RESEARCH ARTICLE

Influence of land-use/land-cover change on landscape connectivity for an endemic threatened amphibian (*Argenteohyla siemersi pederseni*, Anura: Hylidae)

Facundo Schivo · María Cruz Mateo-Sánchez · Valeria Bauni · Rubén Darío Quintana

Abstract

Context Modifications in natural landcover generally result in a loss of habitat availability for wildlife and it's persistence will depend largely on their spatial configuration and functional connections. *Argenteohyla siemersi* is a threatened and endemic amphibian whose habitat is composed of forest patches near rivers and water bodies edges.

Objectives This study aimed to analyse the accessible habitat for this species and identify key elements to maintain its ecological network in two different types of land uses: an anthropized area with extensive cattle raising and a protected area.

Methods The structural and functional characteristics of both landscapes were analyzed. The connectivity at landscape level and the contribution of each habitat patch were evaluated through simulation models with different dispersion distances in the context of the graph theory.

Results In both landscapes, nine types of landcover were identified with different compositions. Remarkable differences were found in habitat connectivity for this amphibian species between both landscapes. As the percentage of dispersion distance increases, reachable habitat increases as well, although with higher percentages in the protected area. Two corridors were identified in the protected landscape and one in the rangeland one; patches and key links constituted all of them.

Conclusions The present work provides spatially explicit results with a quantitative basis. It could be useful as a tool for the development of management plans aimed at guaranteeing the functionality of the ecological network for this endangered species and, therefore, contribute to its long-term conservation.

Keywords Amphibian · Argentina · Connectivity · Conservation · Habitat loss · Landscape degradation · Mburucuyá National Park · Rangelands

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Introduction

The decrease in biodiversity is one of the main environmental problems worldwide (Hanski 2005). Habitat loss and degradation due to factors such as land-use changes and infrastructure development (Forman and Alexander 1998; Sala et al. 2000) have been identified among the main processes responsible for this phenomenon (Chapin III et al. 2000).

Modifying landscape configuration can compromise its structural and functional integrity (With 1997; Begon et al. 1999). The diversity, extent, distribution and shape of the landscape elements have an influence on a variety of critical ecological processes necessary for the persistence of populations (Wiens et al. 1993; Fraterrigo et al. 2009).

Landscape fragmentation process (Begon et al. 1999) is one of the most frequent alterations because of anthropogenic activities. Its adverse effects on wildlife populations are diverse depending on the perception of the species on the landscape. For habitat specialist species, landscape fragmentation could imply a new configuration with an inappropriate composition or distribution of elements used by them for feeding, reproduction or shelter, resulting in an increase in reproductive isolation (Opdam et al. 1993; Forman 1995). From a functional approach, landcover changes associated with the fragmentation process involve landscape connectivity variations, an attribute that measures the ecological flows through the territory (Taylor et al. 1993; Bodin and Saura 2010).

Connectivity determines the effective and attainable habitat area within a territory for a given species (Saura and Pascual-Hortal 2007). Maintaining and improving landscape connectivity is today considered a key part of efforts to protect biodiversity (Crooks and Sanjayan 2006). Facilitates the capacity movement to satisfy ecological requirements, reduces populations isolation and can counteract the potentially adverse effects of fragmentation by favouring genetic flow (Taylor et al. 1993; Minor and Urban 2008).

In this context, amphibians are a sensitive group because they respond to changes in environmental conditions in both aquatic and terrestrial ecosystems (Blaustein and Wake 1990; Stebbins and Cohen 1995; Cushman 2006; Becker et al. 2007; Rustigian et al. 2007). Degradation, fragmentation, or changes in the extent and habitat connectivity within landscape can produce substantial changes in the presence,

distribution and dispersion of individuals in this group (Corn and Fogleman 1984; Fahrig et al. 1995; Joly et al. 2001, 2003; Fahrig 2007). This is the case of *Argenteohyla siemersi* (red-spotted argentina frog, Mertens 1937) that inhabits forests associated with wetlands in Argentina, Paraguay and Uruguay and constitutes a single specie within this genus with two allopatric subspecies: *A. s. siemersi* and *A. s. pederseni* (Cajade et al. 2010, 2017). These environments are currently being degraded or destroyed by productive activities, mainly extensive cattle raising and afforestation (Lavilla et al. 2004). For this reason, this species has been listed as "endangered" by the International Union for the Conservation of Nature (IUCN 2019). Considering this situation and the lack of information on this species that allows establishing actions for its conservation, the objective of the present work is to evaluate landscape structural and functional connectivity due to anthropic use for this frog in a rangeland landscape compared to the natural condition represented in a protected area. In addition, it is attempted to identify those key fragments that contribute to the preservation of this species in the studied area.

Methods

Focal species

The study focuses on the subspecies *A. s. pederseni*, endemic to northeastern Argentina with a restricted distribution to a few locations (William and Bosso 1994; Fig. 1). It presents a typical two-phase life cycle (Grosse and Nöllert 1993). The adult stage have a size snout-vent 70–75 mm length, and a body mass of 19–25 g, and inhabits in the bromeliads present in forests (Álvarez et al. 2002; Cajade et al. 2013) and the emergent vegetation at the edges of semi-permanent ponds constitutes its habitat in the reproductive period (Cajade et al. 2010). Most of published works on this species are about its systematic (William and Bosso 1994; Céspedez 2000; Cajade et al. 2013; Pineiro et al. 2019), and a few describes their reproductive ecology (Diminich and Zaracho 2008; Cajade et al. 2010), songs (Zaracho and Arieta 2008) and anti-predatory defence mechanisms (Cajade et al. 2017). Little is known about habitat requirements of this species (Cajade et al. 2013). In addition, few information
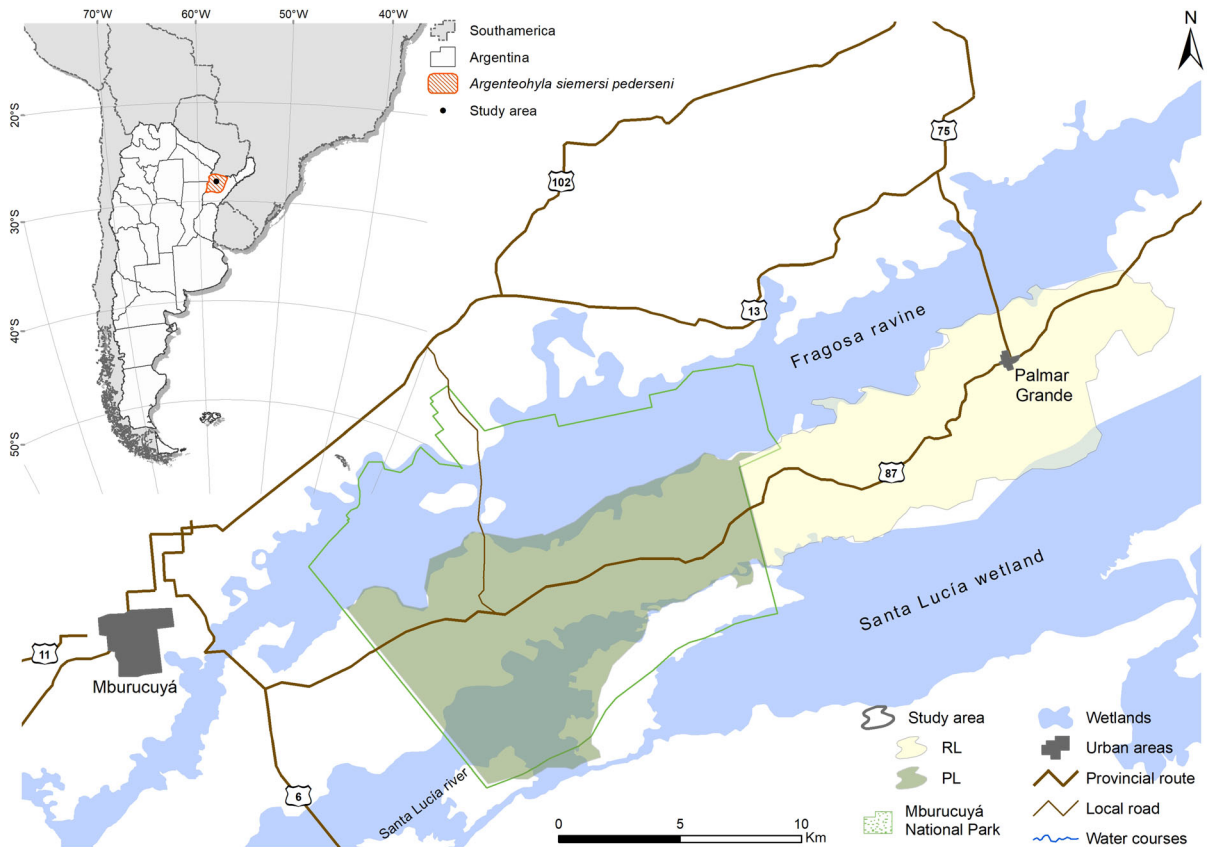
Fig. 1 Geographic distribution of *Argenteohyla siemersi pederseni* (adapted from the IUCN species distribution maps) in northeastern Argentina. Detail of the location of the study area in the Corrientes Province. It includes the protected (PL) and the rangeland (RL) landscapes

about dispersion distance of Neotropical amphibians is available (Alex Smith and Green 2005).

Study area

The study area is located in northeast of Argentina, on a sandy belt developed by fluvial depositions of the Paraná river during the Pliocene (Brea and Zucol 2011) limited by the Fragosa ravine to the north and the Santa Lucía wetlands to the south. On these belts are located numerous circular ponds originated by wind erosion and by pseudo-karstic dissolution or dragging processes (Popolizio 1996; Contreras 2011) along with hydrophilic and sub-xerophilous forests that reach a 21% of the surface coverage. Herbaceous are the most extensive communities, covering 79% of the area. Among them stand out grasslands of 0.5 m of average height, dominated by *Andropogon lateralis* or *Elyonurus muticus,* sometimes interspersed with palm groves of *Butia yatay* or *Copernicia australis* and freshwater or floating marshes (Saibene and Montanelli 1997; Arbo 2004).

The climate in this region is humid subtropical, Cfa under the Köppen climate classification, with mean annual temperature between 21 and 23 °C for the 1961–1990 period, low annual thermal amplitude and an average annual rainfall of 1300 mm (Meza-Torres et al. 2013).

Extensive cattle raising is one of the main activities developed on this area (Forclaz 2001). Mburucuyá National Park (27°58′-26°05′S, 57°59′-58°08′W) was established to preserve 17,086 ha of natural landscape previously to the intense productive transformation of the area. The park is crossed by the Provincial Route 87 (PR87), which connects the towns of Mburucuyá and Palmar Grande. Among the specific conservation objectives of this protected area, the protection of the *A. s. pederseni* is found (APN 2002).

For the present study we consider 9787.4 ha of Mburucuyá National Park as the protected landscape (PL) and the 8394.8 ha of a rangeland landscape (RL) that extends from the east of the PL to the town of Palmar Grande (Fig. 1).

Landcover characterization

Structural landcover configuration

The landscape structure was characterized using a classification based on a multitemporal-stack of Landsat 8 Path/Row 226/79 Tier_1 optical satellite images (visible, NIR and SWIR bands with spatial resolution of 30×30 m), acquired from the U.S. Geological Survey (USGS, https://earthexplorer.usgs.gov/). Representative images of the rainy season (2017/04/22) and the dry season (2018/01/03) were selected to improve the discrimination of coverage associated with water pulses. The classification was performed using the unsupervised algorithm Iterative Self-Organizing Data Analysis Techniques (ISODATA) with the ENVI program. Subsequently, and for operational purposes, a cluster analysis of their spectral signatures was performed in order to reduce the number of classes using the average Euclidean distance. The typology of landscape elements was defined based on field surveys data (Schivo 2015). Roads and houses were digitized from satellite images of higher spatial resolution (Google Earth, https://earth.google.com).

To characterize the structural configuration of both landscapes, eight descriptive metrics (McGarigal and Marks 1995; Table 1) were calculated using V-Late extension (Lang and Tiede 2003) for ArcGis 10.x (ESRI 2011).

Habitat types

Landcover used as habitat for this amphibian species was identified. The forest-associated class, described as permanent habitat (Álvarez et al. 2002; Cajade et al. 2013) and the edges of water bodies used as a reproductive and tadpole development habitat (Cajade et al. 2010) were defined as nodes. For this model, it was assumed that all the patches with the characteristics described by these authors constituted the potential habitat for this frog. This was assumed due to the difficulty of sampling all the patches available in a large area to corroborate the presence of species on a huge spatial scale or when they are small and cryptic as has been pointed out in other studies (Rubio et al. 2012; Schivo et al. 2015).

On the other hand, links are the elements that connect pairs of nodes and represent a possible way of direct dispersion between habitat fragments through the landscape (Saura and Rubio 2010).

Landcover friction

Each habitat type was assigned to a resistance coefficient (with values from 1 to 1000) based on habitat requirements and ecological costs (Table 2). The lowest cost was attributed to the habitat of adult individuals (forested areas; Fo) and breeding sites (water bodies edges; WBE), since they are the usual habitats of the *A. s. pederseni*. Other coverages such as open areas or low herbaceous plants cover, which increases the probability of desiccation or predation, increase the friction value. Similarly, open water habitats increase the friction value as well. The fact that this frog is not a good swimmer and that this habitat type exposes it to predators is that this environment does not good enough for its dispersion.

Table 1 Calculated metrics for each landcover class for the structural landscape characterization

[a]Values near 1 indicate simple circular perimeters and values near 2 indicate complex edge shapes

Metric		Unit	Description
Density	NumP	-	Number of patches
	%DNumP	1/100 ha	Patch density
Area	CA	Ha	Total class area
	MPS	Ha	Mean patch size
	PSSD	Ha	Patch size standard deviation
Edge	TE	Km	Total edge
	MPE	Km	Mean patch edge
Complexity	MFRACT	1/m	Mean patch fractal dimension[a]

Table 2 Habitat quality (DPH) and resistance coefficient (FRC) of each landcover type: Water bodies edge (WBE) and forest (Fo) habitat, bulrushes (Br), sedges (Sd), grassland (Gs), grazed grasslands and bare soil (GG-BS), water bodies (WB), and infrastructure (houses and roads). Values adapted from Joly et al. (2003)

Landcover	DPH	FRC
WBE	100	1
Fo	100	1
Br	75	5
Sd	50	5
Gs	25	50
GG-BS	0	50
WB	0	80
Houses	0	1000
Roads	0	1000

Finally, the higher costs were attributed to dispersal barriers such as houses and roads (Ray et al. 2002; Joly et al. 2003).

Connectivity

Landscape connectivity was evaluated based on graph theory (Urban and Keitt 2001) using Conefor 2.6 software (Saura and Torné 2009), considering as attribute the area of each node. Links were defined as the least-cost path (LCP) and were attributed with the length of the LCP. This was calculated as the minimum distance between each pair of nodes weighted by friction (Wiens 2001; Adriaensen et al. 2003) using the Linkage Mapper software (McRae and Kavanagh 2011). The least cost modelling is considered the most efficient approach applied to amphibians' dispersion analysis even more so when the dispersal mechanisms remain unknown (Joly et al. 2003; Decout et al. 2012).

Both, the global connectivity of the ecological network and the availability of habitat (PC) were estimated using a probabilistic model in which the habitat area of each node, the LCP and the dispersal capacity of the species are the inputs. For global connectivity, the variation of the percentage of reachable area (Equivalent Connected Area, ECA), was evaluated (Saura et al. 2011). Direct dispersal probability between pairs of patches (p_{ij}) was calculated from a decreasing exponential function (Eq. 1,

Saura and Pascual-Hortal 2007). This function was modelled from a known dispersion distance (d_{ij}) and its probability (e.g., the median dispersal distance has a probability of 0.5, while for the maximum dispersal distance the probability drops to 0.01; Saura and Torné 2009). As there is no information about dispersal capacity for this species, connectivity was simulated for 12 distances between 25 and 1000 m depending on the different maximum dispersion distances for other species of hylids (Alex Smith and Green 2005). This range of distances contemplates those most frequent events of short distance dispersion as colonization events of occupation of distant areas.

$$p_{ij} = e^{-k*d_{ij}} \tag{1}$$

where p_{ij} is the probability of direct dispersion between patches i and j, k is a constant that is adjusted from a known distance and probability value, and d_{ij} is the distance between patches i and j.

The importance of each element was evaluated for each distance analyzed as the variation of connectivity (dPC) if that element was removed. For this, the dPC_k value (probability of connectivity of the patch k) and its three complementary fractions were calculated: intrapatch connectivity ($dPCintra_k$), direct dispersion flow probability ($dPCflux_k$) and contribution of the patch k to the connectivity between the rest of the network ($dPCconnector_k$) (Eq. 2, Saura and Pascual-Hortal 2007; Saura and Torné 2009).

$$dPC_k = dPC_{intra\,k} + dPC_{flux\,k} + dPC_{connector\,k}$$
$$= 100 * \frac{PC - PC_{remove\,k}}{PC} \tag{2}$$

where $PC = \sum_{i=j}^{n} \sum_{i=j}^{n} a_i * a_j * p_{ij}^*$, with a_i and a_j attribute values of patch i and j, and p_{ij}^* the probability that patches i and j are connected.

$dPCintra_k$ (Eq. 3) is the available habitat provided by the patch k itself through the area it contains, regardless of its position in the landscape.

$$dPC_{intra} = a_i * a_j, \text{ when } i = j = k\,(a_k^2) \tag{3}$$

$dPCflux_k$ (Eq. 4) is the direct dispersion flow through the patch connections with the rest of the network when it is the point of origin or destination of that flow. This component depends both on the area of patch k and on its position in the landscape.

$$dPC_{flux} = a_i * a_j * p_{ij}*, \text{ when } i = k \text{ or } j = k \text{ and } i \neq j \tag{4}$$

$dPCconnector_k$ (Eq. 5) is the contribution of patch k k to the connectivity between the rest of the network, as a connector element, only if it is part of an optimal or shortest path between two other patches i and j. This component is independent of the patch area k and depends only on their topological position in the landscape network (Saura and Pascual-Hortal 2007; Saura and Torné 2009; Saura and Rubio 2010).

$$dPC_{connector} = a_i * a_j * p_{ij}*, \text{ when } i \neq k, j \neq k \tag{5}$$

Priority patches for maintaining connectivity were identified as those whose dPC value was within the 90th percentile for all dispersion distances evaluated. Similarly, the most important patches were also identified as connector elements from the *dPCconnector* fraction and the links. These were evaluated from the variation of the *dPC* if that connector is removed.

Results

Structural landscape analysis

Nine landcover types were identified: water bodies edges (WBE), forest (Fo), wetlands (bulrushes -Br- and sedges -Sd-), grassland (Gs), grazed grasslands and bare soil (SG-BS), water bodies (WB), and infrastructure (houses and roads) (Fig. 2).

The dominant coverages in the studied landscapes were wetlands (WBE, Br and Sd) and grassland (Gs and SG-BS), which is reflected in the results obtained from the metrics considered. These coverages were the most abundant, both in area and in number and density of patches. However, both landscapes showed structural differences between them. In the rangeland landscape there was a greater area covered by grazed grasslands and bare soil with respect to the protected landscape. Vegetated water bodies edges were also more abundant in this landscape. On the contrary, there was a larger total area and a mean patch size of forest in the National Park because of the management and conservation plans. Although the amount of forest fragments in the rangeland landscape was similar to the protected area, the average patch size was smaller. As for water bodies (WB) their area was similar in

both landscapes (Table 3). In addition, 38 nodes (20 Fo and 18 WBE) were identified in the protected landscape with a key function as connecting elements. Twenty-seven of them (71%) are also critical for maintaining connectivity, reinforcing their importance. Analyzing their spatial distribution along with the 55 priority links identified, two corridors were recognized. These corridors were, composed of a series of permanent and temporary habitat patches, which were linearly concatenated and related with priority nodes, which were functioning as stepping-stone corridors (Fig. 4b). On the other hand, in the rangeland landscape, 45 nodes (20 Fo and 25 WBE) were identified as key connecting elements of which seven (15.5%) were also classified as priority nodes for their contribution to landscape connectivity. In turn, 71 links showed a key role in maintaining connectivity in this landscape. In this case, a single corridor was identified (Fig. 4b).

Finally, three critical areas were identified as barriers within the protected landscape, where both corridors intersect the PR87 (Fig. 4b). These areas are critical because amphibians are exposed to road-kill risk and the lack of vegetation cover increases exposure to predators and desiccation hazard.

Functional landscape analysis

Functional differences between both landscapes were found. In them, as dispersion distance increases, the percentage of habitat achievable also increases. For each of the distances considered, this percentage was always higher in the protected landscape (Fig. 3).

For the shorter dispersion distance, the greatest contribution to connectivity was associated with habitat availability within the fragment (intra-patch connectivity, *dPCintra* component). In particular, and for the shortest distances, this contribution was always greater in the rangeland landscape (Fig. 3) along with a lower percentage of reachable habitat (Fig. 3).

As the dispersion distance increases, a decrease in the participation of *dPCintra* component and an increase in the contribution of interparches components (*dPCflux* y *dPCconnector;* Fig. 3) was observed in both landscapes. In the protected landscape, at the major dispersion distances considered, the *dPCflux* was greater than the *dPCconnector* while in the rangeland landscape this relationship was reversed
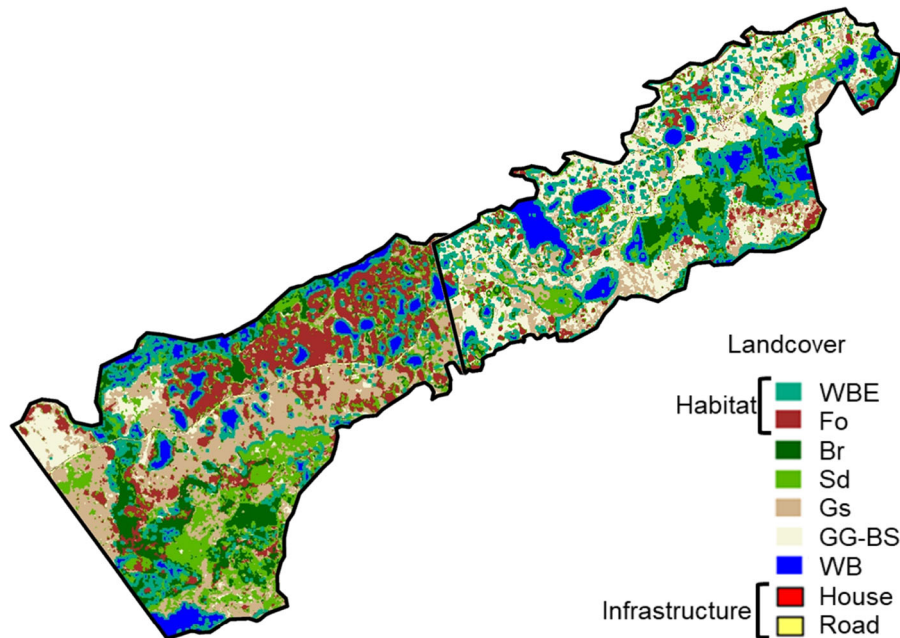
Fig. 2 Landcover map classification of a sector of the studied area, Corrientes Province, Argentina: water bodies edge (WBE) and forest (Fo) habitat, bulrushes (Br), sedges (Sd), grassland (Gs), grazed grasslands and bare soil (GG-BS), water bodies (WB), and infrastructure (houses and roads)

(Fig. 3). In the protected area, the percentage of reachable habitat was always greater.

As for the most important nodes in the contribution to the protected landscape connectivity, 23 (3%) forest fragments grouped 1743.7 ha (96.3%) and 24 (9%) fragments of water bodies edges add up to 694.3 ha (41.8%) of this habitat type. On the contrary, in the rangeland landscape, only 15 nodes from the 54 priority sites identified corresponded to forest (2% of the Fo fragments), which add up to 123.3 ha (30.6%) of the total habitat area for adults of this species. The remaining 39 nodes (1457.3 ha; 65.7%) correspond to water bodies edges and represent 92.2% of the area, with the greatest contribution to landscape connectivity within this landscape (Table 4; Fig. 4a).

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Table 3 Structural landscape characteristics of both studied landscapes

Landscape type	NumP	%DNumP (1/100 ha)	CA (ha)	(%)	MPS (ha)	PSSD (ha)	TE (km)	MPE (km)	MFRACT
Protected									
WBE	268	2.7	1661.4	17.0	6.2	20.0	457.3	1.71	1.33
Fo	718	7.3	1810.6	18.5	2.5	21.2	494.6	0.69	1.42
Br	607	6.2	1005.7	10.3	1.7	9.6	314.5	0.52	1.41
Sd	2118	21.6	1816.5	18.6	0.9	6.6	787.6	0.37	1.40
Gs	1254	12.8	2380.5	24.3	1.9	25.3	662.4	0.53	1.40
GG-BS	724	7.4	501.8	5.1	0.7	7.7	201.2	0.28	1.41
WB	161	1.6	550.2	5.6	3.4	11.0	120.9	0.75	1.37
Houses	5	0.1	0.5	0.0	0.1	0.0	0.6	0.12	1.41
Roads	9	0.1	60.3	0.6	6.7	8.8	40.6	4.52	1.48
Rangeland									
WBE	389	4.6	2217.2	26.4	5.7	17.7	581.8	1.50	1.34
Fo	668	8.0	403.3	4.8	0.6	1.8	197.1	0.30	1.40
Br	547	6.5	602.5	7.2	1.1	5.5	207.2	0.38	1.43
Sd	2116	25.2	1074.5	12.8	0.5	2.7	568.7	0.27	1.41
Gs	1715	20.4	1046.2	12.5	0.6	2.8	538.0	0.31	1.40
GG-BS	889	10.6	2362.7	28.1	2.7	15.7	713.2	0.80	1.40
WB	135	1.6	573.2	6.8	4.2	16.1	100.5	0.74	1.37
Houses	181	2.2	17.0	0.2	0.1	0.0	22.2	0.12	1.41
Roads	24	0.3	98.2	1.2	4.1	7.4	66.6	2.78	1.45

WBE water bodies edge, *Fo* forest habitat, *Br* bulrushes, *Sd* sedges, *Gs* grassland, *GG-BS* grazed grasslands and bare soil, *WB* water bodies, *Ho* houses, *Ro* roads

(Fig. 4b). On the other hand, in the rangeland landscape, 45 nodes (20 Fo and 25 WBE) were identified as key connecting elements of which seven (15.5%) were also classified as priority nodes for their contribution to landscape connectivity. In turn, 71 links showed a key role in maintaining connectivity in this landscape. In this case, a single corridor was identified (Fig. 4b).

Finally, three critical areas were identified as barriers within the protected landscape, where both corridors intersect the PR87 (Fig. 4b). These areas are critical because amphibians are exposed to road-kill risk and the lack of vegetation cover increases exposure to predators and desiccation hazard.

Discussion

The results achieved in our work showed that anthropogenic intervention configured a deeply modified landscape with respect to the natural condition. The structural and functional configuration of the rangeland landscape generated a decrease in connectivity for *Argenteohyla siemersi pederseni*. Therefore, there is a lower availability of reachable habitat, which could affect the genetic exchange, and the flow of individuals. This fact could lead to local extinction of isolated populations of this frog (Hanski and Gilpin 1991; Kindlmann and Burel 2008). In this context, the expansion or intensification of cattle raising could have a negative impact on the conservation of the species, which has, at present, a jeopardized conservation status due to habitat loss and degradation (IUCN 2019).

Landscape configuration and connectivity

As Lambin et al. (2001) suggest, the land-cover and land-use changes is one of the biggest drivers of deforestation as well as of wetland loss and degradation (Gardner et al. 2015). In this study, when
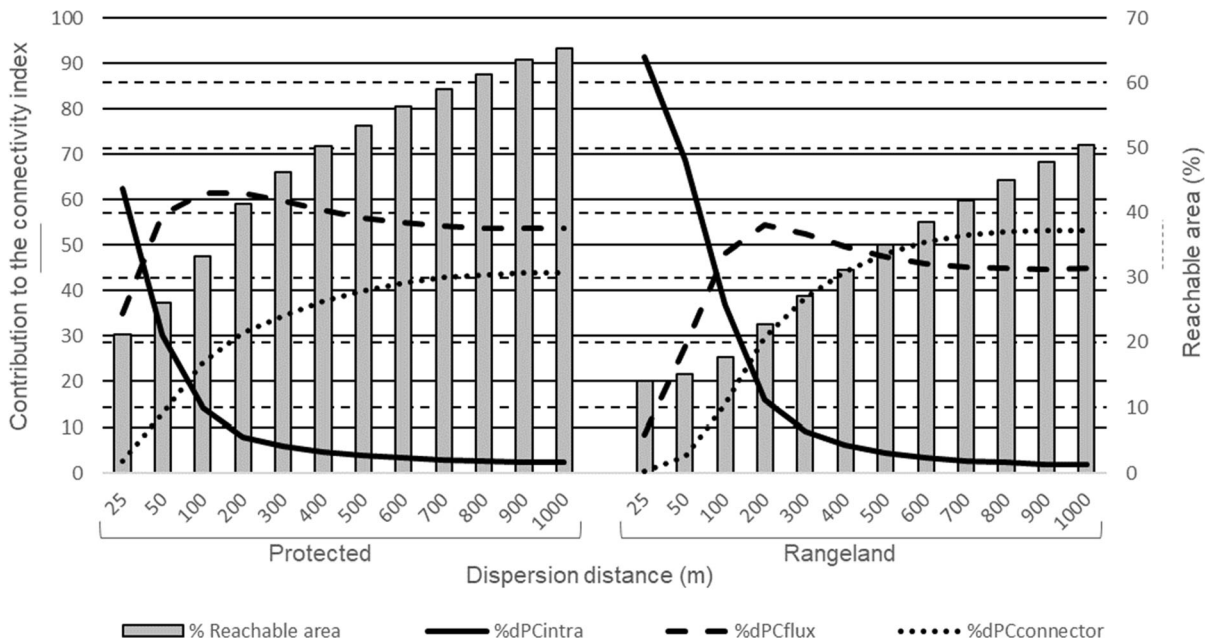
Fig. 3 Reachable area (bars) and contribution of dPCintra (continuous line), dPCflux (dashed line) and dPCconnector (dotted line) components to the connectivity index as a function of the twelve-dispersion distance considered for each landscape

Table 4 Contribution to the connectivity index dPC of those nodes included in the 90th percentile for both considered landscapes

Landscape	Landcover class	Total class		Nodes	Priority nodes	
		Area (ha)	Ratio[a] (%)		Area (ha)	Ratio[b] (%)
Protected	Fo	1810.6	58.1	17	1052.1	60.2
	BS	1661.4	41.8	14	694.3	39.8
Rangeland	Fo	403.3	10.1	3	40.8	3.3
	BS	2217.2	53.2	23	1180.5	96.7

[a] Considering the total class area for each landscape type

[b] Considering the total priority area (Fo + WBE) for each landscape type

comparing both rangeland and protected landscapes, structural differences were observed in which the decrease of the forest cover stands together with an increase of the grazed grasslands and bare soil area. The change in landscape composition had direct impacts on amphibians' population dynamics (Cushman 2006). For instance, adult habitat loss has deep implications for its dispersion because of the increase of both the risk of predation and the probability of desiccation due to having to cross open areas (Mazerolle and Desrochers 2005). This results in an increase in friction that leads to fragmentation of populations by isolation (Fisher and Lindenmayer 2007).

Therefore, this process is critical for the maintenance of endangered species populations (Kerr and Deguise 2004), as we observed when comparing the extension of reachable habitat and the contribution of both intra and interpatch connectivity fractions.

On the other hand, rangeland landscape presented a greater coverage of vegetated border of ponds, the habitat of reproduction and development of larval stages. This may be related with the nutrients supply due to resuspension of sediments because of cattle trampling (Sahuquillo et al. 2012). In addition, resuspension of sediment also increases the abundance of periphyton (Middleton 2010), which could be food

Fig. 4 **a** Nodes contribution to landscape connectivity network. *dPCintra* (**a1**), *dPCflux* (**a2**) and *dPCconnector* (**a3**) nodes contribution. **b**. Priority habitat nodes (Fo and WBE) and links in the studied landscapes. Gray boxes (**b1**) indicate a potentially critical area due to the increased risk of mortality because of the intersection between the provincial road and the identified corridors
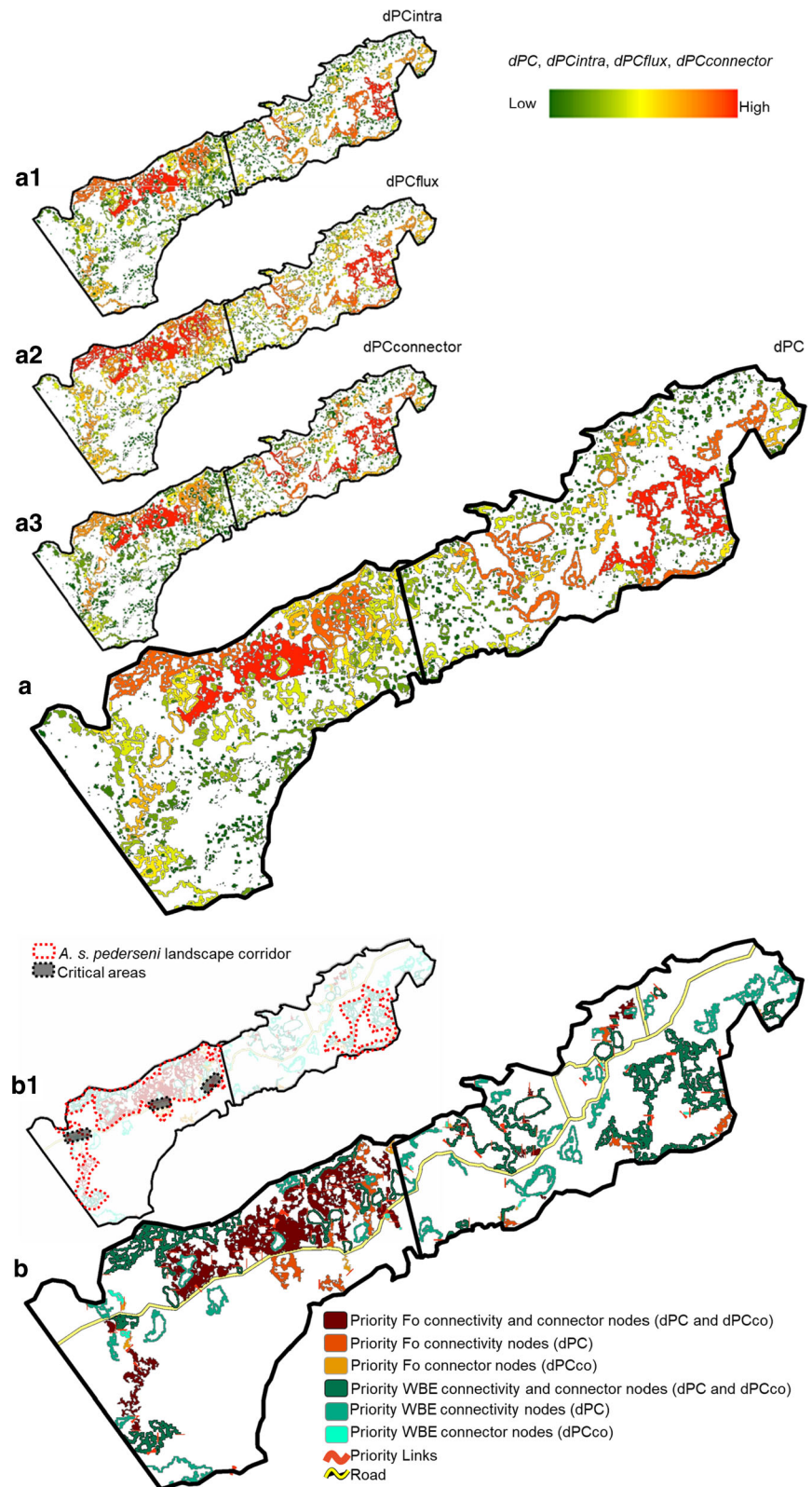
for larvae (Pollo et al. 2015). However, the increase in the area of this habitat type does not compensate in terms of connectivity the low representation of forests in this landscape. Consequently, the total habitat area in terms of intrapatch component was smaller than that found in the protected landscape.

Contribution of landscape elements to connectivity

At landscape scale, functional connectivity is much more complex than structural one (Mühlner et al. 2010) since it considers behavioural aspects such as the ability to move and disperse of each species along with spatial mosaic composition and configuration (Boitani et al. 2007). The modifications observed in the rangeland landscape due to land-cover and land-use changes results in a decrease in connectivity for each of the distances considered compared to the protected landscape. In particular, the relative contribution of each of the three components of connectivity reaches relatively constant levels after 600 m. Therefore, the perception of connectivity for both landscapes is not modified from that distance threshold for this studied species.

For the shortest dispersion distances, which are the most frequent for hylids (Alex Smith and Green 2005), the degree of fragmentation in the rangeland landscape implies a small area of connected habitat (14%). For the same distance, in the protected landscape, 21% of the total available habitat is connected. In the rangeland landscape, the greater contribution to connectivity provided by the *dPCintra* component together with a lower percentage of reachable habitat suggests greater functional fragmentation as a consequence of cattle impact. The decrease in the contribution of this component together with an increase in the flow contribution (*dPCflux*) and the landscape integral connectivity of the connector component (*dPCconnector*) due to the increase in the probable distance of dispersion, suggests an increase in the possibility of establishing new links with other patches too. This increase in the contribution to connectivity of both components does not occur in the same way in both landscapes. In particular, in the protected landscape there is a greater importance of the *dPCflux* component, so it means that individuals can reach other patches directly without the need to use intermediate stepping-stone patches. On the other hand, in the livestock landscape, the greatest contribution of the *dPCconnector* component implies that individuals need to use intermediate fragments as connecting elements (stepping-stones) to reach more distant habitat patches. For this reason, the configuration of the landscape as a consequence of extensive cattle raising causes that *A. s. Pederseni* takes different characteristics as a disperser, behaving as an intermediate mobility species in Mburcuyá National Park and as a limited dispersive capacity species within the livestock landscape as a consequence of landscape degradation (Fisher and Lindenmayer 2007; Saura and Rubio 2010).

Critical elements for connectivity
and management implications

For species conservation, particularly those threatened by habitat loss due to changes in natural coverage, it is extremely important to identify suitable habitat fragments and consider their connectivity within the landscape in order to maintain their populations for a long term. In addition, it is necessary to develop strategies to improve reachable habitat in order to favour the movement of individuals and gene flow among patches (Crooks and Sanjayan 2006). Functional analysis is the key concept that quantitatively calculates the importance of each patch habitat within each landscape. As other authors point out, the results of these analyzes allow the design and evaluation of conservation strategies (Decout et al. 2012; Clauzel et al. 2015). In this sense, the key patches for connectivity that were identified in the protected landscape resulted in both forest and water body edge habitats in similar proportions in terms of number of patches, but with a marked difference in area in favour of the forest habitat. On the other hand, rangeland landscape presented differences in the number of key fragments, as well as in the area of the two types of habitats. The importance of temporary habitats for maintaining landscape connectivity stands out. However, the role of the few key forest fragments becomes more relevant because they are habitats used by adults and they are present throughout the year. For this reason, it would be necessary to improve connectivity in rangeland landscapes implementing conservation strategies for these fragments and plans for habitat restoration in that landscape (Zemanova et al. 2017). From a perspective based on socio-ecological resilience, the restoration and conservation of degraded

environments in agro-productive landscapes should be promoted (Ghazoul and Chazdon 2017). In our work, we found that both the amount of forest fragments and their density are similar between both landscapes, although with a substantial difference with respect to the covered area. For this reason, we consider necessary restoration actions like tree planting in order to reinforce the natural regeneration and expansion processes of forest fragments (Chazdon 2003).

Finally, the analysis of connectivity together with the identification of corridors allowed the identification of three critical areas that could be functioning as dispersal barriers. In these cases, the implementation of specific wildlife crossing structures for amphibians such as sewers and tunnels that allow crossing the PR87 would reduce the risk of roadkill as well as predation or dehydration because of the impacts of crossing open areas (Woltz et al. 2008; Gurrutxaga and Saura 2013). According to Aichi strategic goal C for biodiversity (CBD 2010), the improvement of connectivity, it is necessary in order to safeguard ecosystems, species and genetic diversity. In our work, we could identify the key elements of the landscape along with possible areas to improve the efficiency of biological corridors. This type of landscape interventions ensures permeability, a conservation strategy complementary to the creation of new protected areas to ensure a well-connected system of protected areas with a surrounding rangeland (CBD 2010).

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