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Socioecological transition in the Cauca river valley, Colombia (1943–2010): towards an energy–landscape integrated analysis

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Abstract Agroecosystems are facing a global challenge amidst a socioecological transition that places them in a dilemma between increasing land-use intensity to meet the growing demand of food, feed, fibres and fuels, while avoiding the loss of biodiversity and ecosystem services. We applied an intermediate disturbance-complexity approach to the land-use changes of a Latin American biocultural landscape (Cauca river valley, Colombia, 1943–2010), which accounts for the joint behaviour of human appropriation of photosynthetic capacity used as a measure of disturbance, and a selection of land metrics that account for landscape ecological functionality. We also delved deeper into local land-use changes in order to identify the main socioeconomic drivers and ruling agencies at stake. The results show that traditional organic mixed-farming tended to disappear as a result of sugarcane intensification. The analysis confirms the intermediate disturbance-complexity hypothesis by showing a nonlinear relationship, where the highest level of landscape complexity (heterogeneity–connectivity) is attained when disturbance peaks at 50–60%. The study proves the usefulness of transferring the concept of intermediate disturbance to biocultural landscapes and suggests that conservation of heterogeneous and well-connected mixed-farming, with a positive interplay between intermediate level of disturbances and land-use complexity endowed with a rich intercultural heritage, will preserve a wildlife-friendly agro-ecological matrix likely to house high biodiversity and ecosystem services.

Keywords Biocultural heritage · Disturbance ecology · Human appropriation of net primary production · Landscape agroecology · Latin America · Colombia

Introduction

During the last half-century, the increase of global agricultural production has required growing inputs of synthetic fertilizer, irrigation water and cultivated land (Cassman 1999; Tilman et al. 2002). Triggered by the intensification of specialized high-yielding cash crops, such as for the production of livestock feed and more recently agro-fuels, modernized agriculture is following a historical path of land-cover and land-use change (LCLUC) (Lambin et al. 2003). This phenomenon has negative effects, including the contribution to climate change (Chase et al. 2000), soil degradation (Foley et al. 2005), biodiversity loss and alteration of ecosystem services (Sala et al. 2000). Agricultural land-use areas are growing, particularly in the tropics, as a result of current trends in population and consumption growth, which together are predicted to increase the global food, feed and biofuels demand for at least another 50 years (Thomson et al. 2010). These trends may double the amount of land assigned for agriculture and threaten some of the most biodiverse ecosystems in the world (Foley et al. 2005).

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In relation to the food-biodiversity dilemma (Godfray et al. 2010; Cardinale et al. 2012), recent paradigms on biodiversity conservation acknowledge that sustainable farm systems depend on preserving biocultural landscapes (Vandermeer and Perfecto 2007; Franklin and Lindenmayer 2009), that is, on complex geographic areas characterized by biotic and cultural elements which are interrelated and coevolve at different degrees (Pungetti 2013; Agnoletti and Emanueli 2016). These biocultural landscapes have evolved from multigenerational interventions in agroecosystems, but in ways that retain the functioning of key ecological processes (Chazdon et al. 2009). In contrast, it has been argued that natural reserves and set-aside policies have important limitations ensuring biodiversity conservation and ecosystem services, in the face of the above-mentioned challenges (Franklin and Lindenmayer 2009). A successful strategy of biodiversity conservation has to actively incorporate the whole land-matrix currently managed for human uses (Bengtsson et al. 2003; Tscharntke et al. 2012). This involves a wide interdisciplinary research on a new land-sharing approach that goes beyond the classical land-sparing viewpoints in biodiversity conservation (Green et al. 2005; Matson and Vitousek 2006), to focus on the associated biodiversity that traditional as well as new types of wildlife-friendly farming may provide (Tscharntke et al. 2005; Perfecto and Vandermeer 2010).

The global expansion of sugarcane production may serve as an example for the socioecological challenges that industrial agriculture is facing. According to FAO, 27 million hectares of sugarcane was harvested in tropical and subtropical zones of the world (2014).

Between 2005 and 2014, nearly 1900 million tons of sugarcane was produced per year, accounting for close to two-thirds of the global sugar production (FAO 2015). Colombia is a country where sugarcane production is expanding. In the coming years, it is expected to grow at annual rates of 10% or more (Bendeck 2013). With five million hectares of land suitable for agriculture and an increasing capacity to produce ethanol, this scenario makes Colombia a priority region for biodiversity conservation and sustainable development.

The Cauca river valley, one of the most populated and economically important regions in Colombia, has suffered massive land-cover changes and an increase in land-use intensity driven mainly by the establishment of sugarcane plantations (Delgadillo-Vargas et al. 2016). We investigated the land-use change in the period 1943–2010 and researched how different levels of human disturbance in agroecosystems are associated with landscape processes. To explore the relationship between anthropic disturbances (measured in energy terms) and land-matrix ecological functioning (measured by land-cover metrics) in the Cauca

river valley (Palmira municipality case study), we applied an energy–landscape integrated analysis (Marull et al. 2016a), in order to elucidate the role of sustainable farm systems on maintaining biodiversity and ecosystem services in human-transformed landscapes.

The Cauca river valley

Human occupation of Colombia dates from at least 14500 BP. It was from 2000 BP that increasingly complex societies with well-differentiated cultures accompanied by agricultural intensification became established (Melo 1998). The Cauca river valley was originally a dry tropical forest mixed with rich vegetation suitable for pastures (Berrío et al. 2002). The earlier agricultural system was in place with pre-Hispanic cultures, and this was maintained throughout the Spanish conquest. The main biological changes were the introduction of European crop varieties (Patiño 1977; Zuluaga 1992) and domesticated animals grazing savannah which, as a consequence, replaced the existing dry tropical forests. The origins of industrial agriculture are to be traced back to 1850, when the European colonial power established a trade-dependent economy. In this period, family enterprises were established that would later cement into the agro-industrial model that the region adopted towards the mid-twentieth Century. Japanese immigration since the early twentieth century accelerated the introduction of technological innovations such as machineries for cereal cropping, while livestock specialization (with African pastures and cattle breeding) and better transport infrastructure (such as steam boats on the Cauca river, railways and the opening of the Panama channel) drove the globalization of locally traded commodities (coffee, cocoa, tobacco) and the consequent expansion of the agricultural frontier.

Between 1931 and 1973, agricultural industrialization advanced rapidly. The international situation created by the Cuban Revolution (which gave local farmers the opportunity to participate in the US sugar market, although this never really happened) and the implementation of US cooperation programs such as Peace Corps, and Alliance for Progress resulted in a standardization of agrarian policies for development (Piñeiro et al. 1982). For instance, the overlapping pursuit of agricultural modernization by US Department, Colombian Ministry of Agriculture and Rockefeller Foundation founded the modern agricultural development in Colombia. In the Cauca river valley, technological innovation and intensification were influenced by US-based development works (Lorek 2013), such as Tennessee Authority which promoted the creation of Corporación Autónoma Regional del Valle del Cauca (CVC), a government body managed, in practice, by local businessmen. Between 1954 and 1995, CVC executed major infrastructure

works that were the basis for the expansion of a modern industrial agricultural system focused on cash crop production, with considerable negative effects on the environment.

Between 1950 and 1972, the rural population in Colombia decreased from two-thirds to one half, and the contribution of agriculture to the GDP diminished from 40 to 26%. This was associated with the establishment of a capitalist-based production system focused on cotton, sugarcane, rice and soy, which grew at an annual production rate of 8.2%, that is, five times faster than traditional crops such as beans, yucca and banana, and three times faster than mixed traditional-capitalist crop systems, such as corn, coffee, potato, wheat, tobacco and cocoa (Escobar 1996). The Cauca river valley economy was characterized until 1950s by ranching and small/medium sugar production, with multi-purpose extensive cattle farms (Rivera et al. 2007). The development of irrigation systems and the construction of dikes for flood regulation facilitated this transition, which after the 1970s resulted in a capitalist industrial system with synthetic fertilization, pesticides, mechanization and water pumping. Farmland was redesigned for this purpose, sugarcane expanded, and agroecological processes and work relations were reconfigured. Agrarian industrialization and associated urban sprawl along the Cauca valley (Delgadillo-Vargas et al. 2016) modified the landscape ecology parameters.

Between 1974 and 1993, the most drastic changes took place with the intensive implementation of a new economic system, namely the economic liberalization in the 1980s, along with a subsidized sugarcane sector. All flatland should be devoted to agricultural intensification following the Green Revolution approach based on technological optimization, instead of recognizing the role of pre-Hispanic cultures or local agrarian expertise in agricultural and landscape management. Cauca's managerial elite was able to reconduct government policies towards their agro-industrial interests (such as the CVC). The managerial lobby—represented by family enterprises in its early stages and lately by the powerful Asociación de Cultivadores de Caña de Azúcar—has controlled the sugarcane business and, since 2005, also the agro-fuel market. With its strong influence on Colombian policy-making, infrastructure development and agrarian reconfiguration, it is a key agent in the socioecological transition. In this article, we investigate the landscape ecology impacts of the agricultural transition process in the Cauca river valley.

Research approach and methods

Case study: the Palmira municipality

Cauca river inter-Andean valley is located in the southwest of Colombia and covers an area of approximately 448,000 ha. This bio-geographic unit is one of the five enclaves of tropical dry forests in Colombia, one of the scarcest, most transformed and threatened ecosystems in the country (IAvH 1998). The municipality of Palmira (Fig. 1) is located in the southern area of the valley, with an extension of 100,965 ha and elevations ranging between 900 and 5000 m above sea level. The flat area has 54,000 ha and presents a tropical climate characterized by relatively high and uniform temperatures all year round (24 °C average). Annual precipitation average oscillates between 1500 and 800 mm (CVC 2000).

Several sources were used to estimate land-uses in the Palmira municipality between 1943 and 2010, such as national censuses by the Instituto Geográfico Agustín Codazzi, Corporación Autónoma Regional del Valle del Cauca and the city hall of Palmira. For 1943 the data were taken from aerial photographs. Historical information regarding sugarcane production, yields and management practices was obtained from governmental agricultural censuses, statistical yearbooks and organizations such as Asociación de Cultivadores de Caña de Azúcar and Sugar Cane Research Center. Consulted documentary sources included: deeds, agricultural censuses, general statistics collated by both state and private organizations regarding sugarcane cultivation and processing. Private archives of sugar mills and haciendas in the region were also taken into consideration (Delgadillo-Vargas et al. 2016).

Theory: the intermediate disturbance hypothesis

The intermediate disturbance hypothesis is a non-equilibrium explanation to understand the maintenance of biodiversity in ecosystems (Wilson 1990). It is one of several major hypotheses that explain biodiversity, and it is widely referred to in landscape ecology (Huston 2014). However, it is not uncontested and is in fact criticized by some authors for both empirical and theoretical reasons (Fox 2013). There are different definitions of disturbance (van der Maarel 1993), but a common one is the destruction (or harvest) of biomass (Calow 1987), leading to the opening-up of space and resources for recolonizing species—an approach that foregrounds the variation of its spatial extent in ecosystem communities (Wilson 1994). Coexistence would require spatially patchy disturbance that leads to a trade-off between species able to perform best at different stages of post-disturbance succession (Chesson and Huntly 1997). At intermediate disturbance frequencies both competitive and dispersal species may coexist (Barnes et al. 2006).

Agroecosystems offer habitats to different species, creating a great amount of ecotones (Benton et al. 2003), as well as more permeable land-matrix allowing dispersion among local populations (Shreeve et al. 2004). Thanks to
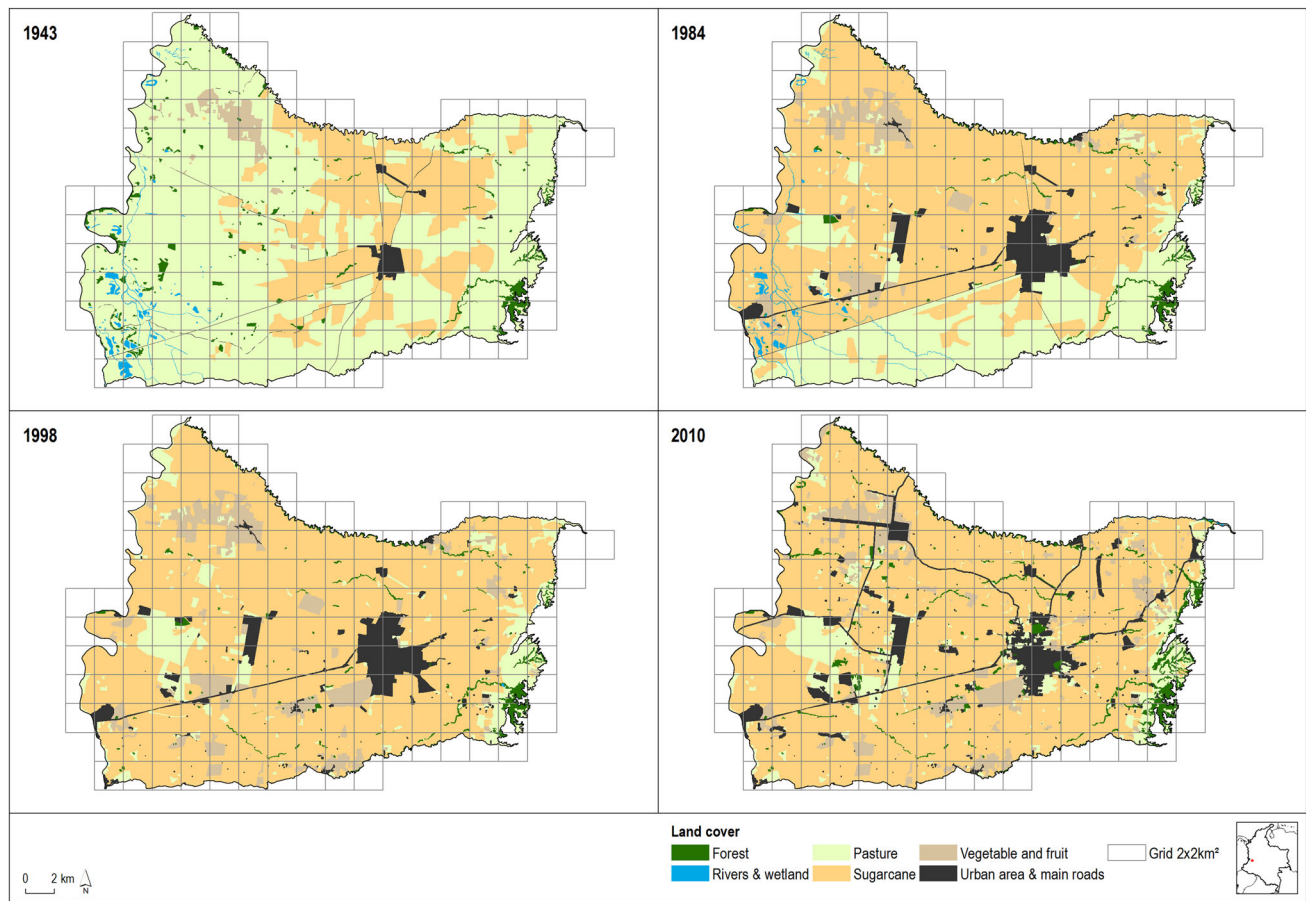
Fig. 1 Land-cover change in the Palmira case study (1943–2010)—Cauca river valley, Colombia. *Source* Our own from Instituto Geográfico Agustín Codazzi, Corporación Autónoma Regional del Valle del Cauca and city hall of Palmira

the edge effect and high connectivity, a complex land-cover pattern may host greater biodiversity than more uniform landscapes (Harper et al. 2005). In order to manage agroecosystems, farmers invest over the land-matrix both energy and their knowledge that shape the spatial patterns of a landscape embodied with a biocultural heritage (Marull et al. 2016b). The impact of ecological disturbance resulting from farming on biodiversity may be either positive or negative, depending on the intensity and shape of these socio-metabolic flows and the vertical/horizontal complexity of land-use mosaics (Swift et al. 2004). Understanding and managing correctly these patchy mosaics require an interdisciplinary approach to biocultural diversity (Parrotta and Trosper 2012) embedded in agroecological landscapes (Matthews and Selman 2006).

Method: the energy–landscape integrated analysis

We present an intermediate disturbance-complexity model (IDC) of how landscape processes are affected by different levels of anthropogenic disturbances on ecosystems, when farmers alter net primary production (NPP) through land-use

change. We have adopted the proposed landscape continuum model as a starting point (Fischer and Lindenmayer 2006). The method relies on a series of topological analysis of land-cover maps, it has entirely been formalized using mathematical language, and it has been developed and implemented using geographic information systems (GIS).

We understand IDC as a measure of the capability of the land-matrix to host biodiversity—the model includes land-cover metrics (landscape patterns and processes), but also the energy flow available for the trophic levels in a study area. The conservation of heterogeneous and well-connected land-matrix with a positive interplay between human energy disturbances and landscape complexity would be able to hold high species richness (Marull et al. 2016a).

In order to test the intermediate disturbance hypothesis, we analysed a set of land-cover metrics as a function of unharvested NPP. To do this, we obtain a new variable Le (landscape ecology metric) using principal components analysis (PCA). Once we have Le, we will perform a regression analysis with the human appropriation of net primary production—HANPP, a measure of the extent to

which humans modify the amount of NPP, that is, the energy available for the agroecosystem's trophic chains after biomass harvest (Haberl et al. 2007a; Krausmann et al. 2013). The following sections present the key components of this energy–landscape integrated analysis.

Metrics: assessing landscape patterns and processes

We use land-cover maps to calculate the following landscape pattern metrics (Dupras et al. 2016): land-cover richness (LCR), Shannon–Weaver Index (H') and effective mesh size (EMS). LCR refers to the number of different patch types per sample cell; the more land-cover classes there are, the more diverse the study site is.

H', commonly used in ecology (Vranken et al. 2015), accounts for land-cover equi-diversity and is based on two components (Eq. 1): the number and proportion of different patch types:

$$H' = -\sum_{i=1}^{k} p_i \log_k p_i \tag{1}$$

where k is the total number of land-covers in the study area, and p_i is the proportion of the land-cover i in a specific sample cell.

EMS (Jaeger 2000) stands for the probability that any two randomly chosen points in a region are connected (Eq. 2). In that sense, the more anthropogenic barriers (i.e. urban areas, infrastructures) fragmenting the landscape, the lower the probability that two points are connected, and the lower the EMS.

$$EMS = \frac{\sum_{i=1}^{k}(A_i^2)1000}{\sum_{i=1}^{k}(A_i)} \tag{2}$$

where A_i is the area of each land-cover polygon.

The following assessment of landscape processes (Lindenmayer and Fischer 2007) was based on the original ecological connectivity model proposed by Marull and Mallarach (2005). This assessment relies on defining a set of ecological functional areas (EFA) and a computational model of cost distance of displacement, which includes the effect of anthropogenic barriers, considering the type of barrier, the range of distances and the kind of land-cover involved. The model was applied to the historical land-cover maps, comprising the whole sample cells required with a GIS. The model defines a basic Ecological Connectivity Index (ECI_b) in a normalized scale from 0 to 10 (Eq. 3). This ECI_b emphasizes the role played by the land-matrix:

$$ECI_b = 10 - 9\left[\ln(1 + x_i)/\ln(1 + x_t)^3\right] \tag{3}$$

where x_i is the value of the sum of the cost distance by pixel and x_t the maximum theoretical cost distance. Then, ECI_a is the absolute Ecological Connectivity Index (Eq. 4):

$$ECI_a = \sum_{m=1}^{m=n} ECI_b/m \tag{4}$$

where m is the number of EFA considered. This index helps to emphasize the role played by all sorts of EFA in keeping up ecological connectivity (Pino and Marull 2012).

Analysis: the intermediate disturbance-complexity model

A focus on landscape functionality stresses the spatial dimension of biodiversity, the interplay between disturbances and land-cover heterogeneity–connectivity and the role of agro-ecological land management in ecosystem service provision (Tscharntke et al. 2005). This perspective relies on the interplay between patch disturbance and land-use diversity as the key mechanism that actually matters in biodiversity maintenance (Harper et al. 2005; Loreau et al. 2010). However, much of this biological diversity is only perceived at scales larger than plot or farm level and depends on landscape-wide heterogeneity of land-covers.

We use a modification of H' to account for landscape heterogeneity (i.e. horizontal structure). It is used for looking at agroecosystems as the spatial 'imprint' of their social metabolism. We calculate L to capture the equi-diversity of potential habitats into sample cells (Eq. 5):

$$L = \left(-\sum_{i=1}^{k} p_i \log_k p_i\right)(1 - p_u) \tag{5}$$

where k is the number of different land-covers (potential habitats). The existence of built-up land-cover p_u results in a loss of potential habitats. Thus, p_i is the proportion of non-urban land-covers i into every cell.

L can be improved (Eq. 6), when data are available, using the following algorithm:

$$Le = \left(aL + b\frac{ECI}{10}\right)1/(a + b) \tag{6}$$

In this way, we obtain a new indicator Le (Marull et al. 2015), capturing landscape patterns (L, heterogeneity) and landscape processes (ECI, connectivity), using PCA (where a and b are the empirical coefficients).

We use HANPP as a measure of human disturbance (i.e. affecting vertical structure). HANPP measures the effect of land conversion and harvest on biomass flows in terrestrial ecosystems of a defined area of land, in other words the combined effect of human-induced land-cover change.

From an ecological perspective, HANPP is a measure of the impact of land-use on the availability of trophic energy (biomass) for heterotrophic food chains and as a resource for building up biomass stocks in terrestrial ecosystems used by humans. HANPP thus has been proposed as an

indicator for land-use induced pressures on biodiversity and is a suitable indicator for disturbance (Haberl et al. 2007b; Krausmann et al. 2009). HANPP is calculated (Eq. 7) using the following identities (Haberl et al. 2014):

$$
\begin{aligned}
\text{HANPP} &= \text{HANPP}_{luc} + \text{HANPP}_{harv}; \text{HANPP}_{luc} \\
&= \text{NPP}_0 - \text{NPP}_{act}
\end{aligned}
\tag{7}
$$

where HANPP_{harv} is the NPP appropriation through harvest, and HANPP_{luc} is the change of NPP through human-induced land conversions. HANPP_{luc} is defined as the difference between the NPP of the potential (NPP_0) and actual (NPP_{act}) vegetation. HANPP is associated with each land-cover of the study area, so that HANPP is calculated multiplying a fixed coefficient (w_i) for some land-cover i by the surface occupied by this land-cover (Eq. 8):

$$
\text{HANPP} = \sum_{i=1}^{k} w_i p_i
\tag{8}
$$

where w_i denotes the weight of land-cover i and p_i the proportion of land-cover i in the study area. Variations in HANPP not only depend on the variations of p, but on the variations of w as well.

HANPP values have been estimated after assessing different NPP and harvested amounts. NPP_0 values have been derived from the Latin America—aggregate of a global HANPP assessment (Krausmann et al. 2013, available at http://www.uni-klu.ac.at/socec/inhalt/5605.htm) and applied to each land-cover. NPP_{act} values have been estimated as the sum of harvested and unharvested values. Harvest ratios from each land-cover and across the time points have been transformed into energy values. To this, conversion factors as the residue/product ratio losses (Guzmán et al. 2014) have been applied to account for unharvested biomass.

It is important to note that the use of HANPP as a measure of disturbance could have limitations in specific situations, in particular as HANPP values can become negative when $\text{NPP}_{act} > \text{NPP}_0$ due to fertilizer input or irrigation. This is not the case, however, in the Cauca river case.

Finally, the intermediate disturbance-complexity (IDC) model (Eq. 9) combines the landscape structure (L) with the biomass (NPP) available for other species (1-HANPP/ 100):

$$
\text{IDC} = L * E = L(1 - \text{HANPP}/100)
\tag{9}
$$

where E is the energy flow, and L is the energy 'imprinted' in the landscape structure (L can be substituted by Le, including functional attributes of the landscape).

The IDC tries to improve our understanding of the functioning of agroecosystems and the subsequent effects on biodiversity (Marull et al. 2015), revealing how and why different management leads to turning points in the relationship of landscape energy profile with landscape patterns and processes.

Results and discussion

Land-use changes

Around the mid-twentieth century (1943), the region was characterized by a diversified landscape around the urban core representing the city of Palmira (Fig. 1). A ring of pasture (38,292 ha) and also sugarcane (12,390 ha) was found outside the city, as well as practicing traditional crop-farming (983 ha) mixed with cattle-farming in a complex multi-use forested-pasture. The municipality held a high density of wetlands in the south-west (475 ha), connected by a network of river corridors, and some forest patches distributed everywhere but mostly in the north-east (1,305 ha). This diversified landscape can still be recognized in the land-use map of 1984 (Fig. 1), when sugarcane was the main land-cover (33,590 ha) for the first time, especially in the northern region, representing 61.9% of the total area (Table 1). In 1984, the process of urban sprawl had already begun (2807 ha), and urban areas covered an area four times larger than in 1943 (from 1.3 to 5.2% of the total area). Wetlands (262 ha) were still maintained, although with a reduction in size, while pastureland decreased from 70 to 25%.

Developed land (urban areas and infrastructures) grew quickly after 1984 (3177 ha in 1998, and 4285 ha in 2010). As a result of the extensive monoculture of sugarcane (40,733 ha in 1998, and 38,723 ha in 2010), pastureland decreased to less than 10% and the wetland system disappeared completely (6 ha in 1998 and 0 ha in 2010). Urban development and sugarcane production invaded large parts of the former pastures and natural areas. Indeed, the cost of this urban-sprawl and agriculture intensification was the large reduction of habitats and their associated biodiversity as forested-pastures have decreased by 86.7% (Table 1).

Moreover, these landscape transformations may be the result of significant changes in the sociocultural dimensions of LCLUC. Above all, the increasing presence of sugar plantations seems to be connected to land tenure and distribution. In 2009, the municipality had a land Gini coefficient between 0.85 and 0.91 (that means high inequality in land distribution), where rural properties larger than 50 ha made up 70% of the total land, and reported some of Colombia's highest number of properties per owner (IGAC 2012).

LCLUC was also related to changes in sugarcane productivity. In 1943, annual sugarcane production was about

Table 1 Land-cover change and land-cover metrics in the Palmira case study (1943–2010)

Land-cover	1943		1984			1998			2010		
	ha	%	ha	%	1943 = 100%	ha	%	1943 = 100%	ha	%	1943 = 100%
Forest	1305	2.4	931	1.7	71.3	1048	1.9	80.3	1942	3.6	148.8
Wetland	475	0.9	262	0.5	55.1	6	0.0	1.3	0	0.0	0.0
Pasture	38,292	70.5	13,629	25.1	35.6	5111	9.4	13.3	5079	9.4	13.3
Veg. and fruit	983	1.8	2825	5.2	287.5	4208	7.7	428.2	4221	7.8	429.5
Sugarcane	12,390	22.8	33,590	61.9	271.1	40,733	75.0	328.8	38,723	71.3	312.5
Rivers	162	0.3	244	0.4	150.3	10	0.0	6.0	35	0.1	21.4
Urban area	694	1.3	2807	5.2	4043	3177	5.9	457.6	4285	7.9	617.2
Total	54,301	100.0	54,288	100.0	1275.2	54,294	100.0	1315.5	54,284	100.0	1542.6

Land-cover metric	1943 (A)	1984 (B)	1998 (C)	2010 (D)
Land-cover richness—LCR (n°)	3.03	3.85	3.60	4.44
	–	A	A	ABC
Shannon–Wiener Index—L	0.26	0.25	0.23	0.28
	–	–	–	–
Effective mesh size—EMS (km^2)	10,759.90	14,717.34	8111.08	4492.83
	CD	ACD	D	–
Ecological Connectivity Index—ECI	6.12	5.56	3.82	3.63
	BCD	CD	–	–
Landscape ecology metric—Le	0.43	0,40	0.31	0.32
	CD	CD	–	–
Human appropriation of NPP—HANPP (%)	55.12	76.68	83.31	81.52
	–	A	AB	AB
Intermediate disturbance complexity—IDC	0.20	0.09	0.05	0.06
	BCD	CD	–	–

The results are based on two-tailed tests assuming equal variances with a significance level of 0.05. For each significant pair, the key under the category (A, B, C, D) shows up beneath the category with a major average value. Using Bonferroni adjustment, tests have been adjusted for all pairwise comparisons

NPP net primary production

43 t ha^{-1} year^{-1}, whereas by 1960, these values had doubled. Between 1960 and 1981, there was a substantial increase in the annual sugarcane production (from 88 to 126.4 t ha^{-1} year^{-1}), as a result of the introduction of new technology (machinery, chemical fertilization, irrigation, pre- and post-harvest burn-down). Between 1982 and 1992, production remained around 120 t ha^{-1} year^{-1} (Fig. 2). Over the last decade, there has been a steady decrease which is related to climate conditions, soil degradation, age of cutting and crop management (Delgadillo et al. 2016). Nevertheless, the productivity of sugarcane in the last decade had an average value of 115.72 t ha^{-1} year^{-1}, one of the highest values in the world (ISO 2012).

Landscape agroecology

As a result of urban sprawl and agriculture intensification (land-use consumption, irrigation and drainage, pesticides and nitrogen fertilization), the ecological quality of the landscape has continued to decline. Both processes of land-use intensity (here illustrated by nutrient management) and spatial structure of land-use (measured by landscape ecology metrics) contribute to biodiversity loss and are intricately linked.

Before the first decade of the twentieth century, sugarcane production did not involve an enormous use of agrochemicals. It was only when a significant rise in the use of synthetic fertilizer became feasible that the valley specialized in sugarcane cultivation (Fig. 3). Nitrogen fertilizer was applied at a rate between 154kgN ha^{-1} year^{-1} in the 1940s and 246kgN ha^{-1} year^{-1} in 2010 with a very low N efficiency (63% in the eighties, 53% in the nineties and 40% in 2010). Later in the forties, other fertilizers appeared that were rich in K, with an N-P-K relation of, for example, 6–8–16 (400 kg ha^{-1} in average), and finally, potassium chloride (KCl) emerged in the sixties, and since then, this has become the main source of K for the sugarcane crop, applied at average annual doses of 150 kg ha^{-1}. As for phosphate, compound P fertilizers were commonly applied to sugarcane crops with recommended doses between 500 and 2000 kg ha^{-1}, depending on the soil type. In the seventies, triple superphosphate entered the market and it became—along with the urea and potassium chloride—the basic formula for sugarcane crop fertilization (Guardiola 1995).
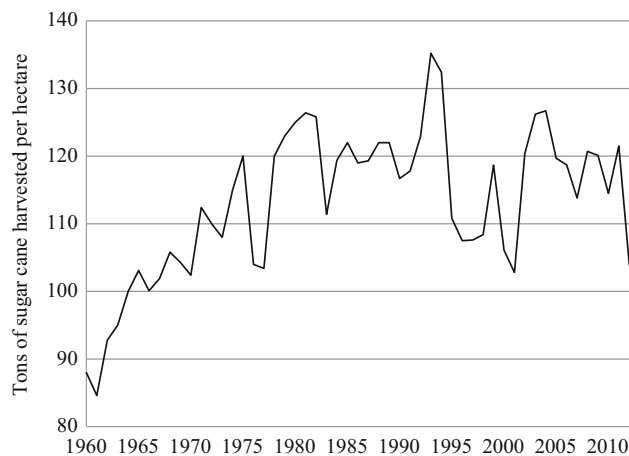
Fig. 2 Sugarcane production in the Palmira case study (1960–2012). *Source* Our own from Asociación de Cultivadores de Caña de Azúcar (Asocaña)

Sugarcane also demands high quantities of water. The annual irrigation volume in the observed period ranged between 12,000 and 20,000 m^3 ha^{-1}. The combination of the application of large amounts of synthetic fertilizers, a low N efficiency and the irrigation practices caused losses of N by leaching which was a main cause for the Cauca river's heavy pollution. Intensive chemical fertilization may have had important impacts on the biodiversity.

As expected, in general terms, land-cover metrics show a decrease in their values during the studied period (Table 1). LCR is higher in 2010 than in the past, especially when compared to 1943. This means there was more land-cover diversity in 2010, although less land-use complexity than in 1943 (decrease in integrated multi-used pastures). But L has remained stable for the entire area; however, there has been an increase in certain areas of the ring around the city of Palmira. EMS has decreased through time showing that fragmentation occurred in the land-matrix.

ECI sharply decreased between 1943 and 2010 (Table 1) due to the impact of the construction of the transport network and the urban development of low-density scattered suburbs. In 1984, the study area still possessed a high-ecological connectivity, similar to that in 1943. The allocation of the increasing population to new urban developments scattered throughout the region since 1984, and the construction of new highways and roads to connect them and facilitate sugarcane commercialization acted as new barriers fragmenting the territory to such a degree that the connectivity was practically erased near Palmira. Substantial negative changes in connectivity levels occurred in the former pasture and agriculture ring around the city of Palmira. Today, acceptable levels of connectivity are maintained only in the very small
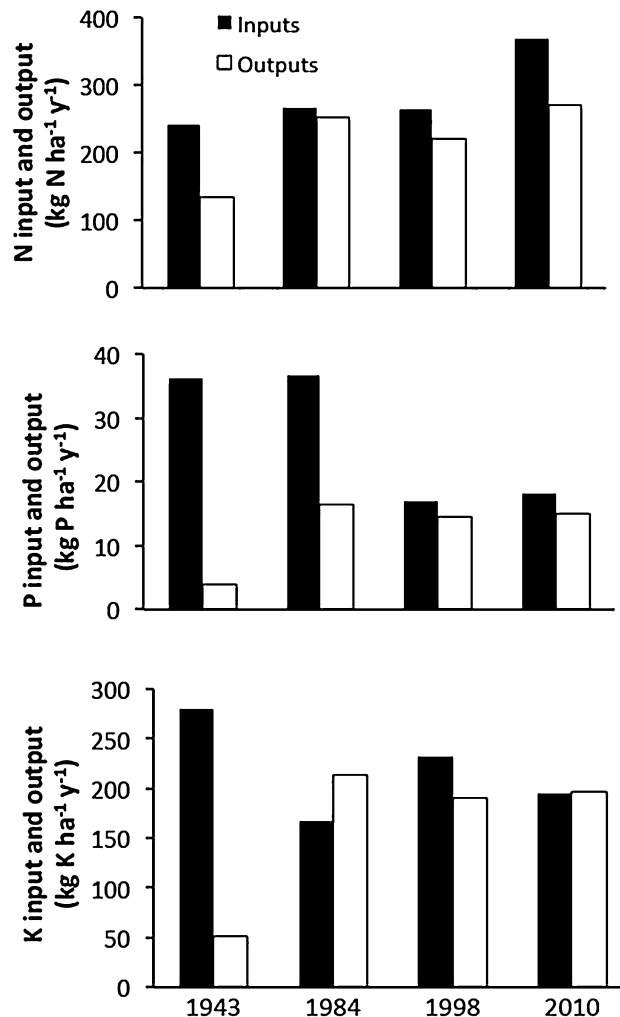

Fig. 3 Sugarcane fertilization (N, P, K) in the Palmira case study (1943–2010). *Source* Our own from Delgadillo-Vargas et al. (2016)

surviving forest and pasture areas of the peripheral ring of the study area.

All these landscape changes are expected to have detrimental effects on biodiversity, as the mixed crop-pasture matrix changed and the forest remnants decreased. Although there are no regional long-term studies available that evaluate the response of fauna communities to these changes, several studies have found that species richness is higher on mixed crop-pastures systems than in sugarcane monocultures for both arthropods (Ramírez et al. 2004) and birds (Cárdenas Carmona 2013). However, more long-term taxa-specific studies are necessary in order to fully comprehend the impact of these socioecological transitions on biodiversity.

Furthermore, the impact of sugarcane expansion on the landscape should also be considered in the context of associated changes on the agricultural frontier. Sugarcane agroindustry has not only affected the ecological value of

highly productive agricultural lands in the Cauca river valley, but has also pushed traditional agriculture and livestock to lands of marginal productivity in the surrounding mountains, therefore increasing deforestation and habitat fragmentation at larger scales, possibly having consequences for the metapopulation dynamics of the local biodiversity.

Energy–landscape integrated analysis

To conclude the IDC analysis, we studied the relationships between HANPP and the land-cover metrics. The correlations (Table 2a) among land-cover metrics (LCR and L; EMS and ECI) aim us to carry out a PCA of the variables involved. Table 2b shows that the major contributors for the first component (C1) are LCR and L—landscape patterns metrics—and for the second component (C2) EMS and ECI—landscape processes metrics. These results have led us to consider only two variables, L and ECI, so that the two components of the landscape are represented, which include landscape heterogeneity and ecological connectivity. Once we have reduced the dimensions of the land-cover metrics, we obtain only one component resulting from the linear combination of L and ECI (Marull et al. 2015). We call this new component landscape ecology metric (Le). Le showed a decline between 1943 and 1998 mainly due to the loss of landscape complexity (Fig. 4).

We have one Le and HANPP value for each sample cell and time period. The issue is how to interpret the empirical data according to the density of pair values of HANPP-Le. We assume that Fig. 5 draws the shape of all possible values adopted by the theoretical relationship between farming disturbance and landscape complexity (Marull and Font 2017), where disturbance-complexity (IDC) value of a given territory can be represented (Fig. 4). HANPP expresses an average number for each sample cell, but can be obtained with different land-cover combinations. Hence, it is important to include land-cover metrics in IDC calculation.

The results confirm the intermediate disturbance hypothesis by showing a hump-shaped nonlinear relationship (Marull et al. 2015), where the highest level of IDC is attained when HANPP tends to 50–60%. IDC shows high values in 1943, medium values in 1984 and low values in 1998 and 2010 (Table 1; Fig. 5), suggesting a decrease in the capability of the land-matrix to host biodiversity, due to a less functional landscape structure and more appropriation of NPP over time.

Figure 5 shows the dynamics of the more important variables used in this study: a notable increase in human disturbance to very high levels of HANPP slightly above 80%, mainly due to sugarcane expansion and intensification (monoculture), and a decrease in landscape complexity

(Le) due to the loss of traditional integrated agriculture (polyculture) and urban development (Table 1). The results confirm that the historical trend which attained the highest IDC in 1943 was also linked to shifts towards sugarcane cultivation that had historically been characterized by much higher HANPP values than pastureland, forestland or wetland values. Nonetheless, the relationship seems to be differentiated locally (Fig. 4), which calls for a further geo-historical study of this complex interplay between driving forces, socio-metabolism, integrated land-uses and landscape ecology. Scientifically based agro-ecological models have to critically examine, select and adapt biocultural heritage to current contexts and new challenges for a sustainable rural development.

IDC has been previously tested in a case study from Mallorca Island, in the Mediterranean bioregion, across a similar time span (1956–2011) and scale (3 × 3 and 1 × 1 matrixes) (Marull et al. 2015). The results are similar in the two studies: while in the past—for the Cauca Valley up to 1984—high values of landscape metrics (patterns and processes) were often associated with medium range levels of human disturbance (HANPP), the present landscape metrics have worsened, while HANPP has either increased towards high levels of disturbance (in both cases, through intensive industrial farming) or decreased in reforested or abandoned lands (mainly in Mallorca).

Socioecological transition

In many parts of the world, farming is still based on site-specific traditional knowledge of how to develop sustainable farming systems that have been adapted to local conditions (González de Molina and Toledo 2014). Thanks to the integrated use of natural resources, this biocultural heritage helps rural communities to manage functional landscapes while meeting subsistence needs, without depending to a large extent on external technological inputs. This traditional knowledge is losing significance, and the corresponding biocultural landscapes are at risk, in particular in the tropics, with consequences for biodiversity.

Protecting biocultural landscapes but allowing their isolation from the rest of the territory will decrease ecological connectivity, with likely impacts on biodiversity. Remaining forested-pastures show very low levels of ecological connectivity because they have become isolated due to traditional rural abandonment and sugarcane intensification, the increasing construction of transport networks and scattered urban villages. This loss of ecological connectivity also stresses the importance of natural corridors, like riparian zones, in maintaining the ecological processes within the landscape. A recent review by Tscharntke et al. (2012) emphasizes that spatial heterogeneity (horizontal but also vertical) creates the ecological dissimilarity that

Table 2 Relationships among land-cover metrics using principal component analysis (PCA) in the Palmira case study (1943–2010)

(a) Correlation Analysis between variables (1943, 1984, 1998, 2010)*

Land-cover metric		L	LCR	EMS	ECI
Shannon–Wiener Index (L)	Cor. (Pearson)	1.000			
	Sig. (bilateral)				
	N	0			
Land-cover richness (LCR)	Cor. (Pearson)	0.427	1.000		
	Sig. (bilateral)	0.000	–		
	N	402	0		
Effective mesh size (EMS)	Cor. (Pearson)	−0.092	−0.114	1.000	
	Sig. (bilateral)	0.064	0.022	–	
	N	402	402	0	
Ecological Connectivity Index (ECI)	Cor. (Pearson)	0.142	−0.195	0.459	1.000
	Sig. (bilateral)	0.004	0.000	0.000	–
	N	402	402	402	0
Control variable (year)	0.025		0.416	−0.434	−0.628

(b) Principal component analysis**

Land-cover metric	1943		1984		1998		2010	
	C1	C2	C1	C2	C1	C2	C1	C2
L	−0.018	0.789	0.895	−0.041	0.878	0.196	0.833	−0.004
LCR	−0.077	0.793	0.790	0.165	0.819	−0.156	0.860	−0.081
EMS	0.793	0.110	−0.078	0.920	−0.388	0.648	0.035	0.820
ECI	0.854	−0.237	0.463	0.610	0.288	0.817	−0.120	0.820
Variance (%)	34.107	32.967	41.136	31.167	41.862	28.752	36.222	33.779

* Correlations are shown considering each time period and all data together

** We obtain a new variable: landscape ecology metric (Le) using the PCA (Le = (a L + b ECI/10)1/2), taking into account landscape patterns (heterogeneity—L) and processes (connectivity—ECI) as proxy of biodiversity in human-modified ecosystems

determines landscape-wide biodiversity. Spatial spillover effects through the movement of organisms and resources across habitats can facilitate their persistence and adaptation to human-managed landscapes. The combination of traditional integrated multi-uses in forested-pastures, with heterogeneous and well-connected landscapes in urbanized regions, enables spatial and temporal insurance, providing higher stability and resilience to ecological processes (Loreau et al. 2010). Therefore, a wildlife-friendly agro-ecological matrix can be seen as a useful strategy to enhance biodiversity and food security.

The policies applied by the government and the sugarcane sector to increase production had not considered the implications on sustainability. Indeed, the monocultural production system has not been able to maintain ecosystem services (i.e. water cycle and soil fertility, ecological processes and biodiversity) as well as traditional agriculture did, nor has it been able to prevent urban sprawl from arising during the real-estate boom in recent decades. The result has been the abandonment of traditional agricultural activities and urban sprawl into forested-pastures. Faced with the situation of progressive impoverishment, farmers, and especially their descendants, prefer to sell their land or designate it to

sugarcane commercial production than to keep traditional mixed-farming, which cannot match the rapid income offered by the latter. Land managers and decision-makers should understand that they are losing biocultural landscapes that provide important ecosystem services. In addition to fibre and food, these landscapes maintain culture and biodiversity; they recycle nutrients, regulate local microclimates, control local hydrological processes, affect the abundance of undesirable organisms and detoxify chemical harmful substances (Alam et al. 2014).

The intensive sugarcane model followed over the last decades, based on a well-planned commercial distribution and an efficient transportation network, has had an extremely negative effect on the ecological quality of the territory, as well as on its long-term sustainability. As a result of the new rural development policy, which promotes large-scale agribusiness projects at the expense of traditional agroecosystems (with serious implications for land tenure), this biocultural landscape loss could worsen and spread to other regions of the country. A similar socioecological transition is ongoing in most of the tropics and represents an important pressure challenging efforts of biodiversity conservation and climate change mitigation.

Fig. 4 Landscape ecology metric (Le) and intermediate disturbance complexity (IDC). Territorial distribution expressed by 2 × 2 km² sample cells in the Palmira case study (1943–2010). *Source* Our own
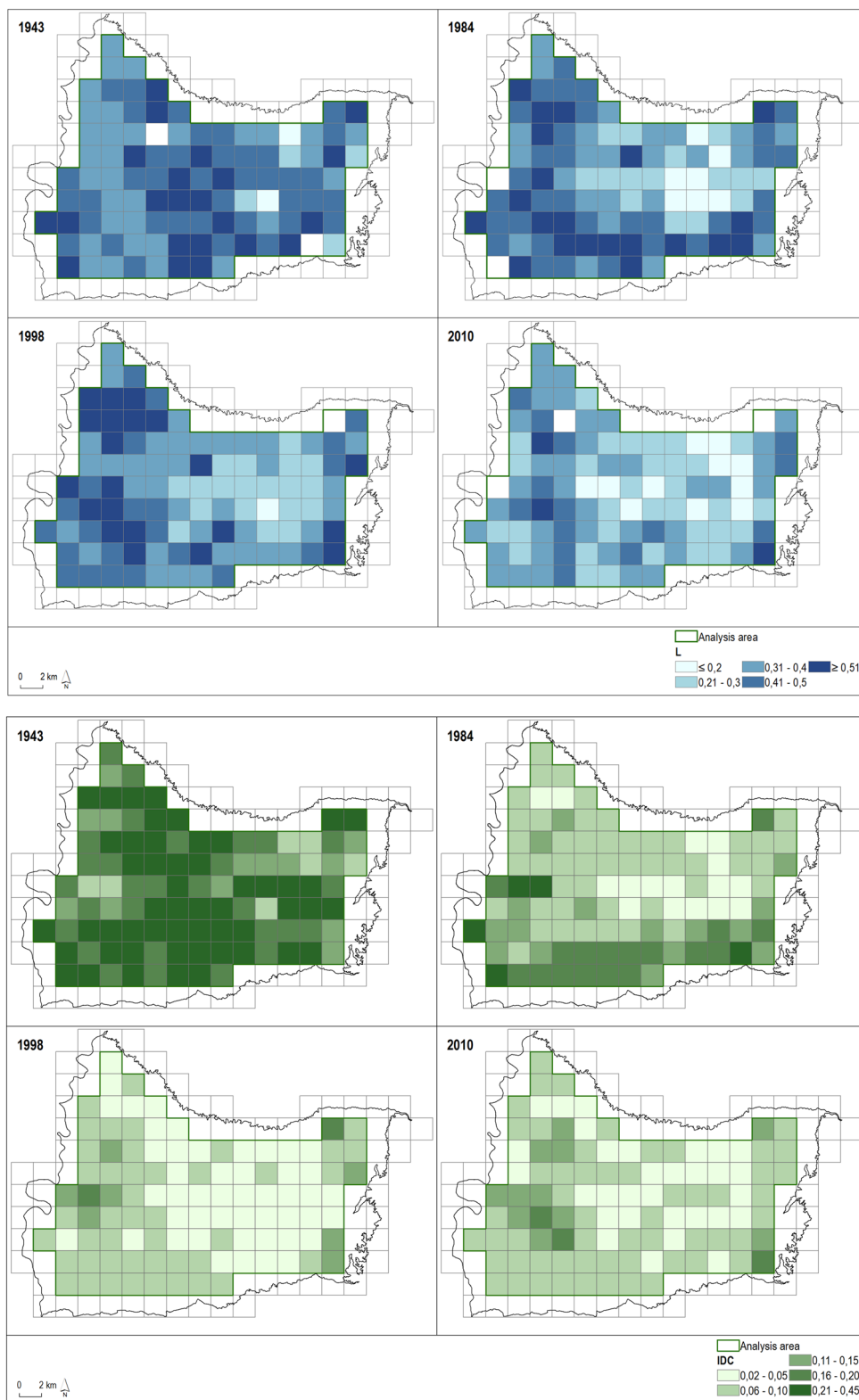
Studies like our investigation in the Cauca river valley are of high significance for Colombia's current political context. As this paper is being written, the Colombian government and the Revolutionary Armed Forces of Colombia—People's Army (FARC) are signing a peace treaty that will open a new era for the country's rural development. The first point of the agenda entails a comprehensive rural reform, which aims to reduce massive
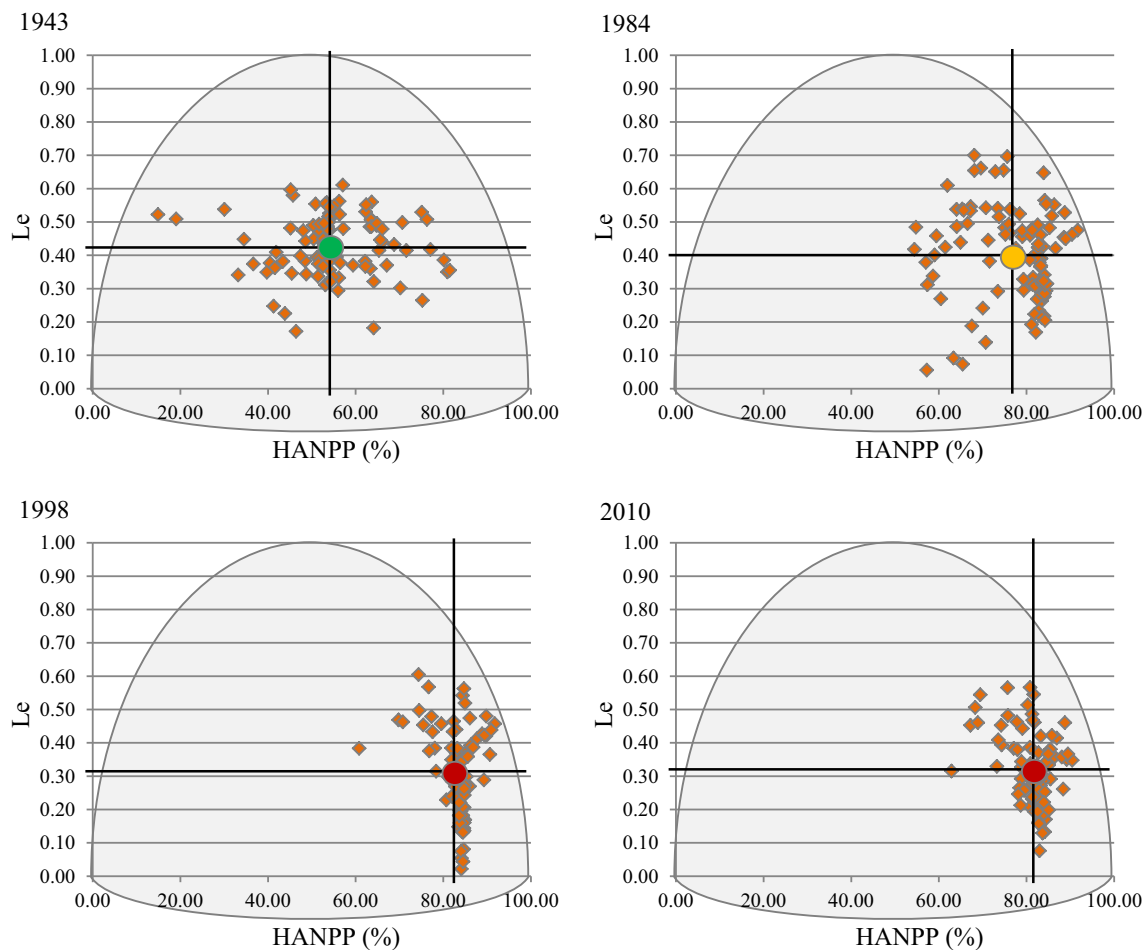
Fig. 5 Intermediate disturbance-complexity (IDC) model* (*mean values* high—*green*, medium—*orange* and low—*red*)**, landscape ecology metric (Le) and human appropriation of net primary production (HANPP), applied to the Palmira case study (1943–2010). *Notes* *Possible theoretical values represented in the grey colour figure. **IDC empirical values (*coloured dots*) come from a nonlinear relationship between Le and HANPP. *Source* Our own

inequality that has led the country to a 52-year civil war, bringing issues such as land tenure and use, access to agricultural resources and food sovereignty into discussion. With more than 30% of the land under detrimental practices that affect soils and ecosystem services negatively, it is crucial that we propose a new approach to preserve the biocultural landscapes. Only with a new approach can we provide fair livelihoods for farmers, homes for biodiversity and sustainable practices for the region's economic development, in the context of a prevailing political agenda that prioritizes agribusiness for sugar and ethanol production.

Conclusions

The last century has been characterized by an unprecedented increase in both global international food production and its associated socioecological conflicts, related to increasingly

industrialized and globalized farming systems (Mayer et al. 2015). Our research is connected to the debate between the so-called land-sparing and land-sharing approaches for biological conservation (Perfecto and Vandermeer 2010). The land-sparing approach is based on intensifying agricultural production in some areas, while reducing pressure from agriculture in others, taking (marginal) land out of production and dedicating it to nature conservation (Matson and Vitousek 2006). The land-sharing approach claims that a set of isolated natural reserves is not sufficient for biodiversity conservation, and promotes wildlife-friendly farming that can provide a complex land-matrix to connect habitats and maintain high species richness at landscape level (Tscharntke et al. 2012). More research is needed to understand how biodiversity can be maintained in human-transformed landscapes (Phalan et al. 2011), and this has been acknowledged even by those who try to combine both approaches to conservation.

We propose an intermediate disturbance-complexity (IDC) model as a measure of the capability of the land-

matrix to host biodiversity (Marull et al. 2016a, 2016b). IDC includes both land-cover metrics (landscape patterns and processes) and the energy flow available for trophic levels. Our results prove that land-use changes, which occurred in the Cauca river valley case study (Palmira municipality) between 1943 and 2010, have caused profound transformations to the structural and functional properties of the landscape. In 1943, Palmira was mainly a mosaic of pasture, agriculture and natural spaces, with medium levels of human appropriation of photosynthetic capacity and higher levels of IDC. Today, however, the region is intensively cultivated with sugarcane, where urban sprawl has also increased and only fragments of traditional forested-pastures remain.

Although migration from Cali to the suburbs led to highway construction and the industrialization of agriculture, in 1984 the Palmira's territory still showed high levels of land-cover heterogeneity and ecological connectivity. Disorganized urban sprawl still did not show its entire effects on the landscape, and the structuring of the territory in land-use areas and wetlands preservation must have helped this situation. However, since 1984, the absence of land-use policies to maintain biocultural landscapes has led to a large decrease in ecological connectivity in the entire Palmira municipality, due to the high fragmentation of the territory produced by sugarcane production, transport infrastructures and uncontrolled urban sprawl.

The integrated method utilized in this study to assess both the landscape functional structure and the net primary production available to maintain biodiversity can reveal spatial processes, such as agricultural intensification and urban sprawl, and assess pressure on ecosystem services. The quantitative and cartographic approach adopted by the IDC model enables the communication of research to planners and policy makers. Successive iterations can be used to examine the impact of different alternative planning scenarios. Future policy for biocultural landscape protection and management in the Cauca river valley could adopt this approach to evaluate whether targets for sustainable farm systems have been met. There is an urgent need for policies that facilitate sustainable land-use planning if we want to preserve the remaining biocultural heritage and the few traditional agroecosystems of the Cauca river valley.

Our results reinforce the land-sharing approach which highlights the key role that a wildlife-friendly farming plays on biodiversity maintenance in human-transformed landscapes (Fischer et al. 2008). These results differ from other recent studies (Hulme et al. 2013) which suggest that land-sparing is a more promising strategy than land-sharing to secure crop production with minimum negative impact on biodiversity. Land-sparing is typically based on high-input farming and global crops like sugarcane, corn or wheat. Monocultures with high inputs of fertilizers and agrochemicals usually attain very high agriculture yields, and it is argued that this helps to reduce the need to convert natural ecosystems into farmland to meet production targets (Grau et al. 2013). This is not the case in the Cauca river valley. Our study gives weight to the idea that sustainable rural development could be achieved with proper new management of traditional knowledge and practices.

The benefits of transferring the concept of intermediate disturbance to the land-cover and land-use change (LCLUC), by using the human appropriation of net primary production (HANPP) and the landscape ecology metric (heterogeneity and connectivity) as variables, have been shown by the IDC model. If this energy–landscape integrated analysis proves to be consistent and fruitful, it may offer a very useful tool for making robust assessments of the impact of land management on ecological landscape functioning, and it could help to solve the global food-biodiversity dilemma (Cardinale et al. 2012).

References

Agnoletti M, Emanueli F (2016) Biocultural diversity in Europe. Springer International Publishing, Cham. ISBN 978-3-319-26315-1

Alam M, Olivier A, Paquette A, Dupras J, Revéret JP, Messier C (2014) A general framework for the quantification and valuation of ecosystem services of treebased intercropping systems. Agrofor Syst 88:679–691. doi:10.1007/s10457-014-9681-x

Barnes B, Sidhu HS, Roxburgh SH (2006) A model integrating patch dynamics, competing species and the intermediate disturbance hypothesis. Ecol Model 194:414–420. doi:10.1016/j.ecolmodel.2005.10.040

Bendeck J (2013) Biocombustibles hoy. Boletín 101. http://www.fedebiocombustibles.com/v3/nota-web-id-1639.htm#editorial

Bengtsson J, Angelstam P, Elmqvist T, Emanuelsson U, Folke C, Ihse M, Moberg F, Nyström M (2003) Reserves, resilience and dynamic landscapes. Ambio 32(6):389–396. doi:10.1579/0044-7447-32.6.389

Benton TG, Vickery JA, Wilson JD (2003) Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol Evol 18:182–188. doi:10.1016/S0169-5347(03)00011-9

Berrío JC, Hooghiemstra H, Marchant R, Rangel O (2002) Late-glacial and Holocene history of the dry forest area in the south Colombian Cauca Valley. Quat Sci 17(7, 2):667–682. doi:10.1002/jqs.701

Calow P (1987) Evolutionary physiological ecology. Cambridge University Press, Cambridge. ISBN 978-0521101653

Cárdenas Carmona G (2013) Comparación de la composición y estructura de la avifauna en diferentes sistemas de producción Universidad del Valle. Cali, Colombia. [electronic resource] (doctoral dissertation)

Cardinale BJ, Duffy JE, Gonzalez A, Hooper DU, Perrings Ch, Venail P, Narwani A, Mace G, Tilman D, Wardle DA, Kinzig AP, Daily

GC, Loreau M, Grace JB, Larigauderie A, Srivastava DS, Naeem S (2012) Biodiversity loss and its impact on humanity. Nature 486:59–67. doi:10.1038/nature11148

Cassman KG (1999) Ecological intensification of cereal production systems: yield potential, soil quality, and precision agriculture. Proc Natl Acad Sci USA 96:5952–5959. doi:10.1073/pnas.96.11.5952

Chase TN, Pielke RA Sr, Kittel TGF, Nemani RR, Running SW (2000) Simulated impacts of historical land cover changes on global climate in northern winter. Clim Dyn 16(2–3):93–105. doi:10.1007/s003820050007

Chazdon RL, Harvey CA, Komar O, Griffith DM, Ferguson BG, Martínez-Ramos M, Philpott SM (2009) Beyond reserves: a research agenda for conserving biodiversity in human-modified tropical landscapes. Biotropica 41(2):142–153. doi:10.1111/j.1744-7429.2008.00471.x

Chesson P, Huntly N (1997) The roles of disturbance, mortality, and stress in the dynamics of ecological communities. Am Nat 150:519–553. doi:10.1086/286080

CVC—Corporación Autónoma Regional del Valle del Cauca, Universidad del Valle (2000) Caracterización del río Cauca. Tramo Salvajina. La Virginia, Cali

Delgadillo-Vargas O, García-Ruiz R, Forero-Álvarez J (2016) Fertilising techniques and nutrient balances in the agriculture industrialization transition: the case of sugarcane in the Cauca river valley (Colombia), 1943–2010. Agric Ecosyst Environ 218:150–162. doi:10.1016/j.agee.2015.11.003

Dupras J, Marull J, Ll Parcerisas, Coll F, Gonzalez A, Girard M, Tello E (2016) The impacts of urban sprawl on ecological connectivity in the Montreal Metropolitan Region. Environ Sci Policy 58:61–73. doi:10.1016/j.envsci.2016.01.005

Escobar A (1996) La invención del Tercer Mundo. Construcción y deconstrucción del desarrollo. Editorial Norma S.A., Bogotá. ISBN 978-980-396-776-5

FAO—Food And Agriculture Organization of the United Nations (2015). FAOSTAT. Retrieved September 10, 2015, from http://faostat.fao.org/

Fischer J, Lindenmayer DB (2006) Beyond fragmentation: the continuum model for fauna research and conservation in human-modified landscapes. Oikos 112(2):473–480. doi:10.1111/j.0030-1299.2006.14148.x

Fischer J, Brosi B, Daily GC, Ehrlich PR, Goldman R, Goldstein J, Lindenmayer DB, Manning AD, Mooney HA, Pejchar L, Ranganathan J, Tallis H (2008) Should agricultural policies encourage land sparing or wildlife-friendly farming? Front Ecol Environ 6:380–385. doi:10.1890/070019

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, Helkowski JH (2005) Global consequences of land use. Science 309(5734):570–574. doi:10.1126/science.1111772

Fox WF (2013) The intermediate disturbance hypothesis should be abandoned. Trends Ecol Evol 28(2):86–92. doi:10.1016/j.tree.2012.08.014

Franklin JF, Lindenmayer DB (2009) Importance of matrix habitats in maintaining biological diversity. Proc Natl Acad Sci 106(2):349–350. doi:10.1073/pnas.0812016105

Godfray HCJ, Beddington JR, Crute IR, Haddad L, Lawrence D, Muir JF, Pretty J, Robinson S, Thomas SM, Toulmin C (2010) Food security: the challenge of feeding 9 billion people. Science 327(5967):812–818. doi:10.1126/science.1185383

González de Molina M, Toledo VM (2014) The social metabolism: a socio-ecological theory of historical change. Springer, New York. ISBN 978-3-319-06358-4

Grau HR, Kuemmerle T, Macchi L (2013) Beyond land sparing vs. land sharing: environmental heterogeneity, globalization and the balance between agriculture and nature conservation. Curr Opin Environ Sustain 5:477–483. doi:10.1016/j.cosust.2013.06.001

Green RE, Cornell SJ, Scharlemann JPW, Balmford A (2005) Farming and the fate of wild nature. Science 307:550–551. doi:10.1126/science.1106049

Guardiola J (1995) Avances tecnológicos entre 1950 y 1980. In: Cassalett C, Torres J, Isaacs C (eds) El cultivo de la caña en la zona azucarera de Colombia. Cenicaña, Cali, pp 9–21

Guzmán G, Aguilera E, Soto D, Cid A, Infante J, García-Ruiz R, Herrera A, Villa I, González de Molina M (2014) Methodology and conversion factors to estimate the net primary productivity of historical and contemporary agroecosystems. Working Paper of the Sociedad Española de Historia Agraria, DT-SEHA n. 1407.   http://repositori.uji.es/xmlui/bitstream/handle/10234/91670/DT-SEHA%201407.pdf?sequence=3

Haberl H, Erb KH, Krausmann F, Gaibe V, Bondeau A, Plutzar C, Gingrich S, Lucht W, Fischer-Kowalski M (2007a) Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. Proc Natl Acad Sci USA 104(34):12942–12947. doi:10.1073/pnas.0704243104

Haberl H, Erb KH, Plutzar C, Fischer-Kowalski M, Krausmann F (2007b) Human appropriation of net primary production (HANPP) as indicator for pressures on biodiversity. In: Hak T, Moldan B, Dahl AL (eds) Sustainability indicators. A scientific assessment. SCOPE, Island Press, Washington, London, pp 271–288

Haberl H, Erb KH, Krausmann F (2014) Human appropriation of net primary production: patterns, trends, and planetary boundaries. Annu Rev Environ Resour 39:363–391. doi:10.1146/annurev-environ-121912-094620

Harper KA, MacDonald SE, Burton PhJ, Chen J, Brosofske KD, Saunders SC, Euskirchen ES, Robert D, Jaiteh MS, Esseen PA (2005) Edge influence on forest structure and composition in fragmented landscapes. Conserv Biol 19:768–782. doi:10.1111/j.1523-1739.2005.00045.x

Hulme MF, Vickery JA, Green RE, Phalan B, Chamberlain DE, Pomeroy DE, Nalwanga D, Mushabe D, Katebaka R, Bolwig S, Atkinson PW (2013) Conserving the birds of Uganda's banana-coffee arc: land sparing and land sharing compared. PLoS ONE. doi:10.1371/journal.pone.0054597

Huston MA (2014) Disturbance, productivity, and species diversity: empiricism vs. logic in ecological theory. Ecology 95(9):2382–2396. doi:10.1890/13-1397.1

IAvH—Instituto de Investigación de Recursos Biológicos Alexander Von Humboldt (1998) Informe Nacional sobre el estado de la biodiversidad 1997-Colombia. Bosque seco tropical, In: Chávez ME, Arango N (eds) Tomo I: Diversidad, pp 56–71

IGAC—Instituto Geografico Agustin Codeazzi (2012) Atlas de la Distribución de la Propiedad Rural en Colombia Imprenta Nacional de Colombia, pp 406–417

ISO-International Sugar Organization (2012) Outlook of sugar and ethanol production in Brazil. MECAS 12:05

Jaeger J (2000) Landscape division, splitting index, and effective mesh size: new measures of landscape fragmentation. Landsc Ecol 15(2):115–130. doi:10.1023/A:1008129329289

Krausmann F, Haberl H, Erb KH, Wiesinger M, Gaube V, Gingrich S (2009) What determines geographical patterns of the global human appropriation of net primary production? J Land Use Sci 4:15–34. doi:10.1080/17474230802645568

Krausmann F, Erb KH, Gingrich S, Haberl H, Bondeau A, Gaube V, Lauk C, Plutzar C, Searchinger T (2013) Global human appropriation of net primary production doubled in the 20th century. Proc Natl Acad Sci 110:10324–10329. doi:10.1073/pnas.1211349110

Lambin EF, Geist HJ, Lepers E (2003) Dynamics of land-use and land-cover change in tropical regions. Annu Rev Environ Resour 28(1):205–241. doi:10.1146/annurev.energy.28.050302.105459

Lindenmayer DB, Fischer J (2007) Tackling the habitat fragmentation panchreston. TREE 22:127–132. doi:10.1016/j.tree.2006.11.006

Loreau M, Mouquet N, Gonzalez A (2010) Biodiversity as spatial insurance in heterogeneous landscapes. P Natl Acad Sci 100(22):12765–12770. doi:10.1073/pnas.2235465100

Lorek T (2013) Imagining the Midwest in Latin America: US Advisors and the Envisioning of an Agricultural Middle Class in Colombia's Cauca Valley, 1943–1946. Historian 75(2):283–305. doi:10.1111/hisn.12008

Marull J, Font C (2017) The energy-landscape integrated analysis (ELIA) of agroecosystems. In: Fraňková E, Haas W, Singh SJ (eds) Sociometabolic perspectives on sustainability of local food systems. New insights for science, policy and practice. Springer, Dordrecht

Marull J, Mallarach JM (2005) A GIS methodology for assessing ecological connectivity: application to the Barcelona Metropolitan Area. Landsc Urban Plan 71:243–262. doi:10.1016/j.landurbplan.2004.03.007

Marull J, Tello E, Fullana N, Murray I, Jover G, Font C, Coll F, Domene E, Leoni V, Decolli T (2015) Long-term bio-cultural heritage: exploring the intermediate disturbance hypothesis in agro-ecological landscapes (Mallorca, c. 1850–2012). Biodivers Conserv 24(13):3217–3251. doi:10.1007/s10531-015-0955-z

Marull J, Font C, Tello E, Fullana N, Domene E (2016a) Towards an energy-landscape integrated analysis? Exploring the links between socio-metabolic disturbance and landscape ecology performance (Mallorca, Spain, 1956–2011). Landsc Ecol 31:317–336. doi:10.1007/s10980-015-0245-x

Marull J, Font C, Padró R, Tello E, Panazzolo A (2016b) Energy-landscape integrated analysis of agro-ecosystems: how the complexity of energy flows shapes landscape patterns (Barcelona province, 1860–2000). Ecol Ind 66:30–46. doi:10.1016/j.ecolind.2016.01.015

Matson PA, Vitousek PM (2006) Agricultural intensification: will land spared from farming be land spared for nature? Conserv Biol 20(3):709–710. doi:10.1111/j.1523-1739.2006.00442.x

Matthews R, Selman P (2006) Landscape as a focus for integrating human and environmental processes. J Agr Econ 57:199–212. doi:10.1111/j.1477-9552.2006.00047.x

Mayer A, Schaffartzik A, Haas W, Sepulveda AR (2015) Patterns of global biomass trade: the implications for food sovereignty and socio-environmental conflict. EJOLT Report No. 20, http://www.ejolt.org/2015/03/patterns-global-biomass-trade/

Parrotta JA, Trosper RL (2012) Traditional forest-related knowledge: sustaining communities, ecosystems and biocultural diversity. World For; 12:1–621. ISBN 978-94-007-2144-9

Patiño VM (1977) Aspectos históricos sobre los recursos naturales y las plantas útiles en Colombia. Instituto Colombiano de Cultura, Bogotá

Perfecto I, Vandermeer J (2010) The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proc Natl Acad Sci 107(13):5786–5791. doi:10.1073/pnas.0905455107

Phalan B, Onial M, Balmford A, Green RE (2011) Reconciling food production and biodiversity conservation: land sharing and land sparing compared. Science 333:1289–1291. doi:10.1126/science.1208742

Piñeiro M, Fiorentino R, Trigo E, Balcázar A, Martínez A (1982) Articulación social y cambio técnico. La producción de azúcar en Colombia. Serie Investigación y Desarrollo (Instituto Interamericano de Cooperación para la Agricultura.). Centro Interamericano de Información y documentación Agrícola, CIDIA. San José, Costa Rica

Pino J, Marull J (2012) Ecological networks: are they enough for connectivity conservation? A case study in the Barcelona Metropolitan Region (NE Spain). Land Use Policy 29:684–690. doi:10.1016/j.landusepol.2011.11.004

Pungetti G (2013) Biocultural diversity for sustainable ecological, cultural and sacred landscapes: the biocultural landscape approach. In: Fu B, Jones B (eds) Landscape ecology for sustainable environment and culture. Springer, Dordrecht, pp 55–76. doi:10.1007/978-94-007-6530-6_4

Ramírez M, Armbrecht I, Enríquez L, Lucía ML (2004) Importancia del manejo agrícola para la biodiversidad: caso de las hormigas en caña de azúcar. Rev Colomb Entomol; 30(1):115–123. ISSN: 0120-0488

Rivera C, Naranjo L, Duque A (2007) De María a un Mar de Caña. Imaginarios de naturaleza en la transformación del paisaje vallecaucano entre, 1950 y 1970. Universidad Autónoma de Occidente, Cali. ISBN 978-958-8122-53-3

Sala OE, Chapin FS 3rd, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Poff NL, Sykes MT, Walker BH, Walker M, Wall DH (2000) Global biodiversity scenarios for the year 2100. Science 287(5459):1770–1774. doi:10.1126/science.287.5459.1770

Shreeve TG, Dennis RLH, Van Dick H (2004) Resources, habitats and metapopulations—whither reality? Oikos 106:404–408. doi:10.1111/j.0030-1299.2004.13516.x

Swift MJ, Izac AMN, van Noordwijk M (2004) Biodiversity and ecosystem services in agricultural landscapes—are we asking the right questions? Agr Ecosyst Environ 104(1):113–134. doi:10.1016/j.agee.2004.01.013

Thomson AM, Calvin KV, Chini LP, Hurtt G, Edmonds JA, Bond-Lamberty B, Frolking S, Wise MA, Janetos AC (2010) Climate mitigation and the future of tropical landscapes. Proc Natl Acad Sci 107(46):19633–19638. doi:10.1073/pnas.0910467107

Tilman D, Cassman KG, Matson PA, Naylor R, Polasky S (2002) Agricultural sustainability and intensive production practices. Nature 418(6898):671–677. doi:10.1038/nature01014

Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Carsten Thies C (2005) Landscape perspectives on agricultural intensification and biodiversity-ecosystem service management. Ecol Lett 8:857–874. doi:10.1111/j.1461-0248.2005.00782.x

Tscharntke T, Clough Y, Wanger TC, Jackson L, Motzke I, Perfecto I, Vandermeer J, Whitbread A (2012) Global food security, biodiversity conservation and the future of agricultural intensification. Biol Conserv 151:53–59. doi:10.1016/j.biocon.2012.01.068

Van der Maarel E (1993) Some remarks on disturbance and its relations to diversity and stability. J Veg Sci 4:733–736. doi:10.2307/3235608

Vandermeer J, Perfecto I (2007) The agricultural matrix and a future paradigm for conservation. Conserv Biol 21(1):274–277. doi:10.1111/j.1523-1739.2006.00582.x

Vranken I, Baudry J, Aubinet M, Visser M, Bogaert J (2015) A review on the use of entropy in landscape ecology: heterogeneity, unpredictability, scale dependence and their links with thermodynamics. Landsc Ecol 30:51–65. doi:10.1007/s10980-014-0105-0

Wilson JB (1990) Mechanisms of species coexistence: twelve explanations for Hutchinson's 'paradox of the plankton': evidence from New Zealand plant communities. New Zeal J Ecol 13:17–42

Wilson JB (1994) The 'intermediate disturbance hypothesis' of species coexistence is based in on patch dynamics. New Zeal J Ecol 18:176–181

Zuluaga F (1992) El valle: Historia y paisaje. In: Historia regional del Valle del Cauca. Universidad del Valle, Facultad de Humanidades, Cali, pp 11–25