Management 13: 230–244.

Perez C, Roncoli C, Neely C, Steiner JL. 2007. Can carbon sequestration markets benefit low-income producers in semi-arid Africa? Potentials and challenges. Agricultural Systems 94: 2–12.

Philpott SM, Lin BB, Jha S, Brines SJ. 2008. A multi-scale assessment of hurricane impacts on agricultural landscapes based on land use and topographic features. Agriculture, Ecosystems, and Environment 128: 12–20.

Polglase P, et al. 2011. Opportunities for Carbon Forestry in Australia: Economic Assessment and Constraints to Implementation. Australian Commonwealth Scientific and Industrial Research Organisation.

Ramachandran Nair PK, Mohan Kumar B, Nair VD. 2009. Agroforestry as a strategy for carbon sequestration. Journal of Plant Nutrition and Soil Science 172: 10–23.

Raudsepp-Hearne C, Peterson GD, Bennett EM. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences 107: 5242–5247.

Reynolds TW. 2012. Institutional determinants of success among forestry-based carbon sequestration projects in sub-Saharan Africa. World Development 40: 542–554.

Ricketts TH, et al. 2008. Landscape effects on crop pollination services: Are there general patterns? Ecology Letters 11: 499–515.

Ritson P, Sochacki S. 2003. Measurement and prediction of biomass and carbon content of *Pinus pinaster* trees in farm forestry plantations, south-western Australia. Forest Ecology and Management 175: 103–117.

Rittenhouse CD, Rissman AR. 2012. Forest cover, carbon sequestration, and wildlife habitat: Policy review and modeling of tradeoffs among land-use change scenarios. Environmental Science and Policy 21: 94–105.

Schoeneberger M, Bentrup G, de Gooijer H, Soolanayakanahally R, Sauer T, Brandle J, Zhou X, Current D. 2012. Branching out: Agroforestry as a climate change mitigation and adaptation tool for agriculture. Journal of Soil and Water Conservation 67: 128A–136A.

Shames S, Scherr S. 2010. Institutional Models for Carbon Finance to Mobilize Sustainable Agricultural Development in Africa. US Agency for International Development.

Smith DM, Jarvis PG. 1998. Physiological and environmental control of transpiration by trees in windbreaks. Forest Ecology and Management 105: 159–173.

Tallis H, Pagiola S, Zhang W, Shaikh S, Nelson E, Stanton C, Shyamsundar P. 2011. Poverty and the distribution of ecosystem services. Pages 278–295 in Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S, eds. Natural Capital, Theory and Practice of Mapping Ecosystem Services. Oxford University Press.

Thomas MB, Wratten SD, Sotherton NW. 1991. Creation of “island” habitats in farmland to manipulate populations of beneficial arthropods: Predator densities and emigration. Journal of Applied Ecology 28: 906–917.

Thompson ID, Okabe K, Tylianakis JM, Kumar P, Brockerhoff EG, Schellhorn NA, Parrotta JA, Nasi R. 2011. Forest biodiversity and the delivery of ecosystem goods and services: Translating science into policy. BioScience 61: 972–981.

Tilman D, Fargione J, Wolff B, D’Antonio C, Dobson A, Howarth R, Schindler D, Schlesinger WH, Simberloff D, Swackhamer D. 2001. Forecasting agriculturally driven global environmental change. Science 292: 281–284.

Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C. 2005. Landscape perspectives on agricultural intensification and biodiversity—ecosystem service management. Ecology Letters 8: 857–874.

[UN-REDD] United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries. 2009. About REDD+. UN-REDD. (25 June 2013; *www.un-redd.org/aboutredd/tabid/582/default.aspx*)

Wuethrich B. 2007. Reconstructing Brazil’s Atlantic rainforest. Science 315: 1070–1072.

Zedler JB. 2000. Progress in wetland restoration ecology. Trends in Ecology and Evolution 15: 402–407.

Zimmerman JK, Aide TM, Lugo AE. 2007. Implications of land-use history for natural forest regeneration and restoration strategies in Puerto Rico. Pages 51–74 in Cramer VA, Hobbs RJ, eds. Old Fields: Dynamics and Restoration of Abandoned Farmland. Island Press.

Brenda B. Lin (brenda.lin@csiro.au) is affiliated with the Australian Commonwealth Scientific and Industrial Research Organisation’s Climate Adaptation and Marine and Atmospheric Research Flagships, in Aspendale; Sarina Macfadyen and Saul A. Cunningham with its Ecosystem Sciences and Sustainable Agriculture Flagship, in Canberra; and Nancy A. Schellhorn with its Ecosystem Sciences and Sustainable Agriculture Flagship, in Brisbane, Australia. Anna R. Renwick is affiliated with the Environmental Decisions Group, in the School of Biological Sciences at the University of Queensland, in St Lucia, Australia.