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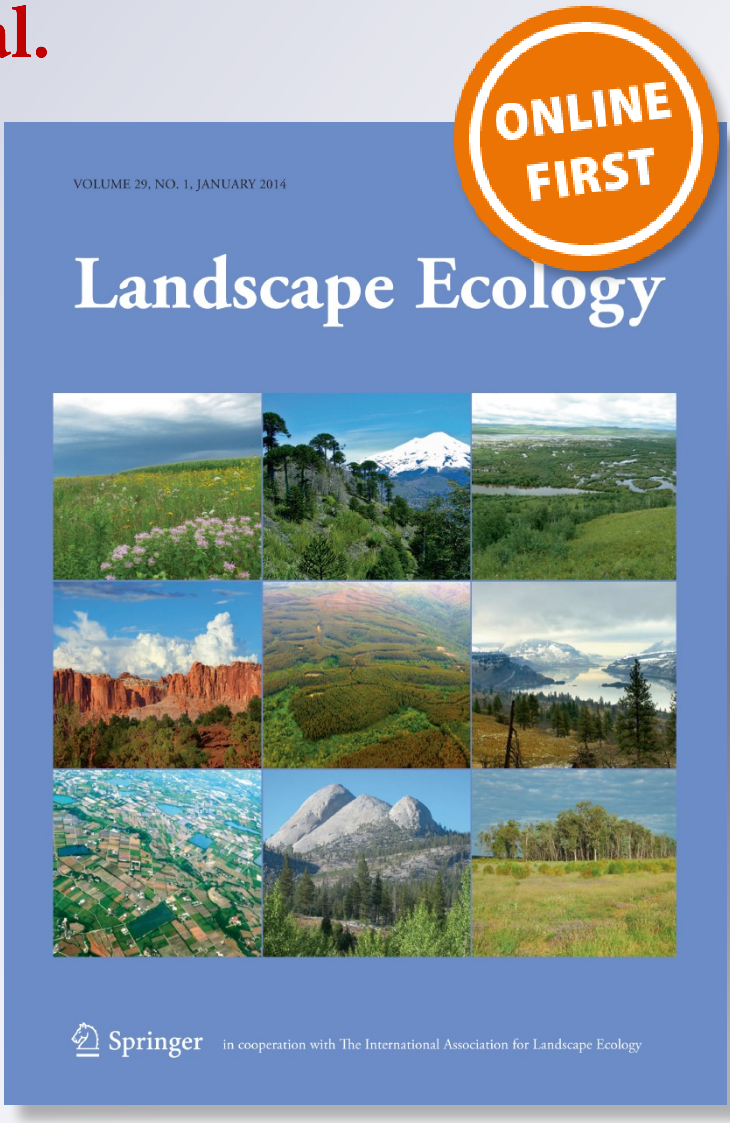

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RESEARCH ARTICLE

How good are tropical forest patches for ecosystem services provisioning?

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Abstract Native forests play an important role regarding ecosystem services related to biodiversity, water, and nutrient cycling, and the intensity of those services should be related to the amount, configuration and quality of the forest. However, in highly dynamic landscapes, such as some tropical regions, ecosystem services are potentially affected not only by the present landscape structure, but also by the historical land use. Here we propose a simple methodological framework to evaluate the contribution of past landscape dynamics and present landscape structure in the provision of ecosystem services. We applied this framework to a traditional agricultural landscape from the Brazilian Atlantic Forest hotspot, where natural forests cover has increased from 8 to 16 % in the last 60 years (1962–2008), and where old forests are being reduced while young forests are being regenerated. Forests of different ages, in association with current landscape structure, reveal a mosaic of forest patches under different conditions, implying different abilities to deliver ecosystem services. With the replacement of old-growth forests by young-regenerating forests and a high level of forest fragmentation, less than 1/4 of the current forest cover is able to fully satisfy the ecosystem service demands. To avoid such tendency, government policies should not only focus on increasing forest cover, but also in conserving old-growth forest fragments or increasing forest quality. The proposed methodology allows integrating historical land use and current landscape structure to evaluate ecosystem services provision and can be useful to establish programs of payment for ecosystem services.

Keywords Agricultural landscape · Biodiversity · Landscape dynamics · Atlantic Forest · Historical land use · Sugarcane · Pasture · Corumbataí

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Introduction

The continuous forest ecosystems that originally covered most of the tropical regions worldwide have been historically transformed by agricultural and urban

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expansion (Lambin et al. 2001) into aging human-modified landscapes (hereafter HML) (Gardner et al. 2010; Tabarelli et al. 2010). In such regions, the landscape matrix is dominated by anthropogenic activities, and the natural ecosystems currently occur as small and isolated forest fragments (Tabarelli et al. 2008; Ribeiro et al. 2009). In addition to strong edge effects caused by such high levels of fragmentation, human-mediated disturbances, such as fires, logging, hunting, and biological invasions, may establish a degradation gradient among forest fragments. Moreover, different forest re-growth processes, mediated by social, economic, and ecological outcomes (Rudel 2012), are also affecting the quality and age of forest patches (Lira et al. 2012a) and thus affecting all ecological processes associated with those patches (Lira et al. 2012b). Together, forest loss, fragmentation, degradation, and regeneration processes have transformed HML into a heterogeneous mosaic of forest remnants in different successional stages. Secondary forests generated by the above-mentioned processes account for more than 70 % of current global tropical forest cover (FAO 2010). By the end of the twentieth century, approximately 12 % of tropical rainforest cover was experiencing different stages of re-growth, following logging, agricultural abandonment, or conversion (Wright 2010).

The importance of secondary forests and the consequences of heterogeneous HML for biodiversity conservation and ecosystem processes and services supply is still an open question. Depending on the quality of these forests and on the level of biodiversity that they harbor, secondary forests can play an important role for biological conservation and ecosystem services (hereafter ES) provisioning (Gardner et al. 2009; Chazdon et al. 2009; Melo et al. 2013). Particularly, these remnants can play an important role in sequestering carbon dioxide (Pan et al. 2011) since secondary patches show a fast accumulation of biomass in the first years of the regeneration process (Brown and Lugo 1990), and thus partially offset carbon losses from tropical deforestation (Zarin 2012). A meta-analysis carried out with 600 secondary tropical forest sites across the world showed that secondary forests may take about 80 years to fully recover above-ground biomass levels of primary forests, while tree species richness took approximately 50 years (Martin et al. 2013). In this context, the quantification of forest quality is crucial to determine their potential for biodiversity conservation and ES provisioning.

Although recent studies have shown an increasing ability to detect modifications in forest structure and quality through imagery techniques (see examples in Asner et al. 2005, 2009; Wright 2010), current methodologies are still unable to concomitantly assess the historical dynamics of deforestation, degradation, and regeneration that have resulted in the mosaic of forests found in HML. For instance, the detection of natural regeneration in remote sensing assessments is often compromised by the small size and diffused spatial distribution typical of young regenerating forests (Chazdon 2012). As recognized by Putz and Redford (2010), many types of forests may result from human interventions affecting the structural and compositional features of natural ecosystems, so that the investigation of the processes affecting the formation, development, and successional maturation of tropical forests has a remarkable relevance for forest conservation and management (Brancalion et al. 2012b).

Therefore, we argue that the provision of ecosystem services by tropical forests depends on the ecological integrity and its landscape sustainability (Wu 2013), and therefore assessments of ES should include more nuanced measures that quantify the gradient of quality and spatial heterogeneity of forested ecosystems. Because of the increasing implementation of policies devoted to payments for ecosystem services (Balvanera et al. 2012; Melo et al. 2013), the assessment of forest quality and regeneration dynamics assumes an utmost importance for decision-making.

In this study, we proposed a methodological framework to assess forest ecosystem services by integrating past landscape dynamics and present landscape structure. Considering the historical processes of deforestation and re-growth of Brazilian Atlantic Forest patches in HML, we aimed to investigate the range of potential performances played by these remnants in terms of ES provision, and discuss the implications of these differential performances for decision-making in biological conservation and ecological restoration efforts.

Methods

Methodological framework for potential ecosystem services assessment

Our assessment was focused on regulation of ecosystem services (erosion control, carbon sequestration,

Table 1 Forest patch and landscape structure indices used to estimate the level of ecosystem services supply

Indices	Weight	Criteria for the classification of ecosystem services potential
Forest age (years)		Old-growth forests have better vegetation structure (Martin et al. 2013), harbor higher biodiversity levels (Liebsch et al. 2008; Santos et al. 2008; Martin et al. 2013), and have higher soil cover (Guariguata and Ostertag 2001), nutrient and water cycling (Davidson et al. 2007), water regulation (Zhang et al. 2001) and carbon storage (Martin et al. 2013) (Brown and Lugo 1990; Letcher and Chazdon 2009; Chazdon 2012).
0–7.9	1	
8–15.9	2	
16–24.9	3	
>25	4	
Local forest neighborhood dominance (%)		Interior forest is less exposed to disturbances like fire (Armenteras et al. 2013), strong winds, drought and biological invasions (Laurance et al. 2002), and can thus provide better conditions for biodiversity conservation (Santos et al. 2008), carbon sequestration (Laurance et al. 2002), carbon stock (Williams et al. 2008), soil protection (Neary et al. 2009) and water infiltration (Tabarelli et al. 2008; Pütz et al. 2011).
0	0	
0.01–0.33	1	
0.34–0.66	2	
0.67–1	3	
Forest proximity (no units)		More connected forest patches allow better biological (Schmiegelow et al. 1997) and physical flows in the landscape (Saunders et al. 1991), thereby supporting biodiversity conservation (Becker et al. 2007; Pardini et al. 2005), crop pollination (Ferreira et al. 2013) and essential fluxes of energy and abiotic materials for delivering regulation of ecosystem services (Tscharntke et al. 2005; Melo et al. 2013).
0–2.7	0	
2.7–5.5	1	
5.5–11.0	2	
>11	3	
Forest contiguity (%)		Larger forest patches harbor higher numbers of species (Turner 1996) and provide better ecological conditions for their long-term perpetuation (Becker et al. 2007; Pardini et al. 2005), as well as increase soil protection (Neary et al. 2009) and water flow regulation (Zhang et al. 2001) (Laurance et al. 2011).
0–0.9	0	
1–1.9	1	
2–3.9	2	
>4	3	

pollination, water flow regulation, etc.) and biodiversity conservation (hereafter referred to as only "ecosystem services"). Assuming that forest ES performance is directly related to local (forest structure, composition, and functioning) (Santos et al. 2008; Martin et al. 2013) and landscape context (habitat amount, fragmentation, isolation, and edge effects intensity) (Banks-Leite et al. 2011), we propose to spatially assess potential ecosystem services offered by forest patches according to patch and landscape indicators (Egoh et al. 2008; Baral et al. 2013). The relationship between indicators and correspondent ES are detailed in Table 1. As there are several ES related to each indicator, instead of estimating the distribution of each ES in the landscapes, we decided to conduct all the analyses based on four indicators. With this procedure, we avoided double counting the same criteria used during estimations of each ES. All indicators should be evaluated in a grid of cells, which resolution should be defined according to the resolution of the land use and land cover map available. The indicators of potential forest ecosystem services were:

Mean Forest Age (FA, in years)—Forest age is used here as a surrogate of forest integrity (or quality), considering that old-growth forests have a better performance on ecosystem services provisioning than early regenerating forests. To evaluate FA, we propose an approach similar to Lira et al. (2012b), which pointed out the importance of considering historical degradation of forest remnants in order to understand current biological community and vegetation structure. FA is defined by temporal overlaying of land cover maps, using the difference between the most recent date and the first year of forest occurrence in the past. A zonal statistical analysis can be performed to calculate area-weighted mean values of FA for each forest cell.

Local Forest Neighborhood Dominance (FORNEIGH, in %)—Considering that interior forest can perform better ecosystem services than forest edge, we propose to assess local forest neighborhood dominance by examining the eight surrounding cells around each forest focal cell, in order to calculate the proportion covered by forest. A moving window analysis can be performed at this step.

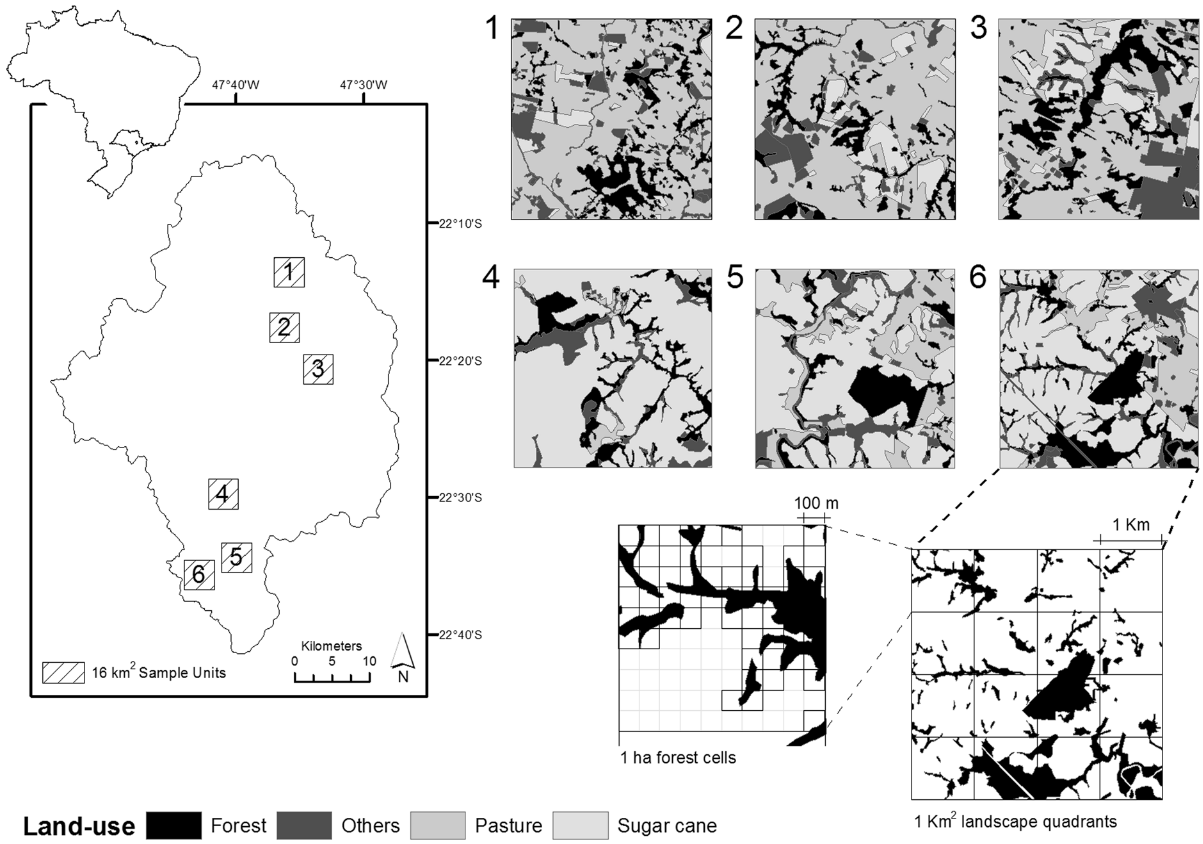

Fig. 1 Distribution of the 16 km^2 focal landscapes dominated by pasture (1, 2, 3) and sugarcane (4, 5, 6) at Corumbataí river basin (southeast Brazil) and corresponding land-use land-cover maps. To conduct the multi-scale analysis, the focal landscapes were divided in 1-km^2 quadrants (7), which were then divided in 1 ha cells to analyze the forest dynamics (8)

Forest Proximity (PROX, unit less)—Proximity was used as surrogate of local habitat connectivity, considering that more connected patches provided higher levels of some regulating ecosystem services (such as pollination and disease regulation) than more isolated ones (Ricketts et al. 2004; Melo et al. 2013). Based on the concept proposed by Gustafson and Parker (1992), we propose to calculate the mean proximity index (McGarical and Marks 1995) of forest present in a 2 km buffer around forest cells.

Forest Contiguity (FORCONT)—This metric brings to forest cells the relative size of their forest patch in relation to focal landscape. We considered that bigger forest patches were able to provide higher levels of ecosystem services provisioning (Laurance et al. 2011). In order to capture the forest contiguity of each forest cell, we propose to use the proportion of focal landscape occupied by forest patches where forest cells are inserted.

To simplify the analysis, we propose to classify each metric into four levels (from 1 to 4 for FA, and 0 to 3 for other variables) according to its level of contribution for ecosystem services provisioning (Table 1).

Studied landscapes

Our case study landscapes are situated in the Corumbataí river basin, a region of 1,700 km^2 (Fig. 1) with more than 200 years of land-use change, which was historically driven by deforestation resulting from logging and expansion of cropland and pastureland. Today, the main land uses in the Corumbataí river basin are sugarcane fields (26 %—mainly on lowlands) and extensive pasturelands of African fodder grasses (44 %—mostly on highlands and slopes), while the remaining native vegetation covers only 12 % of the landscape (Valente and Vettorazzi 2003).

Six samples of landscapes (hereafter focal landscapes) were selected following a diversity variability analysis, as proposed by Pasher et al. (2013). First, the study region was completely divided into five different scales of square grids, using 1, 2, 3, 4, and 5 km square grid cells, and resulting in landscape samples of 1, 4, 9, 16, and 25 km^2. Then, for each grid size, the Shannon landscape diversity index (McGarical and Marks 1995) of each cell was calculated based on a 30 m-resolution land-use map from 2002 (Valente and Vettorazzi 2003). Finally, the mean landscape diversity of each grid size was plotted against the cell size. The focal landscape sample size adopted was 16 km^2, since it represents the smallest sample size that shows no variation when compared to the landscape diversity index of higher sampled sizes, and thus, is able to represent the landscape diversity of the study area.

The final criteria to define the location of the focal landscapes was that sampled landscapes should have had, in 2008 (latest image available), at least 10 % of native forest cover and at least 70 % of matrix, sugarcane (hereafter sugarcane landscapes) or pasture (hereafter pasture landscapes). The study region was thus submitted to a moving window analysis, using the 16 km^2 landscape size established previously, in order to calculate, for each pixel, the land-use proportion of sugarcane, pasture and forest of a sample size window centered on it. Among the potential landscapes, three landscapes were chosen randomly (avoiding any overlap among landscapes) for each predominant agricultural land use (sugarcane and pasture).

Landscape dynamics analysis

For land-use mapping and FA estimation, panchromatic aerial photographs were used from 1962, 1978, and 1995 (1:25,000 scale), and a panchromatic image from High Resolution Panchromatic Camera (HRC) of CBERS (2.7 m of spatial resolution) was used for 2008. Photographs were digitized at 300 dpi resolution, for a final spatial resolution of 2.5 m. The CBERS image was georeferenced based on topographic maps at a 1:10,000 scale, and digital images from other years were spatially registered to the 2008 image, which was used as reference.

Land-use maps were obtained by photointerpretation, using a temporal sequence, from present to past, where incongruent information was corrected in pairwise comparison. The following land-use classes were considered: sugarcane, pasture, old-growth native forests, young-regenerating native forests, orange plantations, eucalyptus plantations, urban areas and others. Land-use transition rates were obtained for 1962–1978, 1978–1995 and 1995–2008. These transition rates were used to build one global transition matrix for each predominant agricultural land use (sugarcane and pasture), using average values for the whole period observed (Ferraz et al. 2005).

In order to better understand the historical processes driving the dynamics of degradation and regeneration of forest patches, focal landscapes were subdivided into 1 km^2 quadrants (hereafter quadrant), resulting in 16 quadrants per focal landscape and 96 quadrants in total (Fig. 1). For each quadrant, the following landscape dynamic indicators were calculated: Mean Annual Forest Change rate (q) and Forest Change Curvature Profile (FCCP). Those indicators were calculated using Land-Use Change Analysis Tools, an ArcGIS extension (Ferraz et al. 2011, 2012):

Mean Annual Forest Change Rate (q) measures annual forest change rate on quadrants using the annual rate of forest change equation (FAO 1995). Positive values represent forest increment and negative values represent forest loss over years. The unit is %/year (forest change proportion per year), calculated as

$$q = \left(\frac{\mathrm{FS_n}}{\mathrm{FS_1}}\right)^{\frac{1}{Y_n - Y_1}} - 1 \qquad (1)$$

where $\mathrm{FS_n}$ = amount of forest in the year n (ha); $\mathrm{FS_1}$ = amount of forest in the first year (ha); $\mathrm{Y_n}$ = final year; $\mathrm{Y_1}$ = initial year.

FCCP represents the maximum deviation of the forest change curve in relation to the linear model linking initial (Pf0) and final (Pfn) forest amount over time. Positive deviations represent forest change concentrated in early years, while negative deviations represent forest change concentrated in recent years. Small deviations represent scattered changes in patterns over time. FCCP is unit-less and its signal is determined by the main position of the forest change curve in relation to the linear model (see Ferraz et al. 2009):

$$\mathrm{FCCP} = \pm\mathrm{MAX}\left[\left|\mathrm{aFP_i} + \mathrm{b.Y_i} + \mathrm{c}\right| / \sqrt{\mathrm{a^2} + \mathrm{b^2}}\right]_1^{\mathrm{n}} \qquad (2)$$

where $\mathrm{FP_i}$ = Forest proportion in the year i (%); $\mathrm{Y_i}$ = Year i, varying between 1 and n;

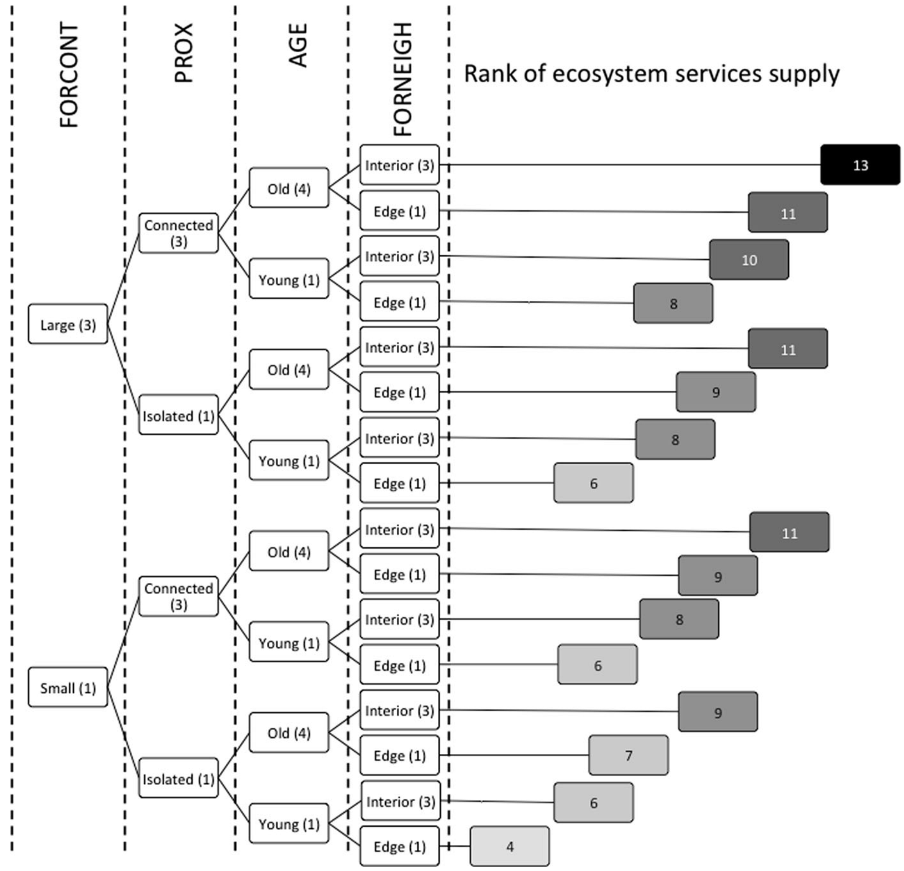

Fig. 2 Logical diagram for integration of ecological conditions of forest cells and their resultant score of ecosystem services potentialities. Only lower and upper limits weights of each indicator are represented

n = number of studied years; a, b, c = coefficients of equation for the general forest change linear model.

Forest ecosystem services provisioning analysis

The four potential ecosystem services indicators were calculated on 1 ha cells of forest fragments present in the 2008 map (hereafter forest cells) (Fig. 1). A 1 ha grid map was overlapped on the 2008 forest map (5 m resolution), creating forest cells with a maximum size of 1 ha (forest cells with less than 100 m^2 were discarded). Forest proximity indicator was calculated considering a 2 km search radius (half of focal landscape edge size).

On forest cells, indicator scores were combined by addition, resulting in values ranging from four (low services will be provided) to 13 (full services will be potentially provided; Fig. 2). In order to summarize results, following quartile distribution of 25, 50 and 75 %, forest cells were organized into four classes: very low, low, medium and high potential of services providing.

Results

During the period 1962 to 2008, in pasture landscapes, pasturelands were reduced by 20 % (from 70 to 50 %) and were replaced by sugarcane fields (from 10 to 15 %) and natural vegetation (10 % of native forests regenerated over previous pasturelands). The expansion of native forests in pasture landscapes started 30 years ago and was consolidated in the 1990's. In sugarcane landscapes, the area covered by sugarcane fields was kept stable at around 60 %, excluding a small retraction in the 1970's. Pastureland cover was reduced from 20 to 10 %, while native vegetation cover increased from 8 to 15 % (Fig. 3). The expansion of native forests in the 1970's also occurred at landscapes dominated by sugarcane fields, but a recent expansion phase was also observed in the 2000's. This recent expansion of new forests occurred mostly in previously existing old growth forest patches; thereby, current forest patches may be highly heterogeneous.

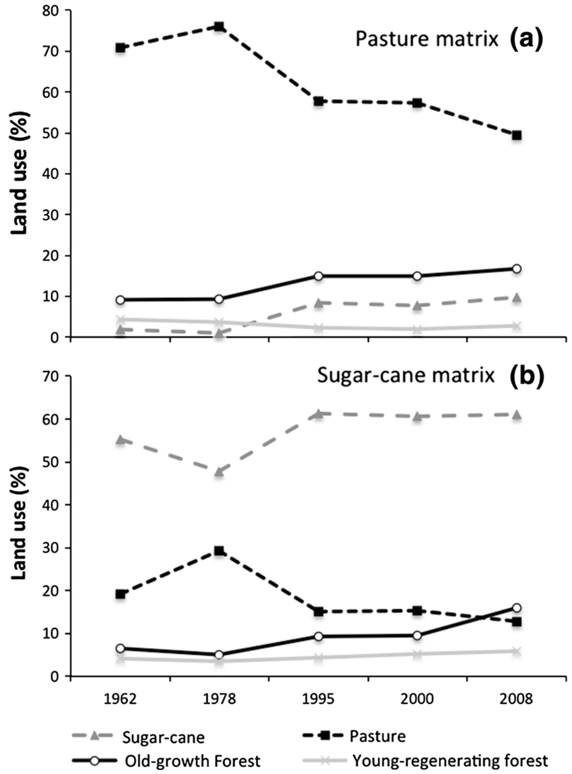

Fig. 3 Temporal variation of land use in pasture (**a**) and sugarcane (**b**) focal landscapes

In pasture landscapes, forest increase was basically related with pasture abandonment (7.9 %, Fig. 4). Pasturelands were very dynamic, having been alternated with other uses, especially sugarcane, according to the economic situation at the time. Old-growth forest remnants were predominantly kept in the landscape, while land abandonment allowed forest re-growth. Land-use conversions to old-growth forest and young-regenerating forest represented 10.3 % of all land-use transitions, while conversions from old-growth forest and young-regenerating forest to other uses amounted to only 5.4 % of the total transitions. Although recent forest re-growth has overcome forest loss, young secondary forests are replacing old-growth forests. Therefore, forest cover has increased, but the quality of the remaining forests has decreased.

In sugarcane landscapes, forest expansion was mainly caused by abandonment of other land uses (8.5 %), despite the low conversion of sugarcane fields to native forests (2.3 %). Total conversion to old-growth forest and young-regenerating forest totaled 16.4 % of transitions, while forest loss transitions represented 3.6 %.

In summary, pasture landscapes were more dynamic, with processes of forest loss and re-growth occurring at the same time. Sugarcane landscapes were more stable than pasture landscapes, with lower rates of forest loss and a recent process of forest regeneration over other land uses.

Forest dynamics at a 1-km^2 quadrants scale showed that both forest loss and regeneration were occurring, although forest re-growth predominated over forest loss (Fig. 5). Few quadrants presented q values near to zero (no size variation), showing that few forest patches were kept without forest suppression or regeneration; most have lost small portions of old-growth forest and received increments of new forests recently.

The assessment of potential ecosystem services provisioning of forest cells using patch and landscape metrics showed that values usually ranked from 3 to 13, with an average of 7 (Fig. 6). Considering 3,806 forest cells, only 20 reached the maximum value of 13 and 17 % showed a high potential (classes 10–13), while 16 % were classified as having a very low potential (classes 0–4), 44 % had a low potential (classes 5–7) and 23 % were classified as medium potential (classes 8–9). The highest values of ecosystem services potential were usually observed in large and old-growth forest patches or large riparian corridors. In sugarcane landscapes (units 4, 5 and 6), forest was more concentrated around streams and large fragments, although these landscapes have provided medium potential for provisioning ecosystem services according to our model.

Discussion

Human modified landscapes are usually submitted to intensive dynamics of land use and land cover changes (Tabarelli et al. 2012b). In our study region in southeastern Brazil, the main land-use transitions observed in landscapes were the loss of old-growth forests to agriculture or pasture, and the regeneration of young secondary forests in abandoned lands, as well as conversion of pasturelands to sugarcane fields. As a result of such human-mediated historical processes, current forest cover is very heterogeneous and, according to our criteria, the forest patches included in such landscapes could have different potentials to provide ecosystem services to society. The

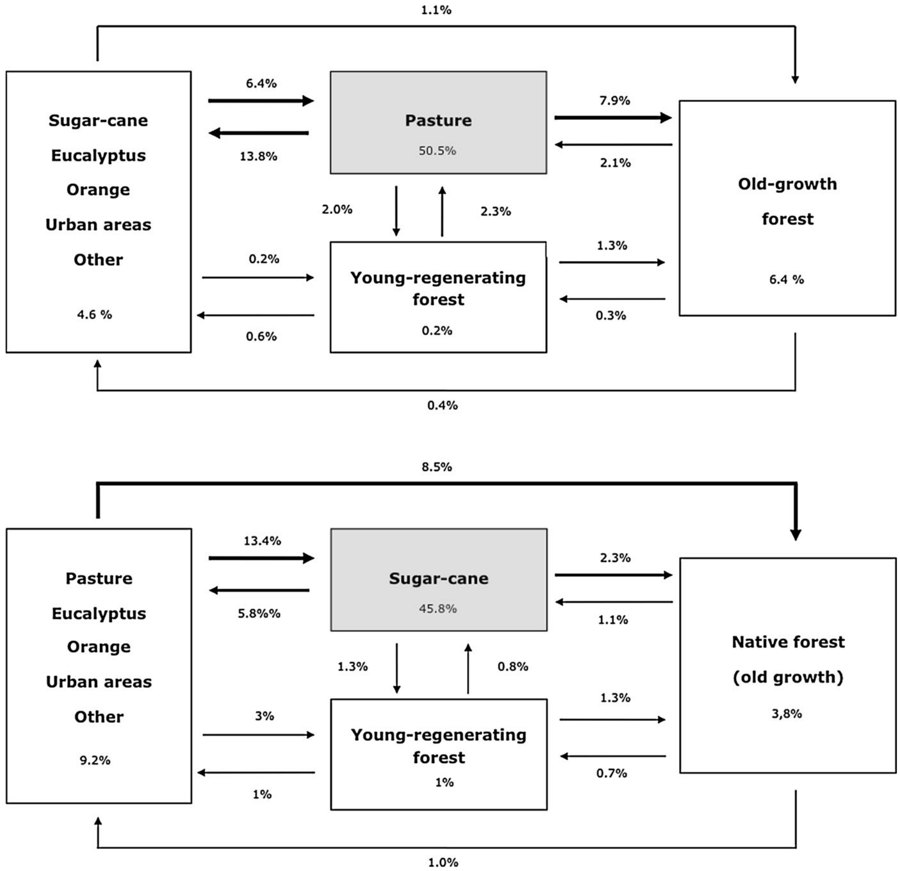

Fig. 4 Land use transitions rates from 1962 to 2008 on pasture focal landscapes (*above*) and sugarcane (*below*) at Corumbataí river basin. *Bold arrows* represent main land-use transitions

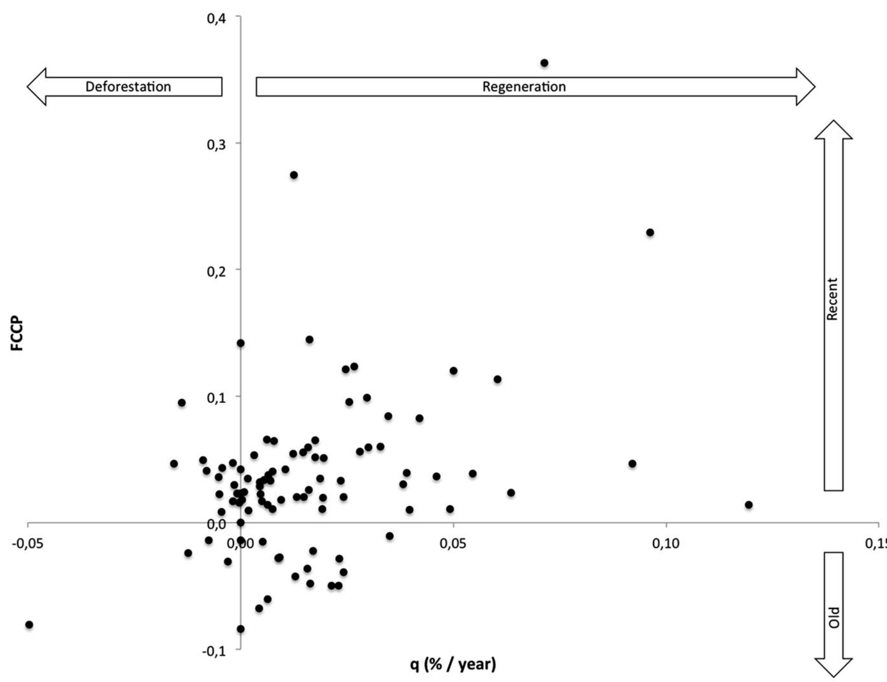

Fig. 5 Forest Change Curve Profile (FCCP) and annual forest change rate (q), calculated for 1 km^2 landscape quadrants between 1962 and 2008

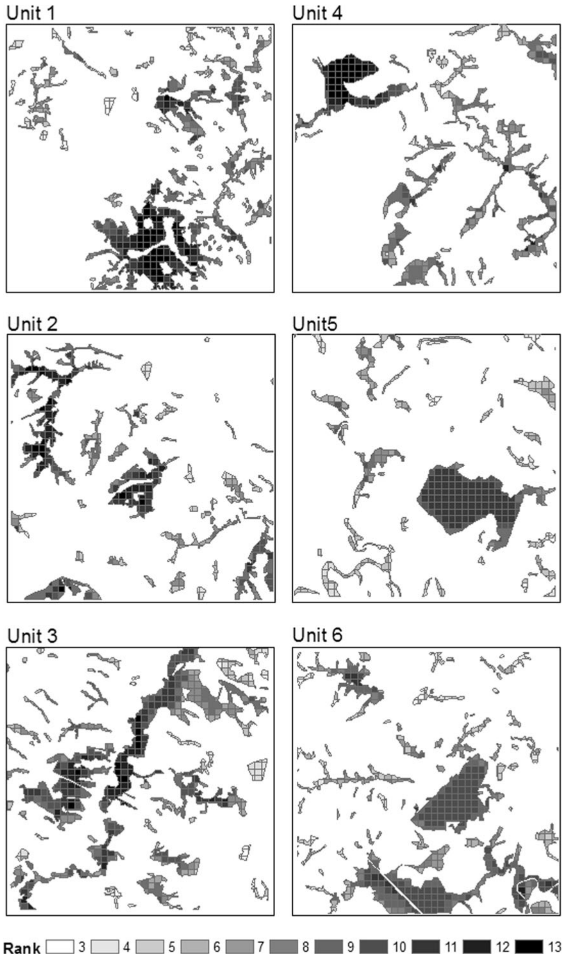

Fig. 6 Potential for supplying ecosystem services for all forest cells of pasture (1, 2 and 3) and sugarcane (4, 5 and 6) landscapes

identification of the many levels of ecological integrity of these forest patches may provide a relevant contribution for supporting the transformation of HML into landscapes more friendly to biodiversity (Melo et al. 2013). The differences found between pasture and sugarcane landscapes may also provide valuable lessons for managing HML in order to favor biodiversity conservation and optimize the provision of ecosystem services.

Pasture landscapes were more dynamic than those where sugarcane fields predominate. This result can be explained by the rationale of land use by extensive cattle ranching, in which marginal lands are occupied and low investments in the production system are made. Indeed, the stocking rate of pasturelands in the Atlantic Forest region is only 1.18 head per ha (36 million head of cattle distributed in 30.5 million ha of planted pastureland—(IBGE 2003; PROBIO 2009). Given that this model of extensive cattle ranching does not require flat terrain or suitable soil conditions to be implemented (which in part explains the low productivity), both the conversion of forests to pasturelands and the regeneration of forests over abandoned pasturelands are more frequent. It is expected that pastures located in suitable areas for intensive soil cultivation nearly be be all converted to agriculture (Rudorff et al. 2010; Macedo et al. 2012), as indicated by the observed expansion of sugarcane fields in pasture landscapes.

On the other hand, sugarcane cultivation is highly dependent on mechanization, which limits its expansion in steeper terrain (Rudorff et al. 2010). Given that most old-growth forest remnants in this region are located in marginal areas for agriculture, forests were less affected in sugarcane landscapes than in pasturelands. In both landscapes, the slight expansion of forests between the 1970's and 1990's was probably a result of the abandonment of areas with very low production potential, such as on steep slopes and sandy, rocky soils. The recent increase of young secondary forests in sugarcane landscapes probably resulted from the environmental planning efforts for compliance with the Brazilian Forest Code (see Rodrigues et al. 2011 for details). This trend was evidenced by the large proportion of young native forests along streams and in large blocks of forests, which is a result of the compliance with this legal instrument (Rodrigues et al. 2011).

The recent increase of forest cover in both landscapes is a clear evidence of the third phase of forest transition, according to the environmental Kuznets curves, after previous periods of high forest loss followed by the intense reduction of deforestation rates and consequent forest cover stabilization (Mather 1992). Although the expansion of forest transition has brought hopes for the recovery of tropical forests across the world (Rudel 2012), including in Brazil (Perz and Skole 2003; Baptista and Rudel 2006), our results showed that old-growth forests are being replaced by young-regenerating forests, confirming trends observed in other Atlantic Forest regions (Teixeira et al. 2009; Metzger et al. 2009; Lira et al. 2012a). The forest cover increase was also observed in other regions of the São Paulo State (Farinaci and Batistella 2012) and probably the same replacement of old-growth forests by young secondary vegetation has been occurring in the whole State, and this could represent a decrease in ecosystem services delivered by forests on a per unit area basis, besides its increase in extension. These landscape dynamics should be carefully observed by public agencies since most government policies and Payment for ES programs are focused on increasing forest cover (Guedes and Seehusen 2011) instead of conserving old-growth forest fragments or increasing forest quality (Brancalion et al. 2012a). The continuous substitution of old-growth by young-regenerating forests will lead to fragments with less biomass (Groeneveld et al. 2009; Martin et al. 2013) and result in the loss of vegetation species richness and functional diversity (Guariguata and Ostertag 2001; Santos et al. 2008; Martin et al. 2013).

Given that law enforcement is relatively effective in this region, we believe that most old-growth forests may have been lost not by deforestation, but through intense degradation caused by fires, since sugarcane straw is burnt before harvesting in the dry season, which causes large-scale forest fires both in sugarcane landscapes and in neighboring pasture landscapes (Durigan et al. 2007; Martinelli and Filoso 2008). The presence of intense anthropogenic perturbation is recognized as important in driving old growth forests to initial successional stages (Santos et al. 2008; Tabarelli et al. 2008) and also in limiting the improvement of biomass, vegetation structure, and species richness in regenerating patches (Guariguata and Ostertag 2001). Old-growth forests are thus continuously being degraded and lost, and despite the net increase of forest cover, there is an important and continuous net loss of forest quality. These results highlight the importance of assessing landscape dynamics and their consequent effects on the profile of forest patches for predicting the benefits and setbacks resulting from forest transition. Moreover, the proposed methodology to assess forest ES, that can represent the "forest quality" within a human perspective, may be helpful for supporting the management of these landscapes to optimize the provision of ES, as well as to guide ecological restoration efforts.

Methodological limitations and implications

The methodology discussed here provides indirect estimation of ES, that is a first step toward assessing the provision of ES by recognizing the historical land use legacy and landscape context (Allen et al. 2002; Bain et al. 2012), which goes beyond inferring ES using only the current landscape cover (Burkhard et al. 2009). Previous authors have suggested that past history and landscape context were important (Ferraz et al. 2009; Lira et al. 2012b), but this is the first study that tries to integrate these factors in a unique methodological framework. Criteria considered for ranking (size, connectivity, age, and edge/interior) have been extensively used in biodiversity studies (Lindenmayer et al. 2000; Cushman et al. 2008); however, better indicators (metrics) could be applied in order to improve results for ecosystem services.

Another important point observed in the ES potential map (Fig. 6) is that due to high variability of ecological condition inside forest fragments, their ES assessment should be considered at the intra-patch scale since average values based on patch metrics could masquerade its real condition. Considering the availability of high-resolution images, landscape units of 1 ha, as used in this study, seem to be an adequate scale.

The proposed framework is simple and can be applied in a wide range of situations, even when little information is available. However, the criteria and classification methodologies proposed by this framework can be improved when more detailed knowledge and information is available. For example, forest cells can be classified considering a simple decision tree of binary conditions (large (3)/small (1), connected (3)/isolated (1), old (3)/new (1), interior (3)/edge (1); Fig. 2), but as knowledge on landscape indicators of ES evolve, not only an intermediary condition (2) could be considered but also the use of continuous variables and the identification of thresholds (de Groot et al. 2010) could be incorporated into the analysis. Moreover, instead of summing or integrating indicators, as performed in the present study, individual analysis of ecosystem services could be conducted, highlighting spatial synergies or trade-offs among different ecosystem services.

Independently of the methodological details, this framework allows the classification of forest cells into categories of services provisioning, which could find extensive application in Payment for Ecosystem Services programs.

Implications for ecosystem services assessment

The evaluation of landscape dynamics and a proposed methodological framework to integrate several patch and landscape parameters in the evaluation of ES provision can have diverse implications for management, conservation and restoration actions.

First, the assessment of the history of degradation/regeneration of each forest remnant may support biological conservation efforts to increase the persistence of biodiversity in such landscapes (Melo et al. 2013), which have been historically submitted to a process of biotic homogenization (Lobo et al. 2011) and proliferation of pioneer trees over late-successional species (Tabarelli et al. 2012a).

Second, the proposed methodology may help carbon projects assess "leakage" in the landscape where the project was implemented, as well as support the classification of forest patches for field assessments to establish a baseline for the project. It is also possible to identify areas with higher potential for carbon stocks, like those with higher forest neighborhood dominance, contiguity, and proximity to other remnants, as well as monitoring Reducing Emissions from Deforestation and Forest Degradation programs (Alexander et al. 2011). This holds true because the establishment and development of late successional tree species, which have higher wood densities and consequently higher potential for carbon sequestration per unit of area (Chave et al. 2006), are favored. The selection of these promising areas for carbon projects would guide the protection of forest patches against disturbance factors, thereby avoiding setbacks in carbon stocks resulting from degradation.

Third, given that variables we have studied were highly associated with forest structure, our landscape assessment model could be used to estimate the potential provision of water-related ES and, consequently, support the organization of programs of payments for these services. Such programs have been established across the Atlantic Forest (Guedes and Seehusen 2011), as well as in other countries (McQueen et al. 2001; Stanton et al. 2010). In Brazil, the value to be paid to farmers has been defined by land opportunity cost, i.e., the government covers the profit that would be obtained from the land taken from the production system to support the protection of watersheds (Brancalion et al. 2012a). However, the real value of forest remnants for water storage and purification has not been assessed so far. The assessment of patch history and landscape context provides an estimate of the potential of provision of water regulation services, which can be used to define the monetary rewards to farmers that have better protected their remnants.

Fourth, the application of any restoration method is dependent on adequate diagnostics of the resilience of the ecosystem to be recovered (Leite et al. 2013; Tambosi et al. 2014). Forest dynamics and current landscape structure can be used for resilience estimates, which may provide key information to plan and implement restoration programs focused in the market of ES (Palmer and Filoso 2009). Also, methodology may support the selection of areas with higher chances of site recolonization and passive restoration (Holl and Aide 2011), thus providing a better management of the resilience of sites and consequently improving the cost-effectiveness of restoration projects focused on ES (Birch et al. 2010; Tambosi et al. 2014).

Conclusions

The proposed methodology offers an integrated view to evaluate the potential provision of ES by remnant forest patches based on the historical land use and current landscape structure that could be useful in programs of payments for ES, as well as for supporting strategic decisions regarding conservation and restoration of forests in agricultural landscapes. The application of this methodology to an Atlantic Forest region shows that HML can be highly dynamic, and as a result the present forest cover is actually a heterogeneous mosaic of forests of different ages, situated in different landscape conditions, and thus having different quality levels for biodiversity conservation and ES provisioning. Furthermore, the studied landscapes showed a tendency of natural forest cover increase due to a positive balance of dynamics of forest degradation and regeneration. Despite forest cover increase, the current ES potential of forest patches is small, as fewer than 1/4 of them are able to supply their full potential, resulting in a gradient of ES supply which goes far beyond what could be inferred only by forest cover.

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