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A SYSTEMATIC EVALUATION OF THE CONSERVATION PLANS FOR THE PANTANAL WETLAND IN BRAZIL

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Abstract: The Pantanal is the world's largest contiguous freshwater wetland spanning Brazil, Bolivia, and Paraguay. It contains the greatest wildlife densities in the Neotropics and was enlisted by all three countries in the Ramsar convention on wetland conservation. The Brazilian government, together the UNESCO's Man and the Biosphere Program, declared a biosphere reserve in the Pantanal in 2000. Other plans to protect the region include expansion of existing reserves and land use regulations following recommendations from the Cerrado-Pantanal priority setting workshop. Here we evaluated how four conservation scenarios complied with the principles of systematic conservation planning and analyzed their representativeness, efficiency, and complementarity using 17 vegetation classes as surrogates for regional biodiversity. We used MARXAN (systematic conservation planning software) to determine the value of the habitat types protected by each conservation scenario. We found that none of the four conservation scenarios met preferred areal targets for protection of habitats, nor did any protect all 17 biodiversity surrogates. The Pantanal Biosphere Reserve provided the best compromise in conservation planning.

Key Words: Biosphere Reserve, CLUZ, efficiency, irreplaceability, MARXAN

INTRODUCTION

Conservation scientists have been advocating a systematic approach to conservation planning for the last 30 years (Margules et al. 1982, Kirkpatrick 1983). Some influential publications set the framework (Margules et al. 2002) and principles (Pressey et al. 1993, JANIS 1997) to which conservation planning should adhere to avoid the inefficiencies and pitfalls of ad hoc biodiversity conservation (Pressey 1994). Considering the scarcity of resources for biodiversity conservation in developing nations, planners must carefully evaluate the effectiveness of alternative conservation actions. Maximizing effectiveness of reserve systems entails the implementation of the CARE principles: Comprehensiveness, Adequacy, Representation, and Efficiency (Possingham et al. 2006a). Using those CARE principles, we evaluate existing and proposed conservation plans for the Pantanal in Brazil.

How well is the Pantanal protected now? A review of the Brazilian protected area system (Rylands and Brandon 2005) showed that reserve creation has been biased towards the Amazon where 80% of the country's reserves are concentrated. Reserves in other biomes, such as the Cerrado and Pantanal, have their own spatial biases, generated by the competition for different land uses (Machado et al. 2004) whereby conservation states are assigned to land not useful for agriculture, housing, and other human activities. This phenomena is referred as the worthless-land hypothesis (Pressey et al. 1993). According to Rylands and Brandon (2005) the Pantanal is one of the worst protected ecosystems with respect to IUCN categories I to III with less than 5% of the ecosystem in reserves. This poor performance has been justified by a widely accepted belief that the Pantanal region is protected by several factors: its traditional farmers, flooding, and remoteness (Mittermeier 1990, Harris et al. 2005). Locally the ad hoc selection of protected areas generated collections of reserves rather than a functional reserve system, under the CARE principles. The ad hoc selection of reserves and a false impression of protection have delayed implementation of representative reserves and adherence to the Ramsar convention (1971), signed by Brazil only in 1993.

In an attempt to correct these flaws, the Brazilian Biodiversity Program (PROBIO) identified 900 priority areas through the national biodiversity strategy (Ministério do Meio Ambiente 2002) that were used to align Brazil with the CBD (Convention on Biological Diversity 2004). The recommendations from the Cerrado-Pantanal Priority Setting Workshop (CP-PSW) suggested two major actions to protect the Pantanal wetland system. The first was to secure that each Pantanal subregion was represented in the reserve system. The second aimed to protect riverine functional structures (e.g., headwater springs, oxbows, swamps) across all subcatchments (Ministério do Meio Ambiente 1999). These recommendations served as official guidelines to all government levels. Although actions advanced biodiversity policies, they lacked a systematic framework of analysis, and disregarded conservation targets and implementation costs (Margules and Pressey 2000).

Developing a comprehensive and efficient framework to protect the largest and the most biodiverse contiguous freshwater wetland of the world is imperative (Mittermeier 1990, Por 1995, Junk et al. 2006a). Yet reviews of regional conservation strategies have shown a slow progress toward systematic planning (Pott and Pott 2004, Harris et al. 2005, Higgins et al. 2005, Junk and Nunes da Cunha 2005, Junk et al. 2006b, Ministério do Meio Ambiente 2007). The lack of a decisive implementation policy presents a great challenge to conservationists in the near future.

The aim of this paper is to investigate the ability of regional conservation plans to meet goals for biodiversity representation and cost-efficiency (i.e., minimizing total area in a plan) (Possingham et al. 2006a). We first assess how much of the existing vegetation classes are currently represented in each scenario (conservation plan) and Pantanal subregion. We then evaluate whether the current and projected conservation plans meet the chosen representation target. Finally we present options to improve the comprehensiveness of these scenarios based on the rarity and vulnerability of each biodiversity surrogate.

METHODS

The first step in the evaluation process was to characterize the study region and the conservation plans (scenarios) assigned to the Pantanal. Then we identified vegetation classes capable to represent regional biodiversity (Da Silva et al. 2000), assigning the contribution of each vegetation class (in hectares) to each planning unit (Margules et al. 2002). We assessed the performance of 4 conservation scenarios proposed under the umbrella of the ministry for the environment of Brazil to safeguard the Pantanal. We also considered how each of the 10 Pantanal sub regions were represented by these scenarios, to uphold the recommendations of the CP-PSW (1999). Finally we evaluated the issues of regional biodiversity representation (using surrogates and targets), relative efficiency, and opportunity cost in all scenarios.

Study Region

The Pantanal wetland is part of the upper Paraguay River basin, a 365,000 km^2 watershed. The Pantanal wetland in Brazil accounts for 70% of the floodable area of the watershed (140,000 km^2) and is covered by a mosaic of vegetation types influenced by floods, biological succession, biogeography, soil types, and human use (Prance and Schaller 1982, Pott and Pott 1994, Pinder and Rosso 1998, Damasceno-Junior et al. 2005). Ten different subregions (Da Silva and Abdon 1998) are commonly identified in the Pantanal based on different combinations of biophysical processes and attributes (Table 1). The floodplain is home to more than 2,500 species of plants, 464 birds, 124 mammals, 177 reptiles, 41 amphibians, and 325 fishes (Junk et al. 2006b), making it one of the most species rich wetlands in the world (Junk et al. 2006a).

Currently the floodplain is mostly used for ranching, fishing, and tourism. These activities are perceived to offer little impact on its biodiversity when compared to their impacts in other Brazilian ecosystems (Da Silva et al. 2005, Harris et al. 2005). However, recent studies are showing that ranching and farming intensification has become a major cause for recent deforestation in and around the floodplain (Seidl et al. 2001, Padovani et al. 2004). Commercial and sport fishing are blamed for depleted fish stocks (Catella et al. 1999). The tourism industry is still to be assessed regarding its social and environmental impacts.

Table 1. Percentage of Pantanal subregions covered by each conservation plan (scenario) where scenario (1) represents the combined existing public and private reserves, scenario (2) represents high-importance core areas from CP-Priority Setting Workshop (CP-PSW), scenario (3) represents Pantanal Biosphere Reserve nuclei and buffer zones, and scenario (4) represents high-importance core areas and connecting corridors of the CP-PSW.

		Average Percent Area included in each Scenario			
Pantanal subregion	Total area (ha)	Scenario (1)	Scenario (2)	Scenario (3)	Scenario (4)
1 Abobral	426,580	20.9	26.5	43.8	10.0
2 Aquidauana	527,580	0.0	4.8	0.0	21.8
3 Barão de Melgaço	1,810,140	5.4	21.6	15.6	63.2
4 Cáceres	1,190,820	0.8	1.1	7.3	18.2
5 Miranda	508,160	10.5	0.9	50.4	10.4
6 Nabileque	1,257,610	1.5	2.5	42.5	29.8
7 Nhecolândia	2,619,330	1.9	8.6	25.2	27.0
8 Paiaguás	2,657,950	0.8	5.0	1.3	25.1
9 Paraguai	940,520	30.4	30.8	39.9	93.8
10 Poconé	1,607,408	2.0	15.4	69.6	33.6
Totals	13,546,098	7.4	11.7	29.6	33.3

The majority of threats to the Pantanal are generated outside the region (e.g., siltation, pollution and water diversion), but several authors also ascribe some of the internal threats (i.e., expansion of cultivated pasture, unsustainable logging, and invasive species) to globalization (Hamilton 1999, Harris et al. 2005, Junk and Nunes da Cunha 2005). The free trade agreement between Brazil and its southern neighbors (MERCOSUR) is acknowledged as an emerging threat to regional biodiversity (Hamilton 1999, Lourival et al. 1999, Junk and Nunes da Cunha 2005). Intensification of agriculture, mining, and industries associated with processing of primary products are expected to increase water diversion and dam construction, altering wetland habitats in some sub-catchments, and boosting erosion, siltation, and chemical pollution in others (Seidl 2001).

Planning Units

We transposed the results of vegetation mapping based on aerial surveys to spatially assign plant contributions to 1,232 planning units, defined as square grid cells with a maximum size of 10,000 ha. We adjusted the grid to have the majority of the planning units represented by a centroid of 10 sample points per planning unit. The centroid summarized the contribution of each biodiversity surrogate (i.e., vegetation class) as the proportion of the total area occupied by each vegetation class in a planning unit (Da Silva et al. 2000). Non-sampled planning units (< 1%), had their centroid values interpolated from neighboring sampled planning units, using ArcMap 9.2 software. One of the limitations of the aerial surveys, however, was that aquatic habitat diversity was underrepresented because those habitats were not a major focus of the survey.

Biodiversity Surrogates

The use of surrogates to represent biodiversity, although controversial, seems appropriate when biodiversity information is scarce (Rodrigues et al. 1999, Ward et al. 1999, Sarkar et al. 2005). Results of studies in South Africa showed that vegetation is one of the best and most convenient surrogates for terrestrial biodiversity (Driver et al. 2003, Lombard et al. 2003). Nevertheless, congruent relationships between species/community and biodiversity surrogate distributions are essential (Ferrier 2002). In the case of the Pantanal, several studies demonstrated that wildlife densities, reproductive behavior, and seasonal migrations are correlated with flood regimes and the structure of the landscape (Coutinho and Campos 1996, Mourão et al. 2000, Campos et al. 2003, Pott and Pott 2004, Junk et al. 2006b).

We based our analysis on 17 vegetation classes as biodiversity surrogates (Figure 1). Vegetation maps were constructed in ArcMap 9.2 based on data from a series of aerial surveys conducted during the 1990s (Da Silva and Abdon 1998, Da Silva et al. 2000). The surveys offer a comprehensive effort to directly sample the Pantanal vegetation communities, while their long term nature and the precautions described in the experimental design seems to minimize much of the spatial bias that is common to ground surveys under restricted accessibility (Caughley 1974, Mourão et al. 1994).

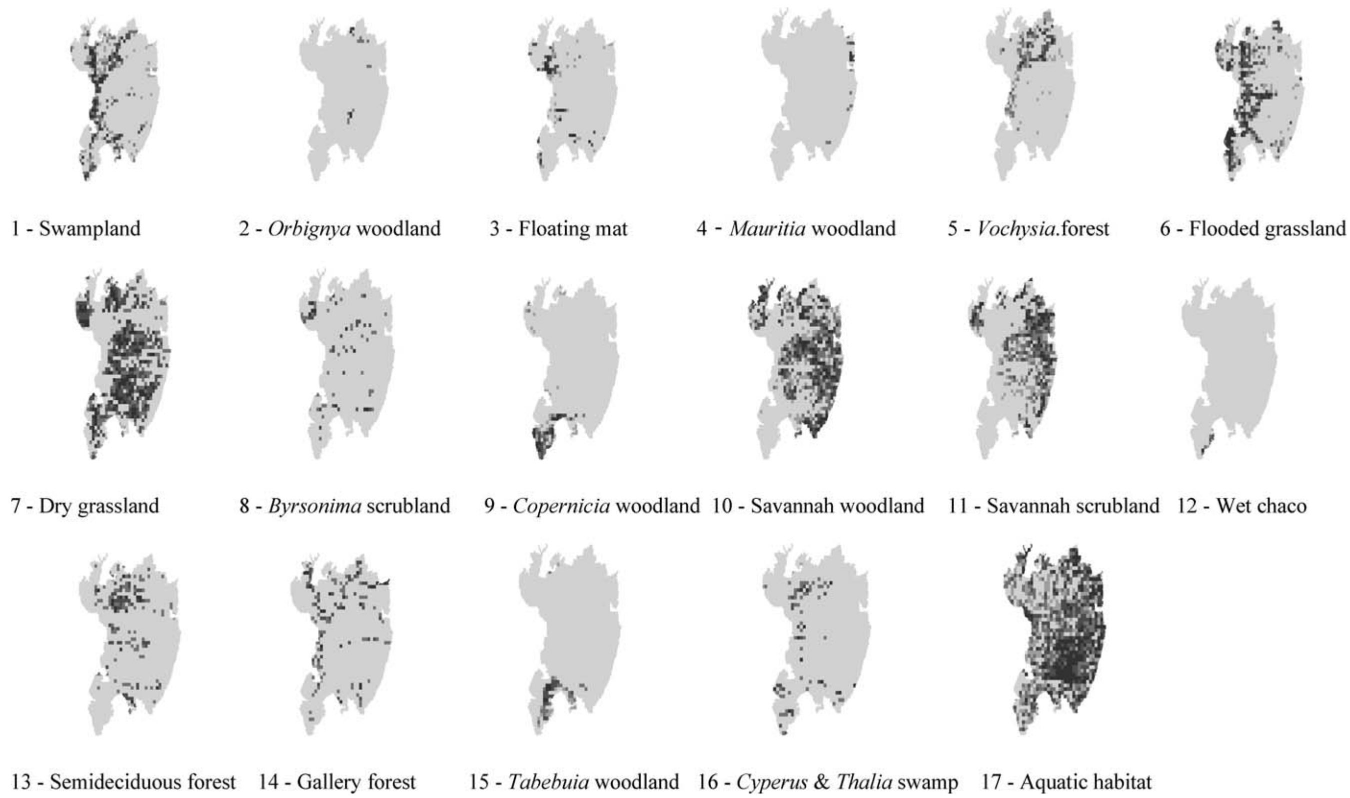

Figure 1. Distribution of vegetation classes in the Pantanal in the same order they appear in Table 2. Tones of grey represent proportion of the area occupied by that vegetation class in a planning unit, the darker the grey the more dominant a vegetation class is in that sample station (Da Silva et al. 2000).

Existing and Proposed Conservation Plans (Scenarios)

We characterized conservation plans for the Pantanal into 4 scenarios. Scenario (1) combined existing public and private protected areas, altogether 19 reserves (3 public and 16 private) covering 132 planning units. Scenario (2) protected core areas identified in the Cerrado-Pantanal workshop (CP-PSW) distributed by 9 core areas covering 194 planning units. Scenario (3) included nuclei and buffer zones of the Pantanal Biosphere Reserve with 470 planning units, although we excluded transition zones because of a lack of clarity in their contribution to conservation targets. Scenario (4) protected core areas and the connecting corridors devised at the CP-PSW with 627 planning units made up of 194 planning units with an additional 433 units as part of connecting corridors (Figure 2).

Biodiversity Representation Targets

To evaluate the Pantanal conservation plans transparently, we set conservation targets based on current Brazilian legislation as well as using targets suggested by international agreements such as the Convention on Biological Diversity-CBD (2004) and Convention on Wetland Conservation (RAMSAR 2007). For the first we chose the Brazilian forest code (act number 7.803) that defined for the central savannahs of the country, including the Pantanal, mandatory reserves of at least 20% of the total rural property (Ministério do Meio Ambiente 1989). For the Convention on Biological Diversity we choose a 10% target and for broader protection goals such as for the Ramsar Convention and Biosphere Reserves, we adopted a 50% representation target (Table 1). We converted percentage targets into number of hectares required for minimum representation of each biodiversity surrogate (vegetation class) and Pantanal sub-region (regional landscape unit).

We imposed a weight on the 10% target, to proportionally represent biodiversity surrogates rarity and vulnerability (Pressey and Taffs 2001a), as proposed by the 8th convention of the parties (COP8) of the CBD. We used the following equation:

$$\text{Revised Target for biodiversity surrogate i} = 10\% + (10\%\mathrm{NR_i}) + (10\%\mathrm{V_i}) \quad (1)$$

where NR_i is the natural rarity of biodiversity

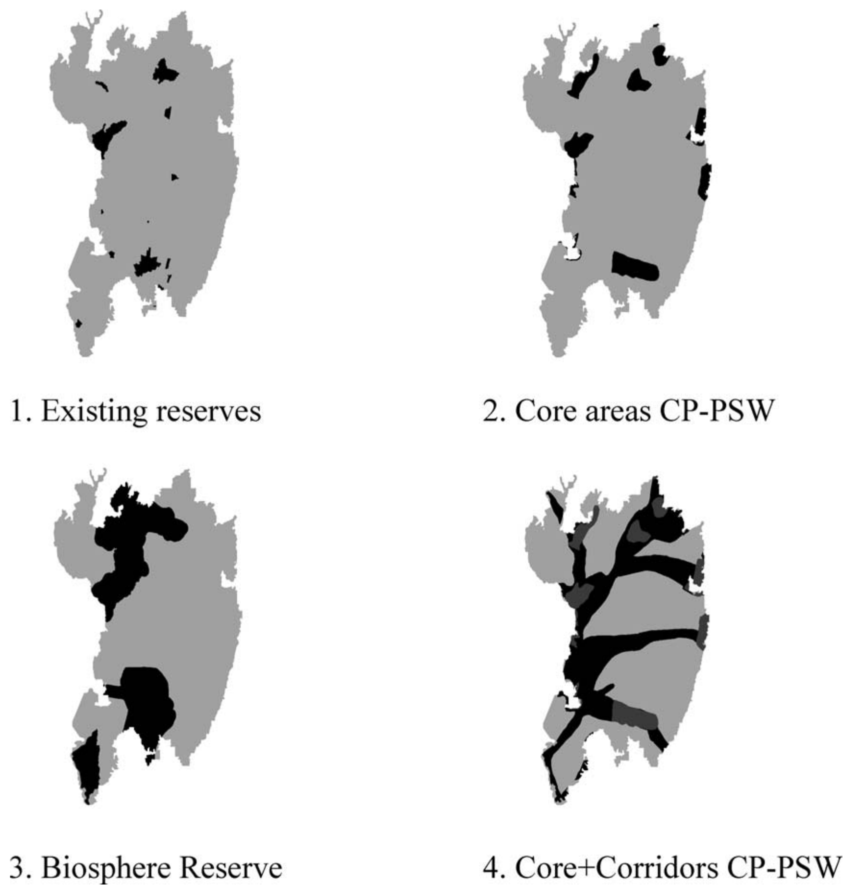

Figure 2. Existing and proposed conservation planning scenarios for the Pantanal floodplain: scenario (1), the combined existing public and private reserves; scenario (2), high-importance core areas from CP-Priority Setting Workshop (CP-PSW); scenario (3), Pantanal Biosphere Reserve (nuclei and buffer zones); and scenario (4), high-importance core areas and connecting corridors of the CP-PSW.

surrogate i, which varies from 0 to 1, and parameter V_i represents vulnerability as determined by the average susceptibility of the biodiversity surrogate to threats such as fire, agriculture/ranching, and water control (i.e., 0 = not vulnerable, 1 = low vulnerability, 2 = moderate vulnerability, and 3 = high vulnerability). NR_i is calculated by:

$$NR_i = [(A_{max} - A_i)/A_{max}], \quad (2)$$

where A_{max} is area required by the most extensive biodiversity surrogate and A_i the area required for the biodiversity surrogate being considered. Using the rarity/vulnerability weighting metrics (Eq. 2), the 10% target for all biodiversity surrogates increased the area to be protected to 29.2% of the Pantanal surface (or 1,057,013 ha). After applying weights, the representation requirement fluctuated between 20 and 39% with an average of 32%. The weighting process significantly enhanced representation of less extensive and more threatened vegetation classes. For the 20% fixed target we only used the value derived from the Forestry code (Ministério do Meio Ambiente 1989). The 50% target was used to determine the performance of the scenarios appropriate for multiple-use zoning, such as Ramsar sites, biosphere reserves, and national heritage sites. In setting those targets we assumed an equal responsibility between the public and private sectors to protect biodiversity, as stated in the Brazilian Constitution (Governo da República Federativa do Brasil 1988).

Relative Efficiency

In the Pantanal case study, we evaluate effectiveness of conservation plans combining the measures of efficiency (Pressey and Taffs 2001b) with measures of opportunity cost (Stewart et al. 2003), where the cost of a planning unit is represented by the area of that planning unit (Bedward et al. 1992). However, the way that efficiency was defined (see Eq. 3 following) does not appropriately convey the matter of off-reserve conservation, considering that some scenarios were composed of off-reserve planning units and their contribution to target achievement was considerable. More importantly, implementation costs tend to be significantly lower than public reserves since there are no acquisition costs. We adapted equation (3) to accommodate scenarios where reserved and off-reserve planning units (corridors and buffer zones) are considered independently. Our measure of relative efficiency is described by equation (4) and is based on cost-weighting factors that allow for rebates in the cost surface to accommodate multiple ownership schemes and zoning. In both cases Efficiency (E) and Relative Efficiency (RE) values range from 0 to 1 such that efficient systems are closer to 1 and inefficient systems are closer to 0. The equation for conservation efficiency is:

$$\mathrm{Efficiency(E)} = 1 - \mathrm{X/T}, \quad (3)$$

where X is cost of all planning units in the conservation scenario and T is total cost of all planning units available to be chosen (Pressey and Nicholls 1989). The equation to calculate relative efficiency (RE) assesses the efficiency of each conservation scenario, while accounting for the costs and benefits of off-reserve conservation, and is:

$$\mathrm{Relative\ Efficiency(RE)} = 1 - \left(\sum_{i=1}^{n} y_i w_i \Big/ T_i \right), \quad (4)$$

where, n is the number of planning units, y_i equals the cost of selected planning units ($y_i = 0$ if planning unit i is not in a scenario), and the parameter w_i is the weight given to planning unit when $y_i \neq 0$. This varies according to its ownership status or management regime ranging from 0 to 1. The denominator term is represented by T which is the total number of planning units available to be chosen.

Ownership and management regimes influence reservation costs in a variety of ways and determine how governments allocate their conservation budgets (Cantrell 1980, Drechsler et al. 2006). Traditionally, cost evaluation in conservation planning has focused on considering publicly funded reserves systems where decisions were driven entirely by governments. Recently, however, systematic conservation and biodiversity protection has also become a matter of private interest and off-reserve conservation has become an essential element for long term persistence of biodiversity. In such cases public or private ownership represents differential acquisition and management costs ($w_i \leq 1$). Regardless of the assumption that off-reserve conservation is less expensive to governments than public reserves, they still represent costs either via tax incentives or direct investments (Wu and Babcock 1996). We assumed that governments are supposed to provide a portion of the costs when dealing with privately owned reserves and multi-zoning conservation plans. Therefore relative efficiency can then be used to account for these differences in implementation and management costs. In our sensitivity analysis, all planning units in public reserves, regardless of the scenario, represent full cost, so that its weight is one ($w_i = 1$). Whereas planning units that are managed in the form of private reserves, buffer zones, or corridors have their cost multiplied by a weighting factor ($0 < w_i < 1$), which represents the proportion of rebate in the cost of a planning unit that belongs to a scenario but is privately owned.

Selection Frequency and Irreplaceability

We used the reserve selection tool MARXAN (Ball and Possingham 2000) and its companion front-end software CLUZ (http://www.mosaic-conservation.org/cluz /index.html) to choose a set of planning units that meets the targets for all the biodiversity surrogates for the lowest total cost (Possingham et al. 2000). Among all the available MARXAN solutions, we chose the output that best summarized each planning units' selection frequency; this output has been considered a good approximation of a planning units' irreplaceability, referred henceforth as the irreplaceability score or just irreplaceability (Stewart and Possingham 2005, Carwardine et al. 2007).

We categorized irreplaceability of each planning unit into three groups based on their selection frequency: 1) *replaceable*, when selection frequency was smaller than the mean frequency of a random selection, 2) *negotiable*, when selection frequency ranged between the mean and one standard deviation, and 3) *irreplaceable*, when selection frequency was higher than one standard deviation above the mean (Stewart et al. 2003). It is important to mention that we used negotiable planning units in two contexts, to identify planning units that had the potential of contributing to off-reserve conservation such as buffer zones and corridors, and to avoid the bias imposed by locked-in planning units to a particular scenario. Locking-in planning units forces MARXAN to include them into all solution by selecting them 100% of the time. In such cases planning units with mediocre contributions to targets become irreplaceable. Thus, we did not lock-in planning units during the scenario evaluation (20% target), but locked them in plans when we proposed improvements to existing scenarios (10% weighted target). In such cases the amount of area needed to reach the target was considerably larger than what was held by some scenarios. Therefore irreplaceable planning units (which were still outside that scenario) needed to be incorporated in the form of new protected areas, while negotiable parcels could be used in the buffer zones or any other off-reserve mechanism.

Opportunity Cost

To evaluate opportunities lost when the scenarios under analysis were defined, we used the irreplaceability scores generated by MARXAN software to determine the opportunity cost of each scenario (Stewart et al. 2003). Opportunity cost was determined by summing the proportion of replaceable planning units included in a scenario and the proportion of irreplaceable planning units left outside of that same scenario. It is an objective way to measure the performance of any conservation plan because it quantifies the inefficiencies associated with ad hoc conservation.

RESULTS

Sub-regional Representation

We mapped the Pantanal sub-regions in order to have an idea of how much of each subregion was represented within each of the conservation scenarios (Table 1). Existing reserves covered an average of 7.4% of each subregion. The core areas of CP-PSW covered 11.7% of them. The biosphere reserve nuclei and buffers covered an average of 29.6% of subregions and the core and corridors of the CP-PSW averaged 33.3% of them. However when the results were inspected closely (Table 2), the existing protected areas failed to meet a representation target of 20% in 8 of 10 subregions, while the core areas of the CP-PSW failed to meet the targets for 7 subregions. The biosphere reserve did not reach the 20% target in 4 subregions, while the core and corridors of the CP-PSW scenario did not represent the Abobral, Cáceres, and Miranda subregions at the 20% level. We found that the Aquidauana subregion was not represented either in the existing protected areas or in the nuclei or buffer of the Biosphere reserve (Table 1).

Biodiversity Surrogate Representation

Of 204 possible combinations of 17 vegetation classes, 4 conservation scenarios, and 3 representation targets, 96 of them (47%) had biodiversity surrogates that were not represented at the target levels (Table 2). Only 3 vegetation classes achieved full representation (swamplands, floating mats, and flooded grasslands) across all scenarios and targets. Habitats that have a restricted range in the floodplain, such as palm woodlands and semi-deciduous forests were not represented at any targets level by the existing protected areas nor by the CP-PSW core areas. In fact those 2 scenarios (1 and 2) failed to represent respectively 12 and 10 of the 17 surrogate classes for all targets. The biosphere reserve and the CP-PSW core and corridors scenarios (3 and 4) were only able to represent 6 of the 17 biodiversity surrogates at all 3 targets.

The nuclei and buffer zones of the biosphere reserve scenario 3 completely failed to represent savannah woodlands and scrublands habitats. The CP-PSW core and corridors scenario was the only plan covering the 10% weighed and the 20% flat target for all 17 biodiversity surrogates. Nevertheless that scenario still fell short of the 50% target for 5 surrogates (Table 1).

Relative Efficiency

We conducted a sensitivity analysis in each of 4 scenarios in order to compare the results of our relative efficiency measure (Eq. 4) with efficiency calculated by traditional methods (Eq. 3). Results (Figure 3) showed that efficiency for the existing reserves (scenario 1) was 0.89 (maximum efficiency would be 1.0), and that this scenario occupied approximately 5% of the floodplain. The CP-PSW core areas (scenario 2) scored 0.84 and covered 11%, the biosphere reserve nuclei and buffers (scenario 3) scored 0.67 and covered 26%, and the CP-PSW core and corridors (scenario 4) only scored 0.49 and covered 35% of the area evaluated.

To compare these metrics, we assigned cost weights to planning units that were not strict governmental reserves. Such weights represented the proportion of a rebate in the cost of private protection when compared to public protection. We found that relative efficiency scores balance the results of traditional efficiency when off-reserve mechanisms are under scrutiny. To illustrate that we set the cost associated with the inclusion of an off-reserve conservation planning unit to 50% (0.5 weighting factor) of the cost of public reserve planning units. The results show that the relative efficiency of the core CP-PSW scenario increased from an efficiency score of 0.84 to a relative efficiency score of 0.86, the biosphere reserve increased from 0.67 to 0.76, and the CP-PSW core and corridors scenario jumped from 0.49 to a relative efficiency score of 0.61 (Figure 4). However, sensitivity analysis of relative efficiency to weighting factors provided evidence that gains in relative efficiency can be limited. For the purpose of inter-

Table 2. Target achievement by each conservation scenario, with (1) indicating that the target was achieved and (0) indicating that the target was missed by the scenario. The last 3 columns represent the overall proportion of achievement for each of the 3 target levels evaluated.

Biodiversity surrogates features[1]	Scenarios	Combined public & private reserves			CP-PSW[2] Core areas			Biosphere Reserve nuclei & buffer zones			CP-PSW Core & corridors			Targets Achieved (Prop.)		
	Targets (%)	10[3]	20	50	10	20	50	10	20	50	10	20	50	10	20	50
	Regional name	Scenario (1)			Scenario (2)			Scenario (3)			Scenario (4)					
1 – Swampland	Brejo	1	1	1	1	1	1	1	1	1	1	1	1	1.00	1.00	1.00
2 - *Orbignya* palm woodland	Babaçual	0	0	0	0	0	0	1	1	1	1	1	1	0.50	1.00	1.00
3 - Floating mats	Baceiro	1	1	1	1	1	1	1	1	1	1	1	1	1.00	1.00	1.00
4 - *Mauritia* palm woodland	Buritizal	0	0	0	0	0	0	0	1*	0	1	1	0	0.25	0.50	0.00
5 – *Vochysia*.forest	Cambarazal	1	1	0	1	1	1	1	1	1	1	1	1	1.00	1.00	0.75
6 - Flooded grasslands	Campo inundado	1	1	1	1	1	1	1	1	1	1	1	1	1.00	1.00	1.00
7 - Dry grasslands	Campo seco	1	1	0	1	1	1	1	1	0	1	1	1	1.00	1.00	0.50
8 - *Byrsonima* scrublands	Canjiqueiral	0	0	0	0	0	0	0	1	0	1	1	0	1.00	1.00	1.00
9 - *Copernicia* palm woodland	Carandazal	0	0	0	0	0	0	1	1	1	1	1	1	0.50	0.50	0.50
10 - Savannah woodland	Cerradão	0	0	0	0	0	0	0	0	0	1	1	0	0.25	0.25	0.00
11 - Savannah scrublands	Cerrado	0	0	0	0	0	0	0	0	0	1	1	0	0.25	0.25	0.00
12 - Wet Chaco	Chaco	0	0	0	0	0	0	0	1*	0	1	1	1	1.00	0.25	0.50
13 - Semideciduous forest	Mata semidescídua	0	0	0	0	0	0	1	1	1	1	1	1	0.50	0.50	0.50
14 - Gallery forest	Mata de galeria	0	0	0	0	1*	0	0	1	0	1	1	1	0.25	0.75	0.25
15 - *Tabebuia* woodlands	Paratudal	0	0	0	0	0	0	0	1*	0	1	1	1	0.25	0.50	0.25
16 – Cyperus & Thalia swamp	Pirizal & Caetezal	0	0	0	0	1*	0	1	1	0	1	1	1	0.50	0.75	0.25
17 – Aquatic habitats	Aquático	0	0	0	0	0	0	1	1	0	1	1	0	0.50	0.50	0.00
Proportion of surrogates by scenario/target		0.29	0.29	0.18	0.29	0.41	0.29	0.59	0.88	0.41	1.00	1.00	0.71			

[1] From Da Silva et al. (2000).
[2] CP-PSW – The scenario name is based on the recommendations of the Cerrado Pantanal Priority Setting Workshop. In scenario (2) only core areas are considered, while in scenario (4) core areas and corridors are considered.
[3] All the 10% targets were weighted by the rarity and vulnerability of the surrogate
*Represents situations where the 10% weighted target is not achieved but the 20% target is.

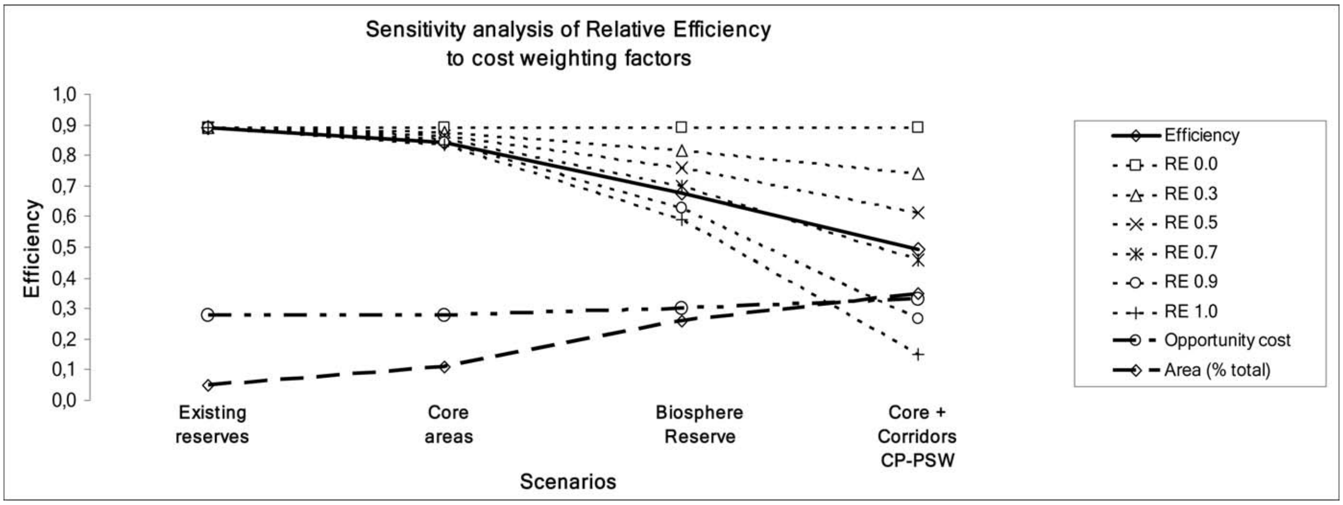

Figure 3. Sensitivity of the relative efficiency (Eq. 3) in each of the scenarios to the variation of weighting factors in the Pantanal. The lower lines represent difference in proportion of opportunity cost and variation in area among scenarios.

scenario comparison we demonstrate that increases in area through off-reserve mechanisms and reductions in cost via weighting rebates (scenarios 3 and 4) have a threshold, beyond which the gains in relative efficiency were not offset by weighting (Figure 3). When weights were higher than 0.7 (70% of the costs of public reserves), the relative efficiencies for the biosphere reserve and the CP-PSW core and corridors scenarios were worse than with the traditional efficiency method.

Opportunity Cost

For scenario (1), 132 planning units were recruited by existing reserves. Adding the proportion of irreplaceable planning units left outside that scenario (0.23) with the proportion of replaceable planning units included in that scenario (0.05), the opportunity cost of existing reserves was 0.28. For scenario (2), 194 planning unit were recruited, the proportion of irreplaceable planning units left outside of the core areas (0.21), summed to the proportion of replaceable planning units included in that scenario (0.07), also produced an opportunity cost of 0.28. For the Biosphere Reserve nuclei and buffer zones, 396 planning units were recruited and the proportion of planning units classified as irreplaceable (0.16) left outside of these 2 zones summed to the planning units classified as replaceable (0.14) generated an opportunity cost of 0.30. The last scenario evaluated included the core and corridors of the CP-PSW with 627 planning units. This scenario (4) recruited half of all planning units of the Pantanal, however it still left irreplaceable planning units outside of its limits (0.11) and the proportion of replaceable planning units included (0.22) generated an opportunity cost of 0.33. We found that adhesion to the entire recommendations of the CP-PSW fully represented all surrogate features for the 10% weighted, and the 20% targets, however it failed to represent 5 of the 17 biodiversity surrogates at the 50% target. It also proved to be the least efficient of all scenarios, whether we used efficiency in its traditional form (Pressey and Nicholls 1989) or the relative efficiency.

DISCUSSION

Our evaluation of existing and proposed conservation plans for the Pantanal wetland in Brazil argues for a comprehensive and systematic review of those plans to protect this wetland of global importance, within the framework of systematic conservation planning (Margules and Pressey 2000). We found that among the existing reserves systems, neither the combined public and private protected areas nor the Pantanal Biosphere Reserve (nuclei and buffers) fully protected the targeted samples of each of the 17 vegetation classes (i.e., biodiversity surrogates). In addition, none of the scenarios analyzed, including core and corridors of the recommendations of Cerrado-Pantanal Priority Setting Workshop, were simultaneously efficient and representative in conserving regional biodiversity. With few exceptions, the majority of the lands protected by the conservation scenarios were biased towards highly floodable habitats, reinforcing the worthless land hypothesis rather than providing complementarity and comprehensiveness (Pressey 1994).

We expanded the use of the concept of opportunity cost, summarized by proportion of irreplaceable

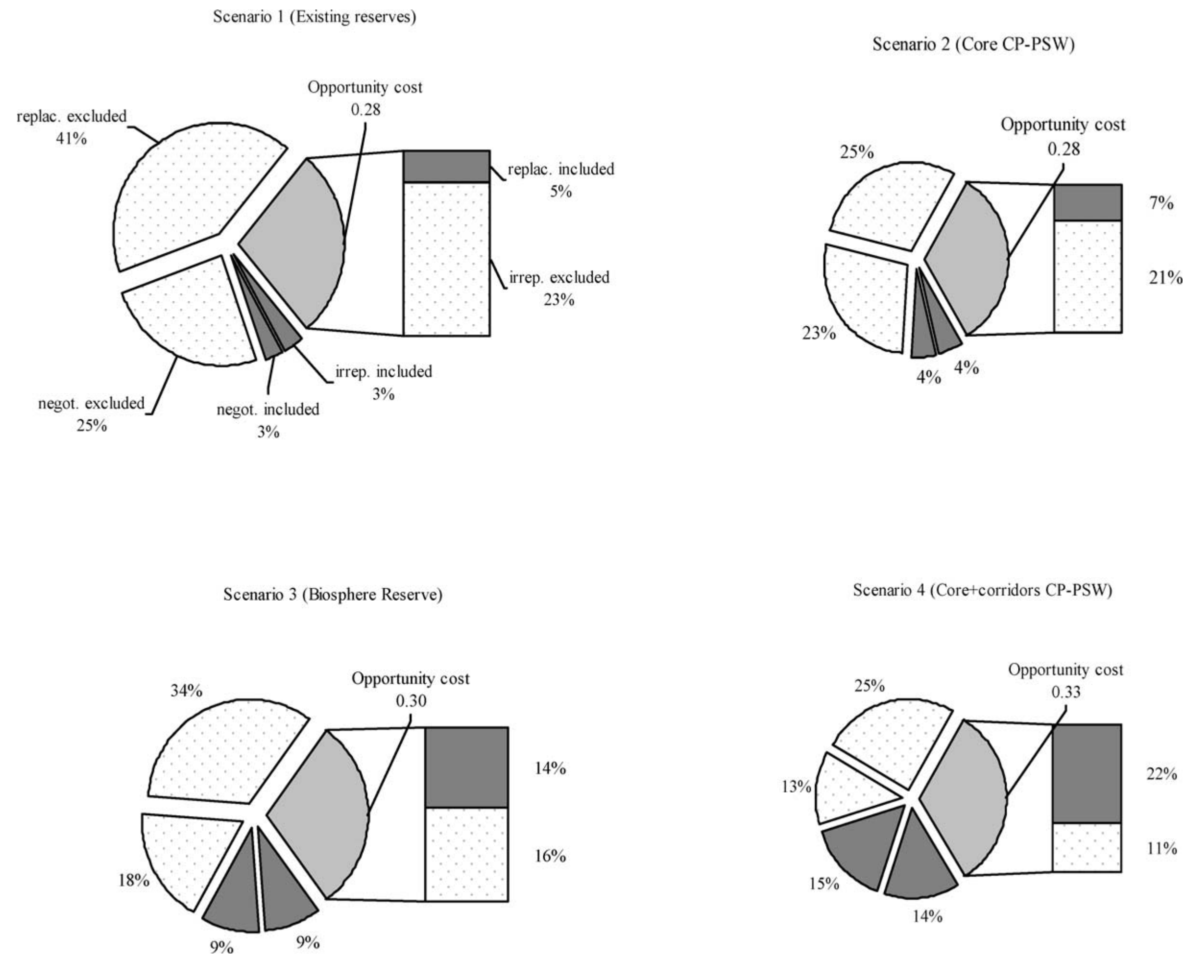

Figure 4. Missed conservation opportunities in the Pantanal for each scenario (i.e., existing reserves, core areas from Cerrado Pantanal priority setting workshop (CP-PSW), Biosphere reserve (Nuclei and buffer zones) and core and corridors from (CP-PSW) represented by the sum of the proportions of irreplaceable (irrep.) planning units excluded and replaceable (replac.) planning units included in each scenario (Figure 5). Proportions were generated in 10,000 runs of the software MARXAN and consider the selection frequency of each planning unit to define irreplaceability.

planning units excluded and replaceable planning units included in a conservation plan, to compare alternative conservation plans. The opportunity cost concept, previously used to evaluate the performance of individual planning units within a reserve system (Pressey and Nicholls 1989, Stewart et al. 2003), also proved useful for scenario comparisons. We found that around a third of planning units selected in each of the 4 scenarios evaluated (using the 20% target) were selected with frequencies lower than at random, and therefore represent high opportunity cost. The results indicated that the contributions of replaceable planning units (i.e., selected with frequency below average) and irreplaceable planning units (i.e., selected with frequency higher than the average plus one standard deviation) varied considerably among scenarios. The utility of the negotiable planning units (i.e., selected with frequency between the average and one standard deviation) towards scenario implementation can be substantial, particularly when they are used to help design buffer zones and corridors (Figure 5).

Providing full representation of specific targets based on the opportunity cost method may seem a trivial procedure, which could be solved simply by exchanging replaceable planning units included in a scenario with irreplaceable planning units that were not included. Yet there is no direct way to derive a tradability value between individual planning units. Trading planning units can alter efficiency in dramatic ways depending on the ownership composition chosen (Polasky 2005, Rothley 2006). Furthermore it is an almost impossible task when scenarios have already been implemented.

To suggest improvements to the 4 scenarios, we choose to illustrate arguments using the 10% target weighted by rarity and vulnerability of biodiversity

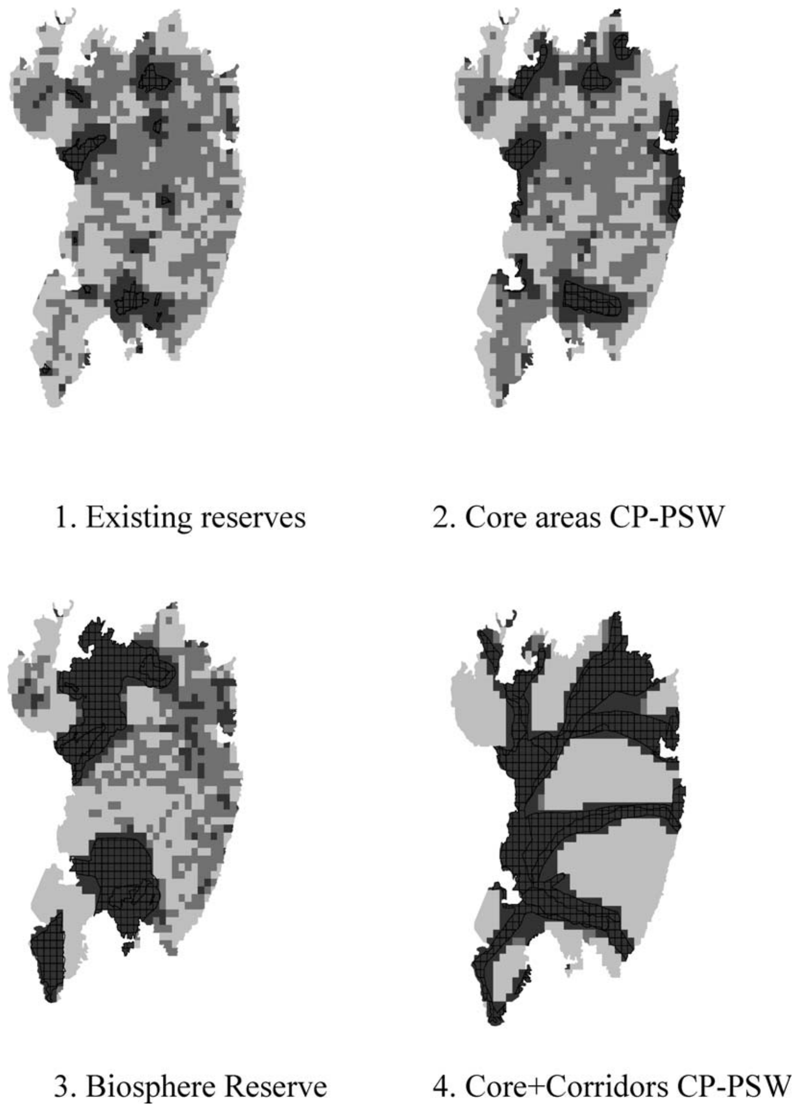

Figure 5. Suggested improvement to the four scenarios evaluated, based on irreplaceability classification criteria. Locked-in planning units in each scenario are represented by black grid in maps (1) to (4), where planning units are classified as replaceable (light grey), negotiable (medium grey), and irreplaceable (dark grey). For all scenarios the representation target chosen was 10% for all 17 biodiversity surrogate classes weighted for rarity and vulnerability (Pressey and Taffs 2001a). MARXAN was set to allow for moderate clumping.

surrogates because it provides proportionally more habitat to specialist species while protecting the vulnerable connections between the plateau and the floodplain (Pressey and Taffs 2001a). To fully represent all 17 biodiversity surrogates we suggested the inclusion of irreplaceable planning units derived from our analysis. For each scenario all planning units included were locked into the solution (Figure 5). Suggesting the addition of planning units to each scenario imposed new spatial configurations and consequently variable impacts over the system's efficiency. However we suggest an improvement in the way the efficiency of a conservation plan can be measured to deal with multi-ownership plans. Relative efficiency, which assumes differential acquisition, implementation, and management costs between public reserves and other forms off-reserve conservation, can be implemented as a weight by which planning units in a particular ownership regime are multiplied. The aforementioned cost differentiation in measuring efficiency also provides an argument to support governmental investments in stewardship mechanisms because overall area is increased without proportional increases in costs. This new metric can be particularly useful when acquisition costs are high (Knight et al. 2006).

Engaging societies in conservation is essential for the future of Pantanal biodiversity because 95% of the land is privately owned, deforestation has already eroded more than 17% of wooded savannas habitats in the floodplain, and water diversion and erosion typically accompany development in private lands. Selective habitat destruction on the edges of the Pantanal is reducing connectivity between the plateau and the floodplain with impacts on critical ecosystem processes and services provided by this link (Seidl 2001, Junk et al. 2006b). Our results showed that the Biosphere Reserve plan presented the best compromise between relative efficiency and opportunity cost among all 4 scenarios. It included only 14% of the replaceable planning units, and omitted only 16% of irreplaceable planning units. More importantly, the Biosphere Reserve plan expanded the boundary of traditional reserves via zonation mechanisms. Nevertheless the assumptions of environmental, socio-cultural, and economic sustainability require further testing.

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