Viable reserve networks arise from individual landholder responses to conservation incentives

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Abstract

Conservation in densely-settled biodiversity hotspots areas often requires setting up reserve networks that maintain sufficient contiguous habitat to support viable species populations. Because it is difficult to secure landholder compliance with an tightly constrained reserve network design, attention has shifted to voluntary incentive mechanisms, such as purchase of conservation easements by reverse auction or through a fixed-price offer. These mechanisms carry potential advantages of transparency, simplicity, and low cost. But uncoordinated individual response to these incentives has been assumed to be incompatible with conservation goals of viability (which depends on contiguous habitat) and biodiversity representation. We model such incentives for southern Bahia in the Brazilian Atlantic Forest, one of the biologically richest and most threatened global biodiversity hotspots. Here, forest cover is spatially autocorrelated and associated with depressed land values, a situation that may be characteristic of long-settled areas with forests fragmented by agriculture. We find that in this situation, a voluntary incentive system can yield a reserve network characterized by large, viable patches of contiguous forest, and representation of subregions with distinct vegetation types and biotic assemblages – without explicit planning for those outcomes.

Introduction

Drastic anthropogenic loss of habitat in biologically outstanding regions has created biodiversity hotspots (Myers *et al.* 2000; Cincotta *et al.* 2000) where the long-term viability of threatened endemic species is questionable in the absence of conservation intervention. Conservation of these threatened ecosystems requires implementation of landscape-scale networks that restore (Rosenzweig 2003) and maintain sufficient contiguous habitat to support viable species populations and ecological processes (Sanderson and Harris 2000; Sanderson *et al.* 2003). In general, and especially in the densely-settled hotspots, this goal will require the cooperation of landholders. This will usually entail landholders' acceptance of some restrictions on land use or land management.

The conservation literature has devoted extensive attention and sophistication to the problem of *where* to impose these restrictions. (See reviews in Margules and Pressey 2000; Stoms *et al.* 2004). The problem is framed in optimization terms: find the landscape configuration that achieves specified specified environmental goals at minimum cost. Many of the earlier exercises focused narrowly on species representation as a goal, and used crude proxies for cost, such as area. More recently, the set of objectives has expanded to include the resilience or persistence of protected biodiversity, and the maintenance of ecological processes. (Cowling, Pressey, Rouget, and Lombard 2003). Economic measures of opportunity cost, rather than land area, are increasingly used as a minimand. Detailed conservation plans have been developed for regions including Papua New Guinea (Faith et al 2001) and the Cape Provinces (Cowling and Pressey 2003). An underlying premise is that explicit central planning is necessary to achieve landscape-level objectives such as connectivity and representation. The result, typically, is a rather tightly-constrained plan that specifies precisely which landscape units are to be included in the conservation system.

Yet according to Faith et al. (2003) "In spite of a decade or more of work on reserve selection methods, no complete set of areas produced by such computer algorithms, to our knowledge, has been implemented anywhere in real-world regional biodiversity planning". Like Faith *et al*, we believe that this is because the optimization approach – though useful for analytic purposes -- may not always frame the problem in a politically realistic way. It has three shortcomings. First, from a political view, the problem is usually not one of minimizing the cost of achieving a set of precisely-defined objectives. Instead, it is determining what kinds of environmental benefits can be achieved with available funds and given constraints on implementation. Second, the optimization approach usually embodies, in its goals, debatable assumptions about trade-offs between different environmental benefits, or between costs and benefits. If the goal is, for instance, to ensure the representation of each of n species in two sites, then there is no benefit to adding a third site for some species, and implicitly an infinite cost to the failure to achieve a second site. In the political sphere, these costs and benefits will be debated by people with different preferences in the matter. Third, and most crucially, the optimization approach focuses on *where* to intervene, not on how to induce landholders to comply with the plan. The result is a plan that is, in theory, efficient in achieving the specified goal, but in practice may not be implementable because it relies on compulsion (which is politically costly) or on nearly universal cooperation of designated landholders (which may not be forthcoming).

In this paper we frame the problem differently, approaching it from the viewpoint of implementability and political acceptability rather than theoretical cost-efficiency. We specify a set of environmental criteria or dimensions on which to assess landscape outcomes, including representation, viability (a function of connectivity or contiguity), and resilience of biodiversity elements. We assume that higher values along each of these dimensions are preferred, but do not presume to specify trade-offs between the dimensions. We describe a class of conservation programs – voluntary responses to incentive offers – which are arguably attractive at the individual and societal level. The choice of program rules and expenditure determines individual landholder responses, which in turn shape the resulting landscape configuration. That outcome can be assessed by conservation scientists, policymakers and civil society. The question we address is not whether the resultant scores on the environmental criteria are achieved at theoretical least cost. They will not be. Rather, we pose the question of whether uncoordinated individual participation decisions can possibly yield desirable landscape-level features such as representation and viability. These features do in fact emerge in a simulation of a incentive-based voluntary program for southern Bahia, Brazil (an important biodiversity hotspot). This result stems from a correlation between low market value and remaining forest cover that may be typical of agricultural landscapes in long-settled biodiversity hotspots.

Methods of securing landholder compliance with conservation plans

Before proceeding to a description of the simulation model, we examine the implementation drawbacks of tightly prescriptive conservation plans, and why voluntary programs may be able to overcome them.

Tightly prescriptive conservation plans must secure the cooperation of particular landholders in order to meet goals such as connectivity. There are three approaches: exhortation, compulsion, and compensation. In the first approach, technical criteria are used to identify areas more or less suitable for different uses, and landholders are exhorted to hew to the recommended use. This can work when the plan provides relevant new information, or when tight informal social controls enforce a consensus that the plan supports a collective goal. In general, however, when privately profitable uses diverge from recommended ones, exhortation is insufficient to change behavior.

The second approach, typified by prescriptive zoning plans, uses the threat of legal penalties to enforce compliance with the land use plan. In practice, however, macro-scale zoning plans have proved unenforceable when they impose substantial *ex post* costs on politically powerful interest groups such as landholders or loggers. This has been the fate (to date) of two prominent statewide zoning exercises in the Brazilian Amazon (Mahar 2000, World Bank 2003). Also instructive is the fate of a technically and institutionally sophisticated effort to minimize conservation-logging tradeoffs in New South Wales (Pressey 1998). This effort involved extensive consultations between conservation and logging interests, informed by detailed data on land characteristics and a powerful decision support system. But the result of the exercise was overridden by state legislation authorizing a plan which, in conservationists' view, was inadequate to meet conservation goals. (Finkel 1998)

In the third case, the government can pay for landholder compliance. If the landholder is not obliged to accept an offer, then owners of properties with crucial locations in the reserve network may exploit their quasi-monopoly situation to demand high payments, or may simply refuse to participate. Alternatively, the government may be able to exercise the power of eminent domain – i.e., compel the landowner to sell. Certainly this approach is widely used in setting up protected areas. However, regulatory proceedings to determine fair compensation can be contentious and incur substantial overhead costs, because landholders are better informed than the purchasing authority about their lands' value (Innes *et al* 1998; Stoneham *et al.* 2003). And the use of public funds to compensate predetermined groups of landholders at individually negotiated rates may be criticized as prone to corruption.

An alternative approach to reserve system implementation starts not with a unique, prescribed configuration, but rather a set of incentive offers to a set of eligible landholders; (Ferraro 2000; Ferraro and Kiss 2002; Faith *et al.* 2003). Rather than negotiate with individual landholders, these programs offer fixed payments, or solicit auction bids, for the delivery of conservation services such as native forest protection, reforestation, and restoration of riparian vegetation (Salzman *et al* 2001). Eligible landowners voluntarily decide whether to apply for participation, and the resultant conservation network emerges as a consequence of many independent choices about participation. Examples includes the US Conservation Reserve Program (CRP) , the

Victoria (Australia) BushTender program (Stoneham *et al.* 2003), and the Costa Rica Environmental Services Payment program (Chomitz *et al.* 1999). Because they depend on voluntary responses to a rule-driven set of incentives, generating competition among landholders, such programs potentially combine transparency, simplicity, low institutional overhead, and low budgetary cost compared to a pre-designed, imposed reserve network design. Indeed, we posit that programs will be more politically acceptable, the simpler are the rules, the broader the eligible set of participants, and the more transparent and streamlined the procedure for prioritizing properties and disbursing funds.

It is not at all obvious, however, that a voluntary approach, based on uncoordinated individual actions, can satisfy the landscape-level connectivity and representation requirements of a biodiversity reserve network. Of course, hybrid systems are possible, where zoning is used to define regions in which landholders can participate in auction-like systems. Examples include tradeable development rights programs in some US counties.(Johnston and Madison 1997) But there is a tradeoff: as the zoning is more tightly constrained, representation and connectivity are theoretically more easy to achieve, but landholder compliance may be more difficult to secure, for the reasons we have described. In this paper we explore the properties of a voluntary system unconstrained by zoning.

Methods and materials

Study area

We simulated the conservation impact of a hypothetical voluntary program, similar to CRP or BushTender, on a 7.46 x 10^6 ha section of the southern coast of Bahia, Brazil. The study area constitutes an important center of endemism within the larger remaining Atlantic Forest, a biodiversity hotspot that harbors as endemics more than 2% of the world's vascular plants and vertebrates (Myers *et al.* 2000) and is often considered one of the world's highest conservation priorities (Galindo-Leal *et al.* 2003). Anthropogenic pressures have reduced the Bahian forest to 5% or less of its original area (Thomas *et al.* 1998; Saatchi *et al.* 2001).

Geographic data and assumptions

Since we lacked data on actual property boundaries, we gridded the landscape into 98 hectare land units assumed to represent properties. To assess the conservation and economic impact of alternative policies, we assembled the following geographic data for each unit:

<u>Land cover.</u> We used a land cover classification (Landau *et al.* 2003) based on 30-meter resolution Landsat data for 1996-97. The classification distinguishes anthropogenic categories including *capoeira* (forest in initial stages of regeneration from cleared land or logged forests – no continuous canopy yet formed); cocoa plantations including *cabruca*, a form of shade cocoa in which the native forest overstory is retained; eucalyptus plantations, pasture and other agriculture, and bare fields. "Mature forest" encompasses intact primary forest and regenerated secondary forest which has reached full height and

has a closed canopy; it is distinguished from *restinga* natural open vegetation, and *caatinga* (dry forest) at the edge of the study area.

Land value A land value surface was computed by Chomitz *et al.* (2005). They regressed sales prices for a sample of 231 properties on geographic characteristics of the properties, and then applied the estimated parameters to area-wide maps of those characteristics. The calibrating equation had an adjusted R^2 of 0.274, indicating that reported land values included some measurement error and the effects of some unobserved variables. Hence the imputed land value surface is smoother than the (unobtainable) actual land value surface. Importantly for the results of the present exercise, Chomitz *et al.* found that forest cover was associated with a 70% reduction in market price, holding constant soil quality, slope, road proximity and other characteristics. The presence of forest cover in this long-settled region may be a marker for poor agronomic qualities. Or its low value may reflect the operation, albeit imperfect, of regulations that restrict deforestation (and hence reduce options for land use), including a law that requires landholders to maintain 20% of each property as a forest reserve.

Bioregions Thomas & Barbosa (in press) and Veloso (1992) classified the vegetation of southern Bahia and provided the criteria for delimiting distinct floral and faunal assemblages. Using these criteria, the region was divided into eastern and western portions – areas with primarily moist tropical forest were separated from those with mostly semi-deciduous forest. Eastern Bahia is home to many species with restricted distributions (Costa *et al.* 2000, Thomas *et al.* 1998). Large rivers running west to east mark geological changes which are expressed as distinct soils, forest types, and biota (Gouvêa et al. 1976) and may directly function as barriers to vertebrate species migration (Prado, Pinto, de Moura and Landau 2003). Thus, seven bioregions were established with distinct vegetation types and biotic assemblages. (An additional bioregion, the coastal/riverine/wetland region was not included in the study because its conservation requirements are different.) Table 2 shows the name and initial forest cover in each of the studied ecozones, which are mapped in figure 1.

Policy simulation

In our hypothetical policy, a government agency with a fixed budget conducts a reverse auction. All landholders are assumed to submit bids specifying the extent and quality of forest cover on their property, and the minimum one-time payment necessary to induce them to put the property under a permanent conservation easement. We assume, conservatively, that the landholder's bid price is the market value of the land. (The bid price may be lower if the proprietor continues to enjoy benefits such as ecotourism revenue, or enhanced value of nearby residential sites.) The purchasing agency rates the environmental quality of the bid using an environmental benefit index (EBI), as in the US Conservation Reserve Program or BushTender. We used an EBI based on forest cover quality, awarding more points to mature than to secondary forest, but more complex EBIs could be defined. The agency ranks bids using a cost-effectiveness index that divides EBI-weighted area by bid price. Conservation easements are purchased in descending order, at each landholder's bid price until the budget is exhausted. With these assumptions, the budgetary or fiscal cost is the same as the social or opportunity cost of conservation.

Auction systems may however not be fully successful at eliciting landholders' private information about the value of their land, especially if the auction is repeated. (Smith 1995). Landholders with low value land or with high personal preferences for conservation may bid strategically, asking for prices above their opportunity cost of farming, thus capturing information rents (Smith and Shogren 2002). Stoneham et al (2003), analyzing actual bid data in a conservation auction, show that bidders are far from capturing all available rents. However, for comparison we evaluated the budgetary outlay under the fixed-price offer system that corresponds to each auction scenario. Under the fixed-price offer system, the purchasing authority offers landholders a fixed payment per EBI-weighted hectare to put their property under a conservation easement. If this offer is set at the same level as the highest accepted bid (per EBI-weighted hectare) as an auction scenario, it will elicit the same participants as the auction, and would have the same social opportunity cost. However, inframarginal bidders would receive rents equivalent to the difference between their opportunity cost and the offer. Such a scheme is simpler than an auction program, and therefore may be attractive on political grounds even though it involves greater expenditure by the purchasing authority. (Costa Rica's Environmental Services Payment program, for instance, employs fixed payments despite the potential efficiency advantages of differentiated payments.) The expenditures under this scheme can be viewed also as the expenditures that would result from an auction scheme in which strategic bidders managed to capture all information rents.

The simulation of landholder response and associated land use configurations and payments was performed using the Toolbox of Applied Metrics and Analysis of Regional Incentives (TAMARIN) (Stoms *et al.* 2004). This software program, an add-on to

Arcview, tracks land cover and land value in a gridded representation of the landscape. It calculates which 'properties' respond to incentive offers, uses simple decision rules to assign land cover outcomes based on incentive responses, and assesses the connectivity of the resultant landscape. Documentation for the program and information on how to obtain it are available at www.tamarinmodel.org. The program is available free of charge.

Evaluation Criteria

Assessing the impact of the policy requires assuming a baseline scenario for land use change. We lack the data to estimate statistically a land use change model. Furthermore, we believe that policymakers, motivated by a precautionary principle and facing an uncertain future, might adopt a pessimistic baseline scenario when considering the long-term survival of an irreplaceable ecosystem. We therefore assume that unprotected mature forest areas will face continuing pressures from pasture expansion, subsistence agriculture, and timber extraction, degrading into secondary vegetation. For land units enrolled in the conservation program (or are already in protected status), we assume that mature forest is retained, existing agriculture is abandoned in favor of forest regeneration, and both *capoeira* and agricultural lands are designated as 'regenerating forest' which, over time, will develop into closed forest, become more biodiverse, and ultimately resemble mature forest. Consistent with evidence (Guevara and Laborde 1993; Landau 2001; Martini *et al.* in press) we assume that unassisted regeneration will proceed naturally within land units that have existing seed sources.

Our evaluation criteria differ from those commonly used in conservation planning, because we frame the problem differently. Conservation plans typically define a fixed environmental goal, based on implicit weightings of the relative importance of representation, redundancy, and resilience in the reserve network. The plans then seek to minimize the cost of achieving that specified goal. In our framework, the public chocie variables are the size of the budget allocated to the program, and the prioritization rules. The resulting landscapes were evaluated on four principal conservation criteria defined at greater length below: viability, representation, and redundancy of surviving forest fragments, and the proportion of surviving forest free of edge effects. The literature does not give clear guidance on the relative importance of these criteria (Stoms et al. 2004) and so we do not aggregate them into a unidimensional index.

Forest fragments are defined as contiguous assemblages of mature or regenerating forest, allowing for gaps of up to 500 meters (for pasture, crops and bare land) or 1000 meters of secondary forest or shaded cocoa (cocoa grown under shade trees). A fragment is deemed *viable* if it is at least 10,000 hectares in extent, based on simulations (Paglia 2003) of extinction probabilities for *Cebus xanthosternos*, a large endemic primate that is one of the most area-demanding endemic species of southern Bahia. Viability is thus a function of connectivity and contiguity. *Representation* is gauged by the number of distinct bioregions that exhibit viable fragments. *Redundancy* is measured by the number of viable fragments within a bioregion. *Edge forest* is defined as that within 300 meters of agriculture other than shaded cocoa or plantations. For the purposes of this study, we excluded the coastal and semi-arid bioregions because their conservation considerations and anthropogenic pressure differ considerably from the those of humid forests not adjacent to the coast.

Results

In the absence of any intervention, future forest area persists and regenerates only within the current protected areas and is predicted to be 85,000 hectares. Of this 73% is within fragments deemed viable (Figure 1). For a hypothetical budget of R$20 million (at the time of the land value study, US$1=R$1.80 approximately), future forest area more than doubles to reach 175,000 ha, of which the proportion in viable fragments is about 50%. Rising budgets induce the expansion and coalescence of large fragments more rapidly than enrollment of unconnected small ones (See Figure 1 and Table 1). As a consequence, the proportion in viable fragments increases to 63.4% at R$80 million . Thereafter, as budgets increase to R$200 million, the proportion stays between 59% and 62%. The proportion of total budget going to these larger fragments rises to 56% at R$80 million and then declines to 51%.

These large fragments are of relatively high habitat quality. The proportion of edge forest in these fragments increases from 13.4% at R$20 million to 22.6% at R$80 million, holding approximately constant at higher budgets. Non-edge mature forest declines gradually from 62.7% at R$20 million to 54.5% at $R200 million.

Small fragments, of less than 200 ha, increase as a proportion of total surviving forest area from 6.1% at R$20 million to 7.8% at R$60 million, holding approximately constant at higher budget levels. They account for a disproportionate but overall small proportion of the budget, ranging from 13.4% to 9.9% of the total. Their conservation potential is low; as budgets increase, the total area of these fragments increases from about 11000 to 49000 hectares, but the portion of these fragments in non-edge mature forest declines from 46.4% to 12.6%.

Overall representation and redundancy increase with budgets up to a point (Table 2). In the baseline, business-as-usual scenario, existing protected areas contain four viable fragments in two of the seven bioregions. At R$80 million, coverage increases to eleven viable fragments in four bioregions. At R$160 million, 14 viable fragments in 5 bioregions are secured.

The number of viable fragments in a bioregion is related to the initial extent and proportion of mature forest (Table 2). Holding budget constant, the number of viable fragments tends to increase as the initial mature forest area increases. The exception is the central semideciduous zone, which contains about 80,000 ha of mature forest, but where this forest is highly fragmented, scattered over a relatively large area, and where bid prices are relatively high due to the lower average level of forest cover. This bioregion does not acquire a viable fragment even in the highest budget scenario, when 44% of its original mature forest is placed under protection.

For each auction scenario we calculate also the equivalent budgetary outlay for the equivalent fixed-price offer scheme. Under this kind of program, the budgetary outlay is 30% to 90% higher than in the corresponding auction scheme; the percentage increases with program size. The implied one-time per-hectare payments are still modest compared to the annual payments under the Costa Rican, US CRP or BushTender programs.

Discussion

In principle, optimization techniques can be used to design a biodiversity reserve network that minimizes the social cost associated with a set of ecological goals such as connectivity, representation, and low ratios of edge to interior forest. These techniques may be particularly valuable in identifying irreplaceable areas of high biodiversity value for inclusion in a network. However, it may be both difficult and unnecessary to impose a tightly constrained reserve network plan on a landscape of unwilling landholders.

Voluntary incentive programs offer potentially greater political acceptability but do not automatically guarantee representation or viability of within the resultant reserve network. However, our results suggest that a voluntary incentive program, with simple, property-specific enrollment criteria, could generate a landscape-level biodiversity reserve network that represents a significant range of the Bahia Atlantic Forest's biodiversity with resilience and redundancy. It does so, furthermore, at relatively low social cost and with relatively high environmental efficiency – for instance, with about 90% of the funds devoted to patches of greater than 200 hectares, which are likely to be more persistent and less subject to edge effects than smaller patches. Connectivity among existing and regenerated forest fragments arises without central planning or costly and time-consuming negotiation with individual landholders. As has been demonstrated with random binary maps (Gardner *et al.* 1987), connectivity is achieved when the proportion of suitable habitat exceeds a landscape-specific threshold. In our simulation, higher payment offers increase the local proportion of planning units under conservation, thus breaching local thresholds for connectivity. Note, however, that, consistent with the random binary map model, the voluntary program fails to create viable patches when the initial proportion of forest fragments is very low (in the central semi-deciduous bioregion).

The results for southern Bahia are the consequence of a strong inverse correlation between forest cover and land value, and spatial autocorrelation of both these variables. These correlations may arise as the joint consequence of typical biophysical landscape features (patchiness of soil types and slopes) and typical economic processes of deforestation (preferential deforestation of more accessible, better quality land). They may also result from partial enforcement of regulations against deforestation. These features may generalize to other hotspot areas where severe habitat fragmentation has prompted consideration of biodiversity corridor construction.

The approach outlined here will not, however, be universally applicable. It will be inappropriate where conservation considerations leave little room for flexibility – for instance, for maintenance of areas with irreplaceable biodiversity, or with nonsubstitutable environmental functions such as riverine forest. It will fail to generate contiguous areas where forest cover is spatially autocorrelated but land value is not.

We did not compare the efficiency of our simulations to an 'optimal' reserve network , because we do not think the latter is implementable. But it is fair to ask whether auction-type programs are themselves implementable. The conservative assumptions used here may greatly overstate the actual opportunity cost of the program, especially if ecotourism and other non-exploitative land uses are developed. We have not however, accounted for recurrent costs of monitoring and enforcement of the conservation agreements. The social acceptability of the program depends on the incidence of costs and benefits. Because forest land is disproportionately held by large landholders in Bahia, payment recipients under the hypothetical scheme would probably tend to be comparatively wealthy; if so equity would require that the program be financed through taxes that fell more heavily on wealthier citizens. For instance, the program could be financed via a tax on other wealthy landholders (those out of compliance with the legal forest reserve obligation, for example – see Chomitz 2004) or through national or international payments for ecosystem services. The program incorporates low-value land and thus would not tend to frustrate aspirations of landless people to obtain land through land reform. Finally, implementation of such a program would require a sophisticated institution for paying and contracting with landholders and monitoring and enforcing the easement contracts. Recently established programs in Costa Rica and Mexico provide potential models for study in the developing world.

An area for further research is the potential for increasing the program's efficiency through fine-tuning of the eligibility, payment, and prioritization rules. For instance, zoning could be used to restrict eligibility to areas of known endemism, or to exclude areas where forest cover is so low that connectivity is difficult. The environmental benefit index could be modified to include a measure of the complementarity of a plot's biodiversity to that in the existing reserve system, following Faith *et al.* (2003), or the potential for connectivity, based on proximity to forests on other properties. To increase connectivity, the payment scheme could be modified to include the 'agglomeration bonus' suggested by Parkhurst *et al* (2002), where landholders receive a premium if their neighbors also enroll. Separate budgets could be allocated for each bioregion. All these approaches potentially incur political costs because of their increased complexity or because they are seen *a priori* to favor certain geographical regions; these potential drawbacks have to be balanced against the possibility of increased cost-effectiveness.

Hotspots hold much of the irreplaceable global biodiversity, in addition to being highly threatened (Rodrigues 2004). The results of our study point to a promising way to implement incentive agreements with potentially far reaching biodiversity conservation benefits. Existing and proposed funding mechanisms to address the Millennium Development Goal for environmental sustainability --calling for significant reduction in current rates of biodiversity loss by 2010 -- could, given adequate institutional arrangements, use these types of approaches to achieve more efficient results in biodiversity hotspots located in developing countries.

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Figure 1 Conserved forest fragments by budget scenario

Table 1 Distribution of conserved forest area by budget scenario (R$$10^8$) and fragment size class

Table 2 Viable fragments by budget scenario and bioregion

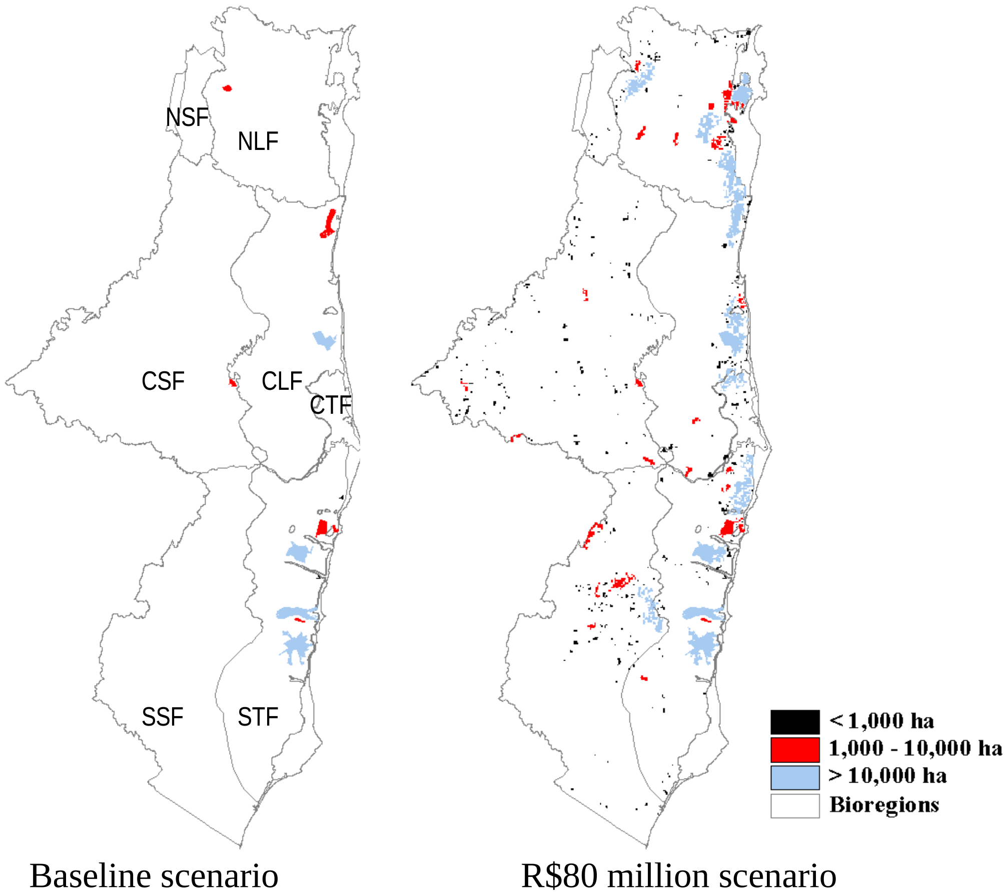

Figure 1 Conserved forest fragments by budget scenario

Table 1. Distribution of conserved forest area by budget scenario (R$$10^8$) and fragment size class

	BUDGET SCENARIO (R$$10^6$)					
FRAGMENT SIZE CLASS	**0**	**40**	**80**	**120**	**160**	**200**
< 200 ha	84	16129	26702	34413	40880	49115
200-500 ha	393	14049	18717	31284	37198	40720
500-1000 ha	550	14636	22762	28663	36291	42068
1000-2000 ha	4424	17227	26300	35523	45282	46235
2000-4000 ha	2109	24809	6848	18020	29085	55604
4000-6000 ha	0	5195	14941	4700	4294	0
6000-8000 ha	6219	7056	6552	12544	21957	14371
8000-10000 ha	9092	8614	8134	9000	0	9115
> 10000 ha	62050	134929	226915	282181	330339	370876
Sum	**84921**	**242642**	**357871**	**456328**	**545325**	**628103**

Table 2 Viable fragments by bioregion and budget scenario

BIOREGION	**Initial Mature forest area**	**Initial ratio, mature forest/total area**	**Viable fragments by scenario (R$$10^6$)**			
			R$0	R$80	R$160	R$200
Northern Semi-deciduous Forest (NSF)	4107	2.7%	0	0	0	0
Central Tabuleiro Forest (CTF)	18662	16.0%	0	0	1	1
Southern Semi-deciduous Forest (SSF)	72336	4.6%	0	1	2	2
Central Semi-deciduous Forest (CSF)	78469	3.8%	0	0	0	0
Central Lowland Forest (CLF)	82486	7.7%	1	2	2	2
Northern Lowland Forest (NLF)	100294	10.8%	0	4	4	4
Southern Tabuleiro Forest (STF)	145656	13.3%	3	4	5	4