Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services

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Although ecological restoration is widely used to combat environmental degradation, very few studies have evaluated the cost-effectiveness of this approach. We examine the potential impact of forest restoration on the value of multiple ecosystem services across four dryland areas in Latin America, by estimating the net value of ecosystem service benefits under different reforestation scenarios. The values of selected ecosystem services were mapped under each scenario, supported by the use of a spatially explicit model of forest dynamics. We explored the economic potential of a change in land use from livestock grazing to restored native forest using different discount rates and performed a cost–benefit analysis of three restoration scenarios. Results show that passive restoration is cost-effective for all study areas on the basis of the services analyzed, whereas the benefits from active restoration are generally outweighed by the relatively high costs involved. These findings were found to be relatively insensitive to discount rate but were sensitive to the market value of carbon. Substantial variation in values was recorded between study areas, demonstrating that ecosystem service values are strongly context specific. However, spatial analysis enabled localized areas of net benefits to be identified, indicating the value of this approach for identifying the relative costs and benefits of restoration interventions across a landscape.

biodiversity | conservation | dry forest | sustainable development

The widespread occurrence of environmental degradation has led to increasing interest in the science and practice of ecological restoration, which seeks to enhance the recovery of degraded land and watercourses (1, 2). Restoration initiatives being undertaken around the world make a significant contribution to sustainable development (3) and are of major importance for adaptation to climate change (4). However, despite the large number of restoration initiatives that have been established, few attempts have been made to systematically evaluate their effectiveness (5). To address this knowledge gap, Rey Benayas et al. (6) performed a metaanalysis of 89 restoration assessments undertaken in a wide range of ecosystem types and found that restoration increased provision of biodiversity and ecosystem services by 44% and 25%, respectively, according to an analysis of response ratios. However, values of both remained lower in restored than in relatively intact ecosystems.

Ecosystem services are the benefits that people obtain from ecosystems (7). According to the Millennium Ecosystem Assessment (MEA) (8), 63% of these benefits are in serious decline at the global scale. Such declines are likely to have a large negative impact on the future of human welfare (9), especially because more than 70% of the 1.1 billion people below the poverty line live in rural areas and are directly reliant on natural resources for survival (10). Publication of the MEA, supported by the previous work of Daily (11, 12), Balmford (13), and others, has identified the need for integrated research into the value of nature for human well-being as a strategy toward achieving sustainable development goals. Although rapid progress has been made in understanding how ecosystems provide services, it has proved more difficult to produce credible, quantitative estimates of ecosystem service values (14). In particular, there is a need for spatially explicit analyses of how the provision of multiple ecosystem services and their associated values might change under alternative land use scenarios (14).

Here we attempt to provide such analyses for the specific example of dryland forest restoration. The problem of environmental degradation is recognized to be most intense in arid and semiarid areas (15), which together constitute half the surface area of the world's developing countries (16). Rural communities in dryland areas are often highly dependent on forest resources to support their livelihoods. However, in many areas dryland forests are severely threatened because of unsustainable land use practices, including livestock husbandry, use of fire, and overharvesting of fuelwood (17). These processes have caused widespread degradation of dryland forests, resulting in negative impacts on biodiversity, soil fertility, and water availability, as well as on the livelihoods of local people (16). Such degradation presents a major challenge to policy initiatives aiming to support sustainable development. Restoration of dryland forest ecosystems is an urgent priority if such policy goals are to be achieved.

In this article, we examine the potential impact of restoring dryland forests on the provision and value of selected ecosystem services. We use a conceptual framework that focuses on quantifying the costs and benefits associated with changes in ecosystem services as a result of policy action, through comparison of two counterfactual scenarios (18). This approach is in line with the emerging consensus about the importance of comparing alternative policy actions rather than a static analysis of current service provision (14, 18, 19). Another key feature of the approach adopted here is that it is spatially explicit, reflecting the fact that both the production and value of ecosystem services varies spatially (19, 20). Relatively few previous attempts have been made to analyze the spatial dynamics of ecosystem services in relation to policy scenarios, although recent progress has been made by the Natural Capital Project and others (14, 18, 21, 22).

Many early ecosystem service assessments focused only on estimating benefits (13), an approach that could potentially mislead decision making (23). Very few previous attempts have been made to perform a cost–benefit analysis (CBA) of restoration projects. In a review of more than 2,000 restoration case studies, the TEEB study (The Economics of Ecosystems and Biodiversity) (4) found that less than 5% provided meaningful cost data, and none provided analysis of both costs and benefits. The approaches for modeling multiple ecosystem services

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adopted here provide a means of estimating such benefits, and when combined with estimates of costs, enable a CBA of restoration actions to be performed. Because policy decisions are often evaluated through cost–benefit assessments, CBA can help make ecosystem service research operational (24).

This study evaluates the cost-effectiveness of dryland forest restoration through a comparative analysis of four study areas in Latin America. The study landscapes varied from 24 kha to 228 kha in extent and consist of a mosaic of pasture, cropland, urban, and dry forest areas. Current land cover maps obtained from authorities in each study area were used to represent the business-as-usual state (BAU). Maps for the restoration scenario states were produced using a spatially explicit model of forest dynamics, LANDIS-II (25). Net present values (NPVs) of carbon sequestration, nontimber forest products (NTFP), timber, tourism, and livestock production for both states were calculated and mapped. Cost-effectiveness of restoration was analyzed by estimating the "net social benefit (NSB) of restoration": the net change in value of the ecosystem services associated with land cover change minus the costs associated with reforestation. We explored the NSB of restoration using different discount rates and performed a CBA for each.

Results

Net Social Benefit. Results are presented for restoration scenarios over a policy-relevant time horizon of 20 y. Other time horizons were also explored (*SI Text*). A discount rate of 5% was used for the results presented here, according to guidelines on long-term project assessments in the region, presented by the World Bank (26). The percentage of dry forest cover under the current land cover situation varied among study areas, from 1% in Nahuel Huapi (NH) to 52% in El Tablon (ET) (Table 1). The increase in dry forest area during the 20-y time horizon was also highly variable, from 0.3% to 65% for NH and Central Veracruz (CV), respectively (Table 1).

For each scenario the NPV of carbon sequestration, livestock production, NTFP harvest, timber production, and tourism was calculated. The NPV is the difference in value between the BAU scenario and the restoration scenarios. For each scenario we also estimated the NPV of all direct costs associated with restoration, including fencing and fire suppression. A restoration scenario's NSB is calculated by summing all of the scenario's NPVs.

In all four study areas, there was a net gain in ecosystem service provision, with four of the ecosystem services increasing in net value as a result of forest restoration. Livestock production value decreased in all areas, representing an opportunity cost of forest restoration. Contrasting results were obtained from the different study areas, with timber and tourism providing relatively high values (>0.8 US$/ha per year) for ET and Quilpue (Q), respectively, compared with the other areas (Fig. 1). With the exception of carbon sequestration, all other positive values were <0.28 US$/ha per year, for all study areas. ET provided the greatest increase in value for NTFP extraction using current harvest rates (0.07 US$/ha per year), but by comparison with other ecosystem services, the increased value derived from NTFPs was relatively low for all study areas. Strikingly, values of carbon sequestration were substantially higher than those of the other ecosystem services in all study areas, with the exception of NH, where the value of carbon was relatively low (0.16 US$/ha per year) and similar to the change in value of timber (0.11 US$/ha per year).

The NSB of "passive restoration" varied between approximately US$ 1 million and US$ 42 million (NH and Q, respectively) over the 20-y time horizon using a 5% discount rate (Table 2). This scenario incorporates the opportunity cost of loss of livestock production, which varied substantially among study areas, ranging from US$ 0.02 million to US$ 1.4 million for NH and CV, respectively. In all four study areas, the costs associated with fencing and fire suppression (as used in the "passive restoration with protection" scenario) were substantially higher than the opportunity cost, as demonstrated by the contrast in negative NSBs for this scenario (Table 2). The additional costs of tree establishment included in the "active restoration" scenario again varied among study areas, primarily reflecting the area reforested and regional variation in material and labor costs. Consequently, the costs of active restoration differed by more than an order of magnitude among study areas (Table 2).

For a restoration action to be cost-effective, CBA requires that the subtraction of costs from benefits results in a positive outcome and that the benefit–cost ratio (BCR) is >1. The passive restoration scenario provided positive BCRs for all study areas, with Q providing the highest ratio (Table 2). Benefits for the passive restoration with protection scenario only accounted for 74%, 88%, 25%, and 33% of restoration costs in CV, ET, NH, and Q, respectively. Active restoration was even less cost-effective in all of the study areas (Table 2).

NSB varied spatially within each study area (Fig. 2). The maps produced indicate that passive restoration is likely to be cost-effective throughout most of the study areas where forest can establish naturally. Areas of net cost were only identified in ET for this scenario, for an area <10 ha in extent, reflecting the relatively high opportunity cost of livestock production in localized areas (Fig. 2). In contrast, the maps illustrate clearly that for the active restoration intervention, costs are likely to outweigh the benefits throughout most of the study areas. However, isolated locations were identified within the two Mexican study areas (CV and ET) where active restoration is likely to be cost-effective (Fig. 2).

Changing Parameters. Variation in discount rate or time horizon had relatively little effect on the outcome of CBA for each restoration scenario in terms of whether restoration produced net benefits or net costs (Fig. 3). Passive restoration was associated with a positive NSB in all study areas for all discount rates, ranging from 0 to 10% (Fig. 3), whereas active restoration was associated with negative NSB. Passive restoration with protection was similarly negative in terms of NSB in all study areas, except in ET at the 0% discount rate. Exploration of carbon value demonstrated that the results were sensitive to the market value. For example, in ET, when the carbon value was increased from US$ 25.30/ton to US$ 43/ton, active restoration became cost-effective at all discount rates except 10%. Similarly, in CV, passive restoration with protection became cost-effective at all discount rates. Altering the carbon value had less impact on NSB of NH and Q, owing to the relatively high costs in relation to benefits in these two areas.

Discussion

The analyses presented here show that passive forest restoration is cost-effective in all four study areas, according to the estimated values of the ecosystem services considered. These results support

Table 1. Extent of forest cover (ha) for the BAU and restoration scenarios

	Study area			
Variable	CV	ET	NH	Q
Total study area size (ha)	29,468	24,754	228,289	170,897
BAU dry forest extent (ha) (% of total study area)	1,300 (4.4)	12,955 (52.3)	2,742 (1.2)	25,891 (15.2)
Restoration scenario dry forest extent (ha) (% of total study area)	20,376 (69.1)	17,949 (72.5)	3,448 (1.5)	130,446 (76.3)

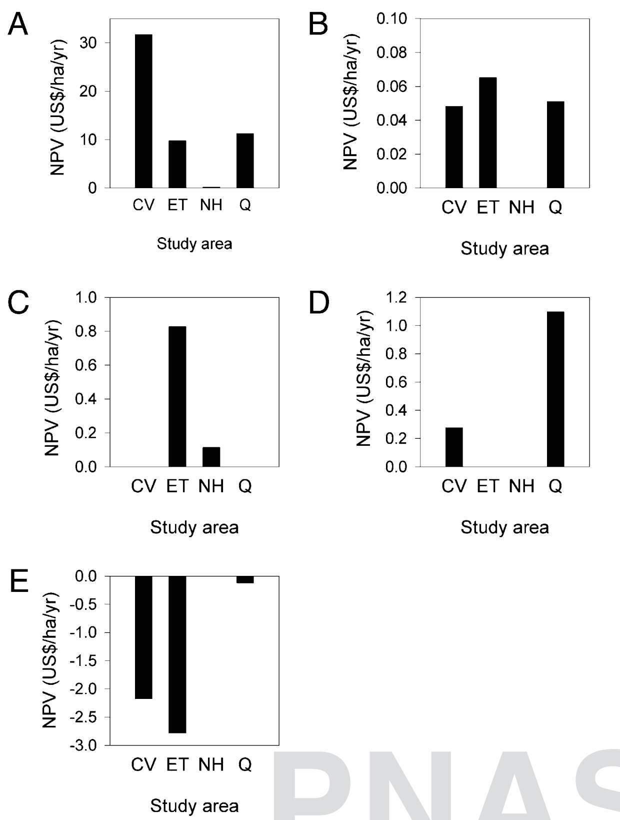

Fig. 1. NPV of ecosystem services (US$/ha per year) for 20-y time horizon at 5% discount rate, excluding restoration costs. (*A*) Carbon sequestration; (*B*) NTFP; (*C*) timber; (*D*) tourism; (*E*) livestock production.

the suggestion that facilitating ecosystem restoration by encouraging natural regeneration has considerable potential for cost-effective landscape-scale restoration (27). In contrast, active restoration was not cost-effective in any of the study areas reported here. Although evidence suggests that ecological restoration is generally effective in increasing the provision of ecosystem services (6), very little information is available regarding whether such restoration is cost-effective (4). TEEB (4) evaluated the cost-effectiveness of restoration projects on the basis of a "benefits transfer" approach, reflecting the lack of studies providing estimates of both costs and benefits. Results of the TEEB review indicated an average BCR of 28.4 for woodlands and shrublands (4), which falls within the range of values recorded here.

This study builds on previous research that has attempted to value and map ecosystem services. The analytical approach adopted, which compares how ecosystem service provision and value differ under alternative scenarios, offers significant advantages over previous efforts that have focused on mapping "total" values (28, 29). Specifically, use of scenarios enables the economic impact of a particular change in land use to be estimated, which is of greater value for informing policy decisions (24, 30). Relatively few investigations have used such comparative approaches to date (13, 14, 22). The advantage of spatially explicit analyses is that they enable the concept of ecosystem services to be integrated into conservation planning (28, 31) and allow areas with the greatest potential benefits per unit cost to be identified, allowing management interventions to be targeted more effectively. In addition, the use of a spatially explicit model of vegetation dynamics to support the development of restoration scenarios has not been used previously in the context of mapping ecosystem services.

A further innovative element of our investigation was the comparison of multiple study areas, which enables the generality of results to be explored. Substantial variation in NPVs was recorded in the results obtained from the four areas, demonstrating that ecosystem service values are strongly context specific. This provides further evidence that use of the "benefit transfer" approach to analysis of ecosystem services, as used by Costanza et al. (29) and TEEB (4), may have significant limitations.

Pronounced variation between study areas was found in the values of the different ecosystem services, emphasizing the importance of assessing multiple services, as noted by Nelson et al. (14). Values were influenced strongly by existing land cover and land use patterns in the four areas. In all study areas, carbon sequestration showed the highest value after restoration. Low net gains in NTFP value suggest that these products would not provide a significant income to enable opportunity costs to be exceeded at the current rate of extraction. Payment for carbon sequestration services would seem to have greater potential for compensating the negative local livelihood impacts that might result from forest restoration.

Our results also indicate that the market value applied to ecosystem services can markedly influence the outcome of CBA. A range of different values has been used in previous investigations (14, 23). Here, the cost-effectiveness of forest restoration was found to be highly sensitive to carbon value. It was also expected that results would be sensitive to the discount rate. A range of different discount rates has been used in the ecosystem service literature; for example, Naidoo and Ricketts (23) used a value of 20%, whereas Nelson et al. (14) used a value of 7%. The impacts of such variation on estimations of NSB have rarely been explored but can be substantial (30). However, in the present investigation, variation in

Table 2. NSB and BCR for each restoration scenario

	Study area							
	CV		ET		NH		Q	
Scenario	US$	US$/ha per y	US$	US$/ha per y	US$	US$/ha per y	US$	US$/ha per y
NSB*								
Passive restoration	17,602,530	597	3,909,721	158	1,244,310	5	41,951,959	245
Passive restoration with protection	−6,156,739	−209	−526,820	−21	−3,652,614	−16	−84,749,260	−496
Active restoration	−22,734,892	−772	−2,374,193	−96	−4,861,293	−21	−123,980,100	−725
BCR								
Passive restoration	14.92†		3.84†		75.14†		100.72†	
Passive restoration with protection	0.74		0.88		0.25		0.33	
Active restoration	0.44		0.62		0.20		0.25	

*NSB represents the summed change in NPV of ecosystem services between the BAU and restoration scenario maps, minus the direct restoration costs for each scenario.

†BCR >1 suggests that an option is cost-effective.

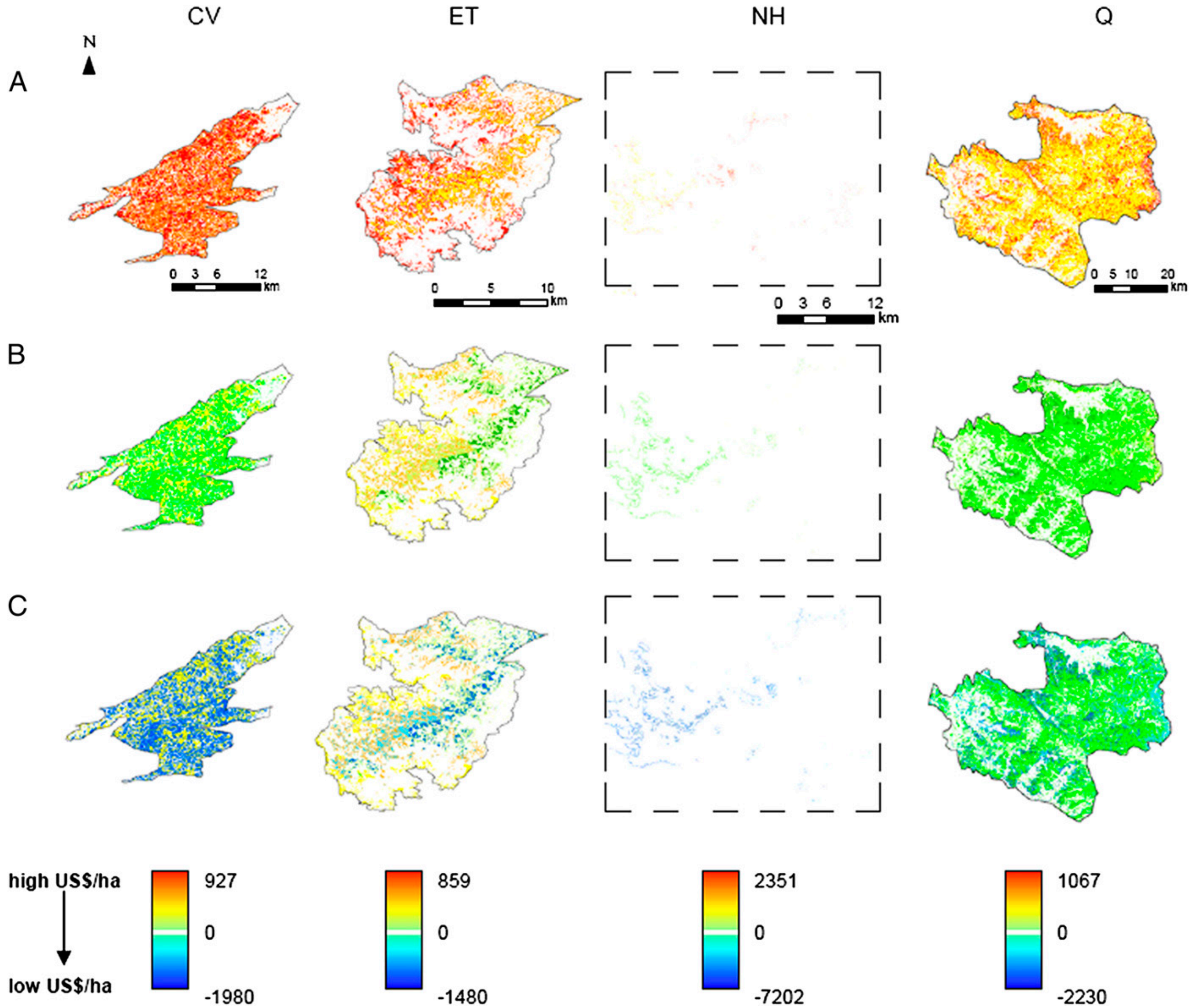

Fig. 2. Maps of NSB (US$/ha) for the combined ecosystem services (20 y, 5% discount rate) for the four study areas under three restoration scenarios: (*A*) passive restoration; (*B*) passive restoration with protection; and (*C*) active restoration. For NH, only the forested part of the study area is illustrated. White space (representing zero value) has been removed for display purposes.

NSB was found to be relatively insensitive to discount rate, implying that the main findings are robust.

The results presented here should be viewed as tentative, given the uncertainties associated with mapping and valuing ecosystem services. In the case of tourism, for example, it would have been useful to identify the limits to tourist numbers that each study area can sustain, but this information was unavailable. Ideally, dynamics in the biophysical supply of any given ecosystem service should be modeled simultaneously with its economic demand (24) and in our case studies we do not know how demand for tourism would respond to increased supply in tourism potential. This is clearly an important relationship in order to deliver a more robust tourism value in the future. In addition, approaches are required that enable interactions between different ecosystem services to be explored in relation to changing socioeconomic conditions. Another important issue not addressed by the present study is that of equity, in terms of the distributions of costs and benefits, and the potential variation between study areas in the marginal utility of a unit of benefit.

Despite these limitations, these results highlight the potential benefits of ecological restoration to human communities and support suggestions that restoration actions should be undertaken in degraded lands (2, 6). Dryland areas should be considered as particularly high priorities for ecological restoration, because environmental degradation is particularly severe in such areas (15). It is widely accepted that poverty alleviation and the conservation of biodiversity are inextricably linked (32). Given the evidence that restoration can enhance both ecosystem services and biodiversity (6), reforestation of degraded lands provides an opportunity to achieve both conservation and socioeconomic development goals. Focusing restoration efforts on dryland forests could potentially enhance the biodiversity associated with a threatened terrestrial ecosystem and could also improve local livelihoods.

However, the costs of ecological restoration can be substantial (2, 4). Billions of dollars are currently being spent across the globe on ecological restoration projects (33), many of which may not be successful. There is a need to identify where restoration projects will incur net benefits for conservation and human well-being, so that efforts can be effectively targeted (33). The present investigation indicates how such CBAs can be provided in a spatially explicit manner. Even in locations where restoration is likely to be cost-effective, financial incentives will need to be provided to

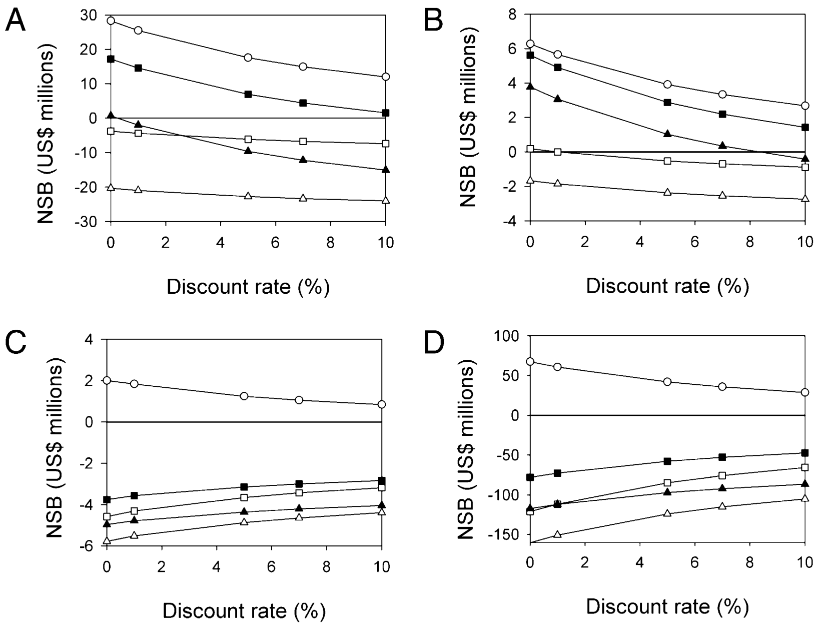

Fig. 3. NSB for the scenarios, using discount rates from 0 to 10% and time horizon of 20 y. The restoration scenarios were passive restoration (open circles); passive restoration with protection (open squares); and active restoration (open triangles). In addition, the results obtained assuming a higher value for carbon are presented for the passive restoration with protection (filled squares) and active restoration scenarios (filled triangles).

support restoration actions. Potential approaches include improved markets and payment schemes for ecosystem services (34), although proper targeting of incentive schemes requires knowledge regarding the distribution of beneficiaries. Further research would be required to identify these in our study areas. One potential mechanism is Reducing Emissions from Deforestation and Forest Degradation (REDD-Plus), which could be used to provide incentives for forest restoration (35). The sensitivity of NSB to carbon value recorded here implies that manipulation of market values through differentiation of forest carbon in the marketplace could have a major influence on the cost-effectiveness of ecological restoration, in relation to such incentives.

Materials and Methods

Study Areas. The investigation was conducted in four study areas (Table S1): CV, Veracruz (Mexico); ET, Chiapas (Mexico); NH, Río Negro/Neuquén (Argentina); and Q, Valparaíso region (Chile). All four study areas are global conservation priorities, being included within priority ecoregions defined by Olson and Dinerstein (36), and in the case of CV, ET and Q, as global biodiversity hotspots (37). Land cover maps for the four areas were derived from remote sensing imagery (Table S1). Spatial analyses and map production were performed using ArcGIS 9.2 (©1999–2006, ESRI).

Scenarios. Three restoration scenarios were developed: passive (no restoration costs); passive with protection (costs of fencing and fire suppression); and active (costs of tree planting, fencing, and fire suppression).

Land cover maps were generated for the restoration scenarios using a spatially explicit model of forest dynamics (LANDIS II). This model is designed to simulate the dynamics of forested landscapes through the incorporation of ecological processes, including succession, disturbance, and seed dispersal, and has been applied to a wide range of forest types (25, 38). The process of tree establishment is modeled according to species life history parameters and habitat suitability, which is determined by edaphic variables. The LANDIS II model was individually parameterized and verified for each of the four study areas (*SI Text*), then used to define the spatial extent of forest restoration for both passive and active restoration scenarios (Table S2).

Services. We estimated the value of multiple ecosystem services under different scenarios. There is often a lack of data to make precise estimates of the value of ecosystem services or how values change under different scenarios. We have therefore made explicit the assumptions made in estimating these values.

Carbon Sequestration. Above-ground living biomass per hectare for dry forest was calculated from published allometric equations using data from randomized forest plots and was then extrapolated to forest land cover area. Biomass per hectare in other pools (deadwood, litter, and roots) and for other land cover types was derived from relevant estimates in the literature (Tables S3–S6). A carbon fraction of 0.5 was used to estimate carbon content according to these biomass values. A value of US$ 25.30/ton of carbon was used, on the basis of the estimated social cost of carbon dioxide emissions for the period 2001–2020 given in Fankhauser (39). The NPV is the difference in carbon storage between BAU and restoration scenario. It was assumed that carbon market value at the end of the time horizons (T) was the same as the current value. In addition, it was assumed that the biomass per unit area of forest and other land cover types at time T is the same as at $t = 0$. Alternative values for carbon of US$ 43/ton [based on Tol (40) and US$ 4.4/ton (based on the lowest over-the-counter (OTC) market carbon credit price of US$ 1.20 /tCO_2 for 2008 presented in Hamilton et al. (41)] were also used as a comparison.

Nontimber Forest Products. Information on which NTFPs had a market value was obtained from local experts within each study area. NTFPs that are not currently traded (i.e., those used for subsistence purposes only) were excluded from the analysis. A local market value was identified on the basis of available scientific literature and expert knowledge. Extraction costs were estimated on the basis of the income foregone owing to the time taken to harvest these products, or in the case of commercial extraction, operational costs. The NPV of harvested NTFPs was calculated using local harvest rates obtained from empirical data on annual extraction of these products. It was not possible to evaluate potential price changes, so it was assumed that the market price of NTFPs remained constant regardless of the change in supply. Harvest rates were also assumed to be sustainable. We recognize that these assumptions are a potential source of uncertainty in the results obtained.

Timber Production. Commercial timber harvesting occurs within three of the study areas. Data on which species of tree are commercially valuable were obtained from local experts. The net timber value in US$/ton was calculated by subtracting the extraction costs (foregone income) from the local market value. An estimate of the mass of harvestable trees in the study area based on forest inventory data and minimum harvest diameters was used to predict the change in harvestable biomass according to land cover change. Sustainable timber availability was calculated according to the International Tropical Timber Organization (2006) guidelines of 1 m^3/ha for sustainable harvesting of tropical forests (42). It was not possible to evaluate potential price changes, so it was assumed that the market price of timber remained constant regardless of the change in supply. The implications of this assumption should be borne in mind when considering the results.

Tourism. Tourist activities were diverse and differed across the study areas. Not all study areas had activities that were dependent on forest cover (Table

S7). For those areas where forest was considered to influence tourism, annual tourism income data and annual visitor numbers were gathered from the scientific literature and interviews with local tourism experts within each study area. The mean annual spend per visitor per unit area of dry forest was calculated as an indication of willingness to pay. Multiplication of this value by the change in dry forest land cover provided an NPV for tourism. This method assumes that each unit of forest has equal value that remains constant over the time horizon and is not affected by total forest area. This assumption may well be incorrect, and further modeling of potential change in tourist preference, spending, and thresholds might provide a different result. However, in the absence of suitable models or evidence suggesting a threshold limit to tourism, this has not been included in the analysis.

Livestock Production. Estimates of livestock numbers, information on which types of livestock were kept, density of animals, and the net value of animal products (market sale prices minus direct production costs) for each area were obtained from interviews with livestock holders, municipality officials, and the scientific literature. The NPV of livestock resulting from forest restoration was estimated by multiplying the net value of animals per unit area of grazing land by the change in the extent of available grazing land.

Costs. Costs varied according to the different restoration scenarios. The principal opportunity cost of each restoration scenario was the loss of income from livestock production, which is the main alternative land use in each of the study areas. The costs considered for passive restoration were therefore limited to these opportunity costs (i.e., loss of livestock) resulting from increased forest cover. It is possible that local people in the study areas may incur additional costs as a result of forest restoration activities, but these were not explored here. The estimated costs for the passive restoration with protection and active restoration scenarios were derived from data obtained from restoration field trials within each of the study areas (*SI Text*).

Cost–Benefit Analysis. CBA was performed to examine whether the gain in ecosystem service benefits outweighed the costs of implementing the forest restoration scenarios (Table S8). By adding all NPVs of ecosystem services and restoration activities, the NSB of restoration was represented in a spatial manner for all restoration scenarios. Initial maps were based on a 5% discount rate, and then subsequent analyses were performed with a variety of discount rates.

Discounting. Benefits and costs were estimated at a range of discount rates to explore the sensitivity of research findings to this variable (see *SI Text* for further details). A discount rate of 5% was used for the results presented here, according to guidelines on long-term project assessments in the region, presented by the World Bank (26).

Additional Materials and Methods. For further details on the methods and materials used in this study, see *SI Text* and Tables S1–S8.

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Supporting Information

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SI Text

Net Social Benefit of Restoration. This study compared two land cover maps: one representing the current land cover of each study area based on satellite imagery, and another representing forest restoration derived from output of a spatial model (LANDIS II). The current land cover map acts as the "current scenario" and is used as a comparison with projected future land cover, as a "business as usual" (BAU) comparison. To examine the change in supply of ecosystem services, a net value of each ecosystem service per unit area of each land cover type was calculated on the basis of empirical data, and this was multiplied by the change in area of each land cover type between the current and projected land cover maps. This method assumes that if land cover were to remain unchanged, there would be no change in the ecosystem service flows provided in T years time. The term "net social benefit (NSB) of restoration" is therefore used in this study to represent the difference between the BAU and the restoration scenarios represented by the projected land cover.

LANDIS II Model. LANDIS II has been extensively used to examine the dynamics of a wide range of different forest types in different parts of the world (1, 2). A detailed description of the LANDIS-II model is provided elsewhere (1, 2) (http://www.landis-ii.org/). In LANDIS-II, forest succession is a competitive process governed by the probability of establishment in different ecoregions, and the life history characteristics of each species, which include longevity, age of sexual maturity, shade tolerance class, effective and maximum seed dispersal distance, vegetative reproduction probability, minimum and maximum age of vegetative reproduction, and postfire regeneration. LANDIS II is based on an object-oriented modeling approach operating on raster maps, with each cell containing species, environment, disturbance, and harvesting information. Tree species are simulated as the presence or absence of species age cohorts in each cell, at a time step specified by the user (1). LANDIS II was parameterized separately for each of the four study areas, using analysis of remote sensing imagery supported by field surveys and forest inventory data. The life history characteristics of tree species were obtained from field surveys, the scientific literature, and knowledge of local experts. Details of model parameterization and verification for each of the four study areas are provided by refs. 1–3.

Ecosystem Services

Carbon Sequestration. Carbon in above- and below-ground biomass (living vegetation, dead wood, litter, and roots) was determined according to land cover type. Soil organic carbon (SOC) was not considered in this study owing to the high variability in SOC estimates. Soil carbon storage is influenced by climate, vegetation, soil physical characteristics, and historical land use, which vary spatially (4). There is no agreed depth for measurements, so studies are often not comparable. Additionally, many studies show that with changing land use, losses or gains in soil carbon stocks are relatively small compared with the change in aboveground biomass, so despite soil holding two to three times more carbon than biomass, it contributes little impact to the change in carbon stocks (5–8). This is particularly the case for the relatively short-term scenario of 20 y. We therefore consider that the exclusion of soil carbon will not greatly affect the overall results obtained in this study.

To determine above-ground biomass in the dry forested areas, allometric equations from the published literature were applied to diameter at breast height measurements from forest inventory data obtained within each study area. In the case of Nahuel Huapi (NH), no estimate of tree density was available, so inventory data could not be used. Reliable estimates from the scientific literature were therefore used instead (Table S3–S6). Biomass values for all plots were summed and converted to metric tons (Mg) per hectare, which were then multiplied by the total area of dry forest represented on the land cover map to produce an estimate of total biomass for the dry forest land cover type. From this value, a carbon storage value was calculated for each study area by multiplying the biomass by 0.5, following other similar studies (5, 9–12). Although there is some variation in the wood carbon fraction between species, a value of 0.5 is typically used for such estimates (10). A root/shoot ratio of 0.26 was used to estimate below-ground biomass, following Cairns et al. (13). Carbon value was again taken to be half of the total biomass. Biomass per unit area for each land cover type was multiplied by the change in area of each land cover type and multiplied by the appropriate carbon value. T is the time horizon between the current situation and the projected land cover under the forest restoration scenario. For this analysis, it was assumed that carbon storage changes only with land cover type, not with land cover quality or habitat condition and that carbon was sequestered at a constant rate during time T.

Nontimber Forest Products. The following nontimber forest products (NTFPs) were considered: firewood [Central Veracruz (CV), El Tablon (ET), NH, and Quilpue (Q)]; charcoal (Q); saponins (Q); *Boldo* leaves (Q); *Chemadora* palm leaves (ET); and bushmeat (NH), having been identified through field survey as the most significant in terms of economic income for each study area. The mass of harvestable product per unit area was based on forest inventory data, information in the scientific literature, and expert knowledge. Net values were calculated by subtracting the extraction costs (estimated according to the income foregone owing to the time taken to harvest these products, or in the case of commercial extraction, operational costs) from current market values (obtained from the scientific literature and expert consultation).

Empirical data on harvest rates were obtained from the scientific literature. In all cases, except firewood extraction in CV, these rates were assumed to be sustainable over the time horizons considered. In the case of CV, the harvest rate was considered unsustainable and hence the rate used in ET was applied. NTFP extraction was permitted at this rate from $t = 0$ because it was deemed sustainable on the basis of the present area of forest. We did not assess the accessibility of NTFPs in reforested areas, which would provide further information about the realization of the benefits of increased NTFP availability. It was assumed that NTFP productivity per unit forest area remained constant over time T.

The net present value (NPV) of NTFPs in each study area was calculated by subtracting net values from BAU from net value of the restoration scenario. For spatial analysis, the land cover type in which each commercially valuable NTFP occurs was identified by elicitation of expert knowledge and by consultation of the scientific literature, and the NPV was mapped to reflect the change in land cover.

Timber Production. The land cover type in which each commercially valuable timber species occurs was identified by elicitation of expert knowledge and by consultation of the scientific literature. The total biomass of timber in each study area was calculated by compiling values of biomass per unit area, based on forest inventory data, including only trees above the local minimum harvest diameter. The change in biomass of timber was calculated according to the land cover change. The value derived for potential timber production was calculated from an estimated sustainable extraction rate of 1 m^3/ha (14) multiplied by the net value of timber (market value of timber minus production costs) per hectare, to provide an estimate of the marginal change in sustainable timber value. The amount of harvestable biomass in each study area was considered in view of the harvest rate, and all areas except CV and Q were deemed to have sufficient biomass to permit harvesting from $t = 0$. In CV, harvesting began at $t = 21$. In Q, no timber harvest occurs because trees are used for NTFPs rather than timber. It was assumed that timber productivity remained constant over time T.

Tourism. Table S7 lists the tourist activities that were considered in each study area. These activities were chosen as a result of interviews with local tourism operators, tourist offices, and Internet-based research into tourism options in these areas. Annual tourism income data and number of visitors were obtained from the scientific literature and interviews with local tourism experts within each study area [CV: Beltran (15); NH: Manzur (16); Q (17)]. These data were primarily visitor entrance fees to the protected areas within the study sites, income to local tourism businesses such as vineyards, and hunting operators. Other studies have used similar methods to evaluate tourism income (18, 19). The values presented should be considered an underestimate of tourism value. In the recent published literature, more complex methods such as contingent valuation through questionnaire surveys of visitors and the travel cost method have been used (20–22). The former approach has been criticized because there are a number of methodological issues that can give rise to significant inaccuracies (23). The travel cost method requires access to detailed information (where visitors are coming from, their mode of transport, group size, etc.) that was not available for the present study. Although the "cash-flow" approach adopted here is relatively simplistic and could be strengthened by further research, it provides a repeatable measure of tourism value that enables a preliminary comparative analysis of the four study areas.

For the restoration scenarios, it was assumed that not all tourist activities would benefit from increased forest area. Only rafting, trekking, and bird watching in CV; local recreation, bathing, and visits to the protected area in Q; and trekking and cycling in NH were considered to be positively linked to forest presence (Table S7) (although the latter were outweighed by activities that were negatively influenced by forest or not at all influenced by forest). Tourism revenue at T was estimated on the basis of a "willingness to pay" (WTP) value per unit area of forest calculated as spend per visitor/area of forest. We assumed that each unit of forest has equal value that remained constant over the time horizon and was not affected by total forest area. Complex modeling of potential change in tourist preference and spending was beyond the scope of this investigation, and this should be considered when interpreting the results.

Here the local WTP value per unit area of dry forest was calculated by dividing annual tourism income by annual visitors and then by the area of dry forest. NPV was calculated by multiplying the WTP value by the change in dry forest land cover area. In ET there are presently no tourist visitors. In NH most activities were not influenced by forest area. NH and ET therefore had an NPV of zero.

Livestock Production. Livestock production can be considered as a service provided by agro-ecosystems, providing a direct economic benefit to people. In the current investigation, however, it was assumed that livestock production is primarily associated with pasture land, which would decline in area as a result of forest restoration activities. For this reason, livestock production was considered here as a potential cost of forest restoration. Data on current livestock densities for each land cover type were multiplied by the change in area of land cover types that support grazing animals, and mean local values for each livestock type (i.e., cattle, sheep, goats), to estimate marginal livestock value. Available pasture land was reduced in the reforestation scenarios, so the NPV of livestock value was negative. It is possible that local people in the study areas may incur additional costs as a result of forest restoration activities, but these were not explored here. Any other such costs are likely to be relatively insignificant in comparison with losses of livestock production.

Restoration Costs. Following the definitions in Lamb and Gilmour (24), "passive restoration" refers to forest establishment through the natural regeneration of tree species, involving the ecological processes of dispersal and succession. This could potentially be achieved in the study areas examined here by removal of grazing animals. In addition, a "passive restoration with protection" scenario was explored, which involved the protection of sites from grazing and fire. "Active restoration" refers to forest establishment by direct intervention (25), in this case by the planting of tree seedlings.

Costs were estimated of implementing each of the three restoration scenarios that were explored. In the case of passive restoration, the only costs included were the opportunity costs of the decline in livestock production associated with loss of pasture land. For passive restoration with protection, the labor and material costs of fencing the study areas were incorporated, in addition to the opportunity costs of declining livestock production. For active restoration, these same costs were included together with the labor and material costs of direct planting of tree seedlings. In addition, the passive restoration with protection and active restoration scenarios also included costs of annual maintenance for fire suppression. Estimates of labor and material costs per hectare were obtained from local experts in each of the study areas, based on forest restoration field trials established previously. Establishment costs (labor and fencing) were applied in year 1. Fire suppression was costed annually over a 20-y period. Maintenance costs were distributed across the total reforested area. Establishment costs were applied to the area that was reforested from all land cover types excluding shrubland, where establishment costs were assumed to be zero.

Discounting. Discount rates are widely used in economic analyses to assess the present value of future benefits on the basis of assumptions such as positive rates of inflation, continual economic growth, and time preference. Typically, the value of any future amount of money is discounted at a chosen rate to estimate current NPV. The use of discounting for cost–benefit analysis in environmental policy is the subject of debate. There is no consensus between economists as to what rate should be applied to environmental projects, if any (26). Different rates can produce highly contrasting economic outcomes and hence provide confusing messages to policy makers (26, 27). In addition, discounting has major implications for the valuation of benefits in the distant future, because the exponential function used shortchanges the value of these services to future generations. The ethos of the ecosystem service approach and sustainable development requires that the future of nature and human well-being be considered, and this has led to suggestions that discounting is not appropriate in relation to ecological restoration projects (25). In the present investigation, a range of discount rates was explored, to enable the sensitivity of results to variation in discount rates to be examined. The following discount rates were applied to the annual benefits and costs for each study area: 0, 1%, 5%, 7%, and 10%. We used the following equations to calculate the NPV of each ecosystem service benefit (B) and present value (PV) of the restoration cost (C) for T (the time horizon considered) to give the NSB of each restoration scenario:

$$\mathrm{NPV}_{(B)} = \sum_{t=0}^{T} (F_t) - PC/(1+r)^{\mathrm{t}}$$

$$\mathrm{PV}_{(C)} = \sum_{t=0}^{T} (C_t)/(1+r)^{\mathrm{t}}$$

$$\mathrm{NSB} = \sum_{t=0}^{T} \left(\mathrm{NPV}_{(B)}\right) - \left(\mathrm{PV}_{(C)}\right),$$

where F_t is ecosystem service flow (F) in US$ at time t, PC is the production costs, C_t is restoration costs (e.g., planting trees, fencing), and r is the discount rate.

Scenarios. The rate of forest spread predicted using LANDIS II differed between study areas, reflecting variation in the dispersal rates of the tree species involved. However, for the study to produce policy-relevant information, a time horizon of 20 y was used for the presentation of data. In addition, time horizons of 100 y and for maximum forest extent were analyzed. LANDIS model runs were carried out for 400 y, and the times required to develop the projected maximum extent of forest cover were as follows: CV, 25 y; ET, 20 y; Q, 60 y. In NH, the forest continued to increase in area at a low rate beyond 400 y. The relatively slow increase of forest area in NH can be explained by the low dispersal rate of the main tree species, *Austrocedrus chilensis*.

Maps. All of the maps produced use a 5% discount rate for the 20-y time horizon. For the passive restoration scenario, maps of the ecosystem services (carbon, livestock, NTFPs, timber, and tourism) were combined. The livestock map had negative value, and hence this calculation produced the NSB for passive restoration. Restoration cost maps were then subtracted from this map to produce a map of NSB for the passive restoration with protection and active restoration scenarios.

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Table S1. Background information for the study areas

Characteristics	Study area CV	ET	NH	Q
Country	Mexico	Mexico	Argentina	Chile
State	Veracruz	Chiapas	Rio Negro/Neuquen	Valparaíso region
Geographic location	19° 06′ 52′′ – 19° 29′ 26′′ N 96° 05′ 33′′– 97° 03′ 42′′ W	16° 11′ 38′′ – 16° 22′ 29′′ N 93° 31′ 57′′ – 93° 44′ 31′′ W	40° 54′ 37′′ – 41° 15′ 20′′ S 70° 41′ 4′′ – 71° 24′ 1′′ W	32° 56′ 7′′ – 33° 22′ 49′′ S 70° 59′ 14′′ – 71° 39′ 53′′ W
Area (ha)	29,468	24,750	228,289	170,897
Elevation range (m asl)	0–1315	675–1,537	700–>1,000	15–2,129
Mean min–max temperature (° C)	19.8–30.7	13.9–33	2–14	8.7–19.4
Mean annual rainfall (mm)	966	913–1,500	500–1,800	531
Designation	Not protected	Within La Sepultura Biosphere Reserve	Part of study area includes Nahuel Huapi National Park	Contains Lago Peñuelas National Reserve and La Campana National Park. Study area is proposed Biosphere Reserve
Land cover map details				
Map source	InEcol, Jalapa, Veracruz	ECOSUR, San Cristobal de las Casas, Chiapas	Universidad Nacional del Comahue, Bariloche, Rio Negro	Universidad de Concepción, Concepción
Imagery and year	SPOT images were acquired in December 2007 and January 2008	QuickBird images were acquired in November–December 2004 (3 images)	Landsat ETM image was acquired February 2003	Landsat TM imagery acquired November 2008–March 2009
Spatial resolution (pixel size) (m)	80	49.8	28.5	90
Projection	WGS 1984 UTM Zone 14N	WGS 1984 UTM Zone 15N	Campo Inchauspe Transverse Mercator (National Grid Argentina Faja1)	WGS 1984 UTM Zone 19S

ECOSUR, El Colegio de la Frontera Sur; ETM, enhanced thematic mapper; SPOT, Satellite Pour l'Observation de la Terre; TM, thematic mapper; UTM, universal transverse mercator; WGS, world geodetic system.

Table S2. Change in land cover (ha) projected under the restoration scenarios for time horizons t = 20 y (t_{20}); t = 100 y (t_{100}); t_{max}

	Study area CV			ET			NH			Q		
Land cover type	t_{20}	t_{100}	t_{max}	t_{20}	t_{100}	t_{max}	t_{20}	t_{100}	t_{max}	t_{20}	t_{100}	t_{max}
Dry forest	19,075	19,455	19,460	4,994	4,994	4,994	706	2,411	8,938	104,555	111,819	111,884
Other forest				1,602	1,602	1,602	−211	−547	−1,449			
Shrubland	−6,854	−6,856	−6,856	−4,140	−4,140	−4,140	−280	−868	−2,663	−74,469	−74,469	−74,469
Grassland (pasture)	−12,221	−12,598	−12,604	−2,456	−2,456	−2,456	−5	−36	−314	−12,314	−15,865	−15,909
Dry grassland							−56	−227	−1,176			
Bare ground							−135	−641	−2,928	−17,771	−21,485	−21,506
Rocky outcrops							−20	−91	−408			

Table S3. Carbon values by land cover type for Central Veracruz

Land cover type	Above-ground carbon (Mg C/ha)			Below-ground carbon (roots) (Mg C/ha)	Total carbon (Mg C/ha)
	Living trees*	Dead wood	Litter layer		
Dry forest	53.01[†]	7.95[‡]	3.65[§]	13.78[¶]	78.39
Secondary forest	20.10	0.00	0.00	0.00	20.10
Plantation	18.15	0.00	0.00	0.00	18.15
Cultivated	6.00	0.00	0.00	0.00	6.00
Pasture	14.05	0.00	0.00	0.00	14.05
Bareground	0.00	0.00	0.00	0.00	0.00
Urban	0.00	0.00	0.00	0.00	0.00
Water	0.00	0.00	0.00	0.00	0.00

*Values from Cairns et al. (1) classes: 15, 1, 19, 2 except for dry forest. Values include carbon in roots, dead wood, and litter, so no values recorded separately in this table.

[†]Values for dry forest obtained from forest plot data using allometric equation in Navar (2).

[‡]Mean value of 15% of above-ground biomass in living trees used, taken from ranges reported in Brown (3) and Gibbs et al. (4).

[§]Mean value for forested areas in tropical climatic zone from IPCC (5), Chapter 3.2, "Forest Land" (Table 3.2.1).

[¶]Value derived from Cairns et al. (6) as 0.26 fraction of above-ground biomass in living trees.

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Table S4. Carbon values by land cover type for El Tablon

Land cover type	Above-ground carbon (Mg C/ha)			Below-ground carbon (roots) (Mg C/ha)	Total carbon (Mg C/ha)
	Living trees*	Dead wood	Litter layer		
Dry forest	39.65[†]	5.95[‡]	3.65[§]	10.31[¶]	59.56
Cloud forest	74.90	0.00	0.00	0.00	74.90
Shrubland	20.10	0.00	0.00	0.00	20.10
Cultivated	6.00	0.00	0.00	0.00	6.00
Pasture	10.90	0.00	0.00	0.00	10.90
Urban	0.00	0.00	0.00	0.00	0.00

*Values from Cairns et al. (1) classes: 2, 8, 1, 5 except for dry forest. Values include carbon in roots, dead wood, and litter, so no values recorded separately in this table.

[†]Values for dry forest obtained from forest plot data using allometric equations in Navar (2, 3) and Acosta-Mireles et al. (4).

[‡]Mean value of 15% of above-ground biomass in living trees used, taken from ranges reported in Brown (5) and Gibbs et al. (6).

[§]Mean value for forested areas in tropical climatic zone from IPCC (7), Chapter 3.2, "Forest Land" (Table 3.2.1).

[¶]Value derived from Cairns et al. (8) as 0.26 fraction of above-ground biomass in living trees.

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Table S5. Carbon values by land cover type for Nahuel Huapi

Land cover type	Above-ground carbon (Mg C/ha) Living trees	Dead wood*	Litter layer	Below-ground carbon (roots) (Mg C/ha)	Total carbon (Mg C/ha)
Dry forest	67.10[†]	10.07	8.50[†]	17.45[‡§]	103.12
Wet forest	15.00[¶]	2.25	27.50[‖]	3.90[‡]	48.65
Shrubland	7.40**	1.11	27.50[‖]	20.72[††]	56.73
Wet grassland	3.00[‡‡]	0.00	0.00	8.40[††]	11.40
Dry grassland	1.50[§§]	0.00	0.00	4.20[††]	5.70
Bareground	0.00	0.00	0.00	0.00	0.00
Urban	0.00	0.00	0.00	0.00	0.00
Water	0.00	0.00	0.00	0.00	0.00
Ice/snow	0.00	0.00	0.00	0.00	0.00

*Mean value of 15% of above-ground biomass in living trees used, taken from ranges reported in Brown (1) and Gibbs et al. (2).
[†]Values from Laclau (3).
[‡]Value derived from Cairns et al. (4) as 0.26 fraction of aboveground biomass in living trees.
[§]Value from Cairns et al. (4) was used because the value in Laclau (3) was unusually low.
[¶]The main species in this area is *Nothofagus pumilio*. Value adapted from Frangi et al. (5).
[‖]Mean value for forested areas in cold temperate dry climatic zone from IPCC (6), chapter 3.2, "Forest Land" (Table 3.2.1).
**Value taken from Ruesch and Gibbs (7), Table 1f, "Shrub cover for temperate FAO ecofloristic zone."
[††]Derived from root/shoot ratios in IPCC (6), Chapter 3.4, "Grassland" (Table 3.4.3).
[‡‡]Value taken from Ruesch and Gibbs (7), Table 1g, "Grassland for temperate steppe FAO ecofloristic zone."
[§§]Value taken from Ruesch and Gibbs (7), Table 1h, "Sparse grassland for temperate steppe FAO ecofloristic zone."

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Table S6. Carbon values by land cover type for Quilpue

Land cover type	Above-ground carbon (Mg C/ha) Living trees	Dead wood*	Litter layer	Below-ground carbon (roots) (Mg C/ha)	Total carbon (Mg C/ha)
Dry forest	26.64[†]	4.00	27.50[‡]	6.93[§]	65.07
Shrubland	7.40[¶]	1.11	27.50[‡]	20.72[‖]	56.73
Plantation	18.20**	2.73	27.50[‡]	4.74[§]	53.16
Cultivated	5.00[††]	0.00	0.00	0.00	5.00
Pasture	3.00[‡‡]	0.00	0.00	8.40[‖]	11.40
Urban	0.00	0.00	0.00	0.00	0.00
Bareground	0.00	0.00	0.00	0.00	0.00
Water	0.00	0.00	0.00	0.00	0.00

*Mean value of 15% of above-ground biomass in living trees used for forested areas, taken from ranges reported in Brown (1) and Ruesch and Gibbs (2).
[†]Values for dry forest obtained from forest plot data using allometric equation in Navar (3).
[‡]Mean value for forested areas in cold temperate dry climatic zone from IPCC (4), Chapter 3.2, "Forest land" (Table 3.2.1).
[§]Value derived from Cairns et al. (5) as 0.26 fraction of above-ground biomass in living trees.
[¶]Value taken from Ruesch and Gibbs (2), Table 1f, "Shrub cover for temperate FAO ecofloristic zone."
[‖]Derived from root/shoot ratios in IPCC (4), Chapter 3.4, "Grassland" (Table 3.4.3).
**Mean value used from Walsh et al. (6). The higher value for *Eucalyptus globulus* was not used because the Chilean climate does not allow these trees to grow productively.
[††]Value taken from Ruesch and Gibbs (2), Table 1i, "Other classes."
[‡‡]Value taken from Ruesch and Gibbs (2), Table 1g, "Grassland for temperate steppe FAO ecofloristic zone."

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Table S7. Tourism and recreation activities in the four study areas and the degree of dependence on forest land cover

	Category								
Study area	Cultural		Adventure		Nature		Recreation		Overall
CV	Cave paintings	=	Rafting	+	Bird watching (raptors)	+			
			Climbing	=					
			Abseiling	=					+
			Trekking	+					
ET									=
NH	Cave paintings	=	Trekking	+	Bird watching (*Vultur gryphus*)	=	Deer hunting	–	
			Cycling	+			Fishing	=	
			Horse riding	–					=
			Skiing	=					
Q	Vineyards	=			Rare species (*Jubaea chilensis*)	+	Bathing	+	
							Fishing	=	
							General local visitor	+	+

+, dependent on forest cover; –, negatively impacted by forest cover; =, not influenced by forest cover.

Table S8. Benefit–cost ratios for the reforestation scenarios for time horizons t = 20 y (t_{20}); t = 100 y (t_{20}); t_{max}

Study area/scenario	Discount rate (%)				
	0	1	5	7	10
CV					
Passive					
t_{20}	14.72	14.72	14.72	14.72	14.72
t_{100}	13.65	13.64	13.61	13.60	13.59
t_{max}	13.65	14.51	13.89	13.60	13.19
Passive with protection					
t_{20}	0.88	0.85	0.74	0.69	0.62
t_{100}	0.90	0.61	0.24	0.19	0.15
t_{max}	0.91	0.73	0.38	0.31	0.25
Active					
t_{20}	0.58	0.55	0.44	0.39	0.33
t_{100}	0.60	0.39	0.14	0.11	0.08
t_{max}	0.60	0.47	0.23	0.18	0.13
ET					
Passive					
t_{20}	3.84	3.84	3.84	3.84	3.84
t_{100}	2.84	1.97	0.90	0.76	0.67
t_{max}	3.84	3.84	3.84	3.84	3.84
Passive with protection					
t_{20}	1.03	1.00	0.88	0.83	0.75
t_{100}	1.03	0.69	0.28	0.22	0.18
t_{max}	1.03	1.00	0.88	0.83	0.75
Active					
t_{20}	0.79	0.75	0.62	0.57	0.49
t_{100}	0.79	0.52	0.20	0.15	0.12
t_{max}	0.79	0.75	0.62	0.57	0.49
NH					
Passive					
t_{20}	120.58	122.58	75.51	119.64	118.29
t_{100}	338.81	349.15	67.25	333.93	326.94
t_{max}	1,169.18	1,284.07	58.46	1,148.97	1,089.45
Passive with protection					
t_{20}	0.30	0.30	0.25	0.23	0.21
t_{100}	2.83	1.97	0.75	0.58	0.45
t_{max}	34.98	9.53	2.35	1.80	1.37
Active					
t_{20}	0.26	0.25	0.20	0.18	0.16
t_{100}	2.39	1.65	0.61	0.46	0.34
t_{max}	29.55	7.96	1.88	1.42	1.05
Q					
Passive					
t_{20}	101.72	101.72	101.72	101.72	101.72
t_{100}	96.09	96.09	96.09	96.09	96.09
t_{max}	96.04	101.77	97.88	96.04	93.39
Passive with protection					
t_{20}	0.36	0.35	0.33	0.32	0.30
t_{100}	0.42	0.29	0.12	0.10	0.08
t_{max}	0.42	0.17	0.05	0.04	0.03
Active					
t_{20}	0.30	0.29	0.25	0.24	0.21
t_{100}	0.35	0.23	0.09	0.07	0.06
t_{max}	0.35	0.14	0.04	0.03	0.02

Based on carbon value of $25.30 at different discount rates. CV: t_{max} = 60 y; ET: t_{max} = 20 y; NH: t_{max} = 400 y; Q: t_{max} = 240 y.