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# Landscape dynamics of *Paspalum quadrifarium* grasslands analyzed by Morphological Spatial Pattern Analysis (MSPA)

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**Background.** Despite its wide distribution worldwide, only 4.6% of temperate grasslands are included within systems of protected areas. In Argentina, this situation is even more alarming: only 1.05% is protected. The study area (central area of the southern Salado River basin) has a large extent of grasslands of *Paspalum quadrifarium* (Pq) which has been target since the last century of a variety of agricultural management practices including fire burning for cattle grazing.

**Methods.** Were used as base data binary images of presence-absence data of Pq coming from a 42-year (1974-2016) land cover change study performed over Landsat Imagery (MSS, TM, ETM, and OLI sensors). MSPA (Morphological Spatial Pattern Analysis) and Network Analysis were performed to the data using Gidos Toolbox for the estimation of habitat and connectivity dynamics of the Pq patches (fragments).

**Results.** Was observed a loss of area and habitat nuclei of this grassland between the beginning and the end of the study period. A drastic reduction in connectivity was also evident in resulting maps. The number of large Pq grassland fragments (> 50 ha) decreased during the study period, and fragmentation measured as number of components (patches) was higher at the end of study period. The Pq pajonal nuclei had their minimum representativeness in 2000, and recovered slightly in 2011, but with a significant percentage increase of the small patches (=islets) and linear elements as bridges and branches. Large corridors (mainly edge of roads) could be observed at the end of study period, while the total connectivity of the landscape pattern drops abruptly.

**Discussion.** The habitat reduction could have an impact on the ecosystem functioning and the mobility of some species of native fauna. The connecting elements of the landscape were maintained and/or recovered in percentage in 2011 and 2016. This fact, although favoring the dispersion of the present diversity in the habitat nuclei could cause degradation by an edge effect. On the methodological side, the use of a proved tool as Gidos Toolbox for evaluating forest fragmentation could also be useful for monitoring dynamics of a grassland-habitat fragmentation.

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3

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17 **ABSTRACT**

18 **Background.** Despite its wide distribution worldwide, only 4.6% of temperate grasslands are included  
19 within systems of protected areas. In Argentina, this situation is even more alarming: only 1.05% is  
20 protected. The study area (central area of the southern Salado River basin) has a large extent of  
21 grasslands of *Paspalum quadrifarium* (Pq) which has been target since the last century of a variety of  
22 agricultural management practices including fire burning for cattle grazing.

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25 were performed to the data using Guidos Toolbox for the estimation of habitat and connectivity  
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27 **Results.** A loss of the coverage area and habitat nuclei of this grassland was observed between the  
28 beginning and the end of the study period. A drastic reduction in connectivity was also evident in  
29 resulting maps. The number of large Pq grassland fragments (> 50 ha) decreased during the study period,  
30 and fragmentation measured as number of components (patches) was higher at the end of study period.  
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37 recovered in percentage in 2011 and 2016. This fact, although favoring the dispersion of the present  
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39 use of a proved tool as Guidos Toolbox for evaluating forest fragmentation could also be useful for  
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42 Keywords: Land use, Cattle grazing, Morphology, Connectivity, *Paspalum quadrifarium*

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## 44 INTRODUCTION

45 The loss of habitat and the fragmentation of ecosystems are one of the main threats to the  
46 conservation of biodiversity worldwide (Fahrig 2003, Hobbs and Yates 2003, Henle et al 2004,  
47 Wilson et al. 2016). In Argentina, the conversion of natural ecosystems to agricultural lands has  
48 consequences such as the loss of habitat and biodiversity, the alteration of biotic interactions and  
49 biogeochemical processes (water cycles, carbon and nutrients), the reduction of the capacity to  
50 provide ecosystem services and the transformation of the landscape (Herrera et al., 2012;  
51 Volante et al., 2012; Gandini et al. 2014). In this way, given the magnitude of anthropogenic  
52 activities on natural systems (Vitousek et al., 1997), understanding if and how biodiversity  
53 recovers from disturbances is an important focus of ecology and conservation biology. Woods et  
54 al (2016) emphasized the importance of considering spatial scale while the impacts of a  
55 disturbance on an ecosystem are quantified. Across the globe, many once- pristine natural  
56 ecosystems have been replaced by human-dominated mosaic landscapes, wherein a patchwork of  
57 human land-use patterns has been superimposed on pre-existing patterns of heterogeneity in  
58 natural environmental conditions. In such landscapes, species have experienced their  
59 environment across a range of spatial scales (Tschardt et al. 2012), so it is important to  
60 evaluate these effects across both time and space (Van Horne 2002).

61 A characteristic type of grassland landscape in the Salado river basin is the “pajonal” of  
62 *Paspalum quadrifarium* also known as *Paspaleum* (Vervoorst 1967). It represents one of the  
63 twelve plant communities identified for this area and it is a type of grassland characterized by  
64 marked abundance of *P. quadrifarium*, a grass that can grow into dense tufts reaching 1 to 1.50  
65 m (Frangi 1986), and various companion species in different proportions. *Paspaleum* is

66 characterized by its distribution in a wide range of topographies (Lara & Gandini 2013a) forming  
67 different vegetation units (Perelman et al. 2003; Lara & Gandini 2013b).

68 Pajonal grasslands have been under fire and grazing disturbance for a long time. Since the  
69 introduction of domestic livestock by European settlers and almost without interruption, this  
70 grassland has been managed changing their coverage and land use in different ways (Foley et al.  
71 2005, Vazquez et al. 2012). Mainly fire is currently used in the winter-spring period with the aim  
72 of increasing net productivity and thus livestock receptivity (Lattera 2003). In this way, the  
73 interaction of fire with cattle grazing lead to deep changes that can be seen across scales of  
74 analysis (Herrera et al 2009; Lara & Gandini 2011).

75 Ecologists distinguish between a particular disturbance event -like an individual storm or fire-  
76 and the disturbance regime that characterizes a landscape (e.g., White and Jentsch 2001).

77 Disturbance is a “hot topic” in land and resource management and particularly in grassland  
78 management, because many disturbance regimes seem to be changing due to human activities,  
79 especially climate change. For example, the risk of large fires is increasing in many areas of the  
80 world. The disturbance regime refers to the spatial and temporal dynamics over a longer time  
81 period and is described by characteristics such as the spatial distribution of disturbances;  
82 disturbance frequency, return interval, and rotation period; and disturbance size, intensity, and  
83 severity (Turner & Gardner 2015).

84 Despite its large distribution worldwide, only 4.6% of temperate grasslands are included within  
85 national protected area systems. In Argentina, this situation is even more alarming since only  
86 1.05% is protected (Bilenca and Miñarro 2004). The underestimation of the productive value of  
87 these natural grasslands starts from the difficulty of objectively visualizing the goods and  
88 ecosystem services that they provide.

89 Human encroachment on the environment through resource extraction and urban expansion have  
90 led to fragmentation (Maguire et al. 2016), with consequences for biodiversity (Chapin et al.  
91 2000), ecosystem processes (Díaz & Cabido 2001, Harrington et al. 2010) and the ecosystem  
92 services that they are supporting (Mitchell et al, 2014).

93 The current increase in agricultural and livestock pressure in the region (Cañibano et al., 2004;  
94 Vázquez et al., 2012) pre-supposes a little encouraging scenario due to the replacement of natural  
95 pasture coverage. However, Paspaleto (ecological community of Paja Colorada – *Paspalum*  
96 *quadrifarium*) remnant patches persist in the centre of Buenos Aires province (Herrera et al.,  
97 2009). These patches were maintained as pasture sites in good state of conservation according to  
98 studies of Fundación Vida Silvestre Argentina (Bilenca and Miñarro 2004). So, these sites were  
99 classified as Valuable Grassland Areas (PVAs), given their importance as a source of great  
100 native animal diversity and the numerous ecosystem services they provide.

101 Fahrig (2003) considered fragmentation as one of the most damaging threats to biodiversity  
102 conservation in recent times because the population viability in fragmented landscapes depends  
103 to a large extent on the structural and functional integrity of the landscape. In this context, it is  
104 necessary to carry out studies that allow analysing the trends of change in these habitat fragments  
105 and their connections to implement management and conservation strategies in areas of high  
106 regional ecological importance.

107 The sustainable management of fragmented landscapes will depend on understanding the spatial  
108 ecology of the ecosystem services needed over the long-term (Maguire et al. 2016). In terms of  
109 functional integrity of ecosystems, landscape connectivity is considered one of the key properties  
110 to maintain biodiversity. Landscape connectivity is defined as the degree to which landscape  
111 facilitates the movement of species and other ecological flows (Taylor et al., 1993). It is

112 considered a key aspect to take in account for biodiversity conservation efforts around the world  
113 and one of the best responses to counteract the negative effects of habitat fragmentation and to  
114 facilitate species adaptation to changes in their natural habitats (Crooks and Sanjayan 2006).  
115 Morphological Spatial Pattern Analysis (MSPA) has been promoted in the last decade by the  
116 Joint Research Center of the European Commission (JRC) to contribute to the knowledge and  
117 exchange of information on issues related to ecosystem patterns of disturbance in human-  
118 managed ecosystems, assessing fragmentation and connectivity in Europe and in the world.  
119 MSPA is described by Vogt et al. (2007a) as "a customized sequence of mathematical  
120 morphological operators to describe the geometry and connectivity of the components of an  
121 image."  
122 MSPA approach uses a binary method of image classification based on the geometry and forms  
123 of the elements to classify the patterns into seven categories: core, islet, loop, bridge, perforation,  
124 edge, and branch (Soille & Vogt, 2008). MSPA approach has been applied in landscape ecology  
125 to identify and map the structural patterns of forests at pixel level, allowing identification of  
126 fragmentation issues (Vogt et al., 2007a) and the connective elements of a landscape as the  
127 corridors (Vogt et al., 2007b). MSPA also has been implemented to analyse the connectivity of  
128 forests in Europe through the identification of key structural elements that play the role of  
129 connectors and the integration of connectivity indexes based on habitat availability (Saura et al.,  
130 2011). Other structural elements such as riparian corridors have been identified through the  
131 MSPA approach to study their contribution to structural connectivity and to establish  
132 conservation valuation criteria (Clerici and Vogt, 2013). Likewise, the US green infrastructure  
133 has been morphologically classified and mapped to know its distribution with a view to forest  
134 protection and correct decision-making in landscape planning (Wickham et al., 2010). This



135 approach was used to identify and classify the morphological types of fragmentation, based on  
136 the availability of habitat and to recognize the temporal variation between the elements that  
137 contribute to the maintenance of landscape connectivity.  
138 Improved landscape connectivity is increasingly considered a viable management strategy to  
139 maintain biodiversity, ecosystem functions, and services (Ziter et al. 2013). In a part of the study  
140 area -the Salado River basin- the habitat fragmentation pattern has been reported to have  
141 increased considerably in the last 40 years (Lara & Gandini 2014), Thus becoming one of the  
142 major environmental problems in the basin. In this way, this research presents a novel  
143 application of the MSPA approach in the monitoring of change over time in the fragments of  
144 grassland, the *Paspalum quadrifarium* habitats.  
145 In this work the MSPA was applied to a remote sensing classification of Landsat images in order  
146 to analyze the 40-year temporal change in the habitat fragments of the “pajonal” grasslands  
147 (*Paspalum quadrifarium*).

## 148 **MATERIALS & METHODS**

### 149 Study Area

150 The study was carried out in the Flooding Pampa and Inland Pampa areas of the Salado River  
151 basin, in the Buenos Aires province (Argentina), covering mainly two different agroecological  
152 zones, the “Flooding and Inland Pampas” (Figure 1), which concentrates the greatest  
153 representativeness of the ecological community of interest (Gandini et al., 2014).

### 154 Data acquisition, pre-processing and land cover classification

155 A series of five Landsat images (path 225, row 85) for the years 1974 (MSS sensor), 1988 (TM  
156 sensor), 2000 (ETM + sensor), 2011 (TM sensor) and 2016 (OLI sensor) was used. The digital  
157 numbers (DN) were converted to reflectance (except the thermal band) according to Chander et

158 al. (2009), and the reflectance values were then adjusted for atmospheric scattering using the  
159 Improved Dark Object Subtraction method by Chavez (1996).

160 In accordance with previous reports (Herrera et al., 2009, Lara and Gandini, 2013b), the initial  
161 land cover types used were: pajonal, short-grass matrix, pastures, crops and water bodies. To  
162 identify the land cover types, supervised classifications were employed using the maximum  
163 likelihood algorithm (Lu and Weng 2007). The classifications were performed using all  
164 reflective Landsat bands; for 1974: MSS4, MSS5, MSS6 and MSS7; for 1988 and 2011: TM1,  
165 TM2, TM3, TM4, TM5 and TM7; for 2000: ETM + 1, ETM + 2, ETM + 3, ETM + 4, ETM + 5  
166 and ETM + 7, and for 2016, Bands 2, 3, 4, 5, 6 of OLI Sensor. The thermal bands of platforms  
167 were discarded.

168 For 1988, the training sites were located by visual interpretation on 44 aerial infra-red  
169 photographs (scale 1: 20,000) taken in 1988 summer, following criteria such as texture, shape  
170 and colour (Chuvienco,2010). For 2011 and 2016, the control points were selected using a global  
171 positioning system (GPS) in the field within relatively homogeneous areas. For 1974 and 2000,  
172 the training sites were selected by visual analysis following the medium spectral signature for  
173 each land cover type and using areas with similar spectral characteristics –over land cover  
174 remained unchanged- (Chuvienco, 2010; Schulz et al., 2010).

175 Classification results were filtered using a 7 x 7 median filter to remove isolated pixels. Later,  
176 the MSS classification (1974) was re-sampled to a 30 x 30 m pixel size to allow multi-temporal  
177 comparison with the rest of the series.

178 The accuracy of the classification maps was assessed with the use of quantity disagreement and  
179 allocation disagreement (Pontius and Millones, 2011). These indexes are more useful and  
180 simpler than standard Kappa (Congalton, 1991) and allow us to focus on two components of

181 disagreement between maps and reference points in terms of the quantity and spatial allocation  
182 of the land cover types.

183 Pajonal fragments were identified based on these classifications, and binary Pajonal presence-  
184 absence maps were created. These binary maps of 1974, 1988, 2000, 2011 and 2016 were  
185 analyzed using a morphological classification (MSPA) with the software Guidos Toolbox (Soille  
186 and Vogt, 2008; Vogt, 2014).

187 A comparative analysis of landscape connectivity was carried out, evaluating the variations in  
188 the size and connectivity of the patches of pajonal. The analysis was performed considering each  
189 set of connected patches as a single landscape element. In this way the size of the elements  
190 varied as the patches were fragmented and disconnected by the effect of the livestock  
191 management (fire and grazing disturbance).

192 In addition ECA was calculated. ECA is defined as the size of a single habitat patch (maximally  
193 connected) that would provide the same value of the probability of connectivity than the actual  
194 habitat pattern in the landscape. It is calculated as the square root of the numerator of the PC  
195 index (Saura et al. 2011).

196

## 197 **RESULTS AND DISCUSSION**

198 A complete set of resulting images of Classification, MSPA and connectivity for a set of 5  
199 analyzed years is stored in supplementary material that could be opened by free software as  
200 QGIS (2018) and similar, as well as Guidos Toolbox (Vogt 2014) .txt outputs for seeing running  
201 details. For reasons of simplicity of interpretation only maps of extreme years in time (1974-  
202 2016) are shown in the main body of text.

### 203 *Image classification of land cover*

204 Classification error issues are shown in Table 1. Good thematic accuracies for Pq were obtained,  
205 and the separation of overall disagreement into two components was used to learn about sources  
206 of error and give guidance on how to improve each classification (Pontius and Millones, 2011).  
207 Quantity disagreements, and not allocation disagreement, were particularly taken into  
208 consideration to assess classification quality since the main aim of this work was focused on  
209 regional changes and not pixel-to-pixel changes (Keller and Smith 2014)

### 210 *Fragmentation and habitat loss*

211 The observation of the MSPA image product (Fig. 2) indicates a loss of area of grassland as a  
212 whole, and of the habitat nuclei between the beginning and the end of the study period. A drastic  
213 reduction in connectivity is also evident. The number of large pajonal fragments (> 50 ha)  
214 decreased during the study period, but the greatest fall was observed in 2000, although in 2011  
215 this trend continued at a lower rate (Table 1).

216 Fragmentation measured as number of components (patches) resulting from habitat pressure is  
217 higher in 2011 and 2016 (Table 1). This observation agrees with MSPA data, in which the  
218 pajonal nuclei had their minimum representativeness in 2000, and recovered by 2011, with a  
219 significant percentage increase in the small patches (=islets). While edges and curls remained  
220 relatively stable, the linear elements (bridges and branches) increased their representativeness  
221 (Table 1, MSPA; see also Figure 2). This fact, although favoring the dispersion of the present  
222 diversity in the habitat nuclei could cause degradation by an edge effect. The connecting  
223 elements of the landscape were maintained and/or recovered in percentage. Larger corridors  
224 (mainly edge of roads) could be observed while the total connectivity of the landscape pattern  
225 drops abruptly (Table 1, connectivity and Figure 2) in terms of mean and median connectivity  
226 after 1988 and %ECA after 2011.

227 The effect on diversity could be more important when considering that habitat patches form a  
228 "network" and that they are not connected to each other (Maguire et al. 2016). This loss of  
229 connectivity is clearly evidenced in Figure 3.

230 Temporal variation of ECA% shows the dynamics of the fragmentation process in the study  
231 area: from a value of 28, doubling the patch size in 1988 is necessary to reach similar  
232 connectivity (median values). In successive years, the connectivity and ECA decline to a  
233 minimum in 2011, showing a slight recovery in 2016.

234 Within the frame of environmental conservation issues, monitoring these environments and  
235 conducting further research on their impact on biodiversity are deemed necessary. In this work,  
236 as Wickham et al. (2010) this approach was used to identify and classify the morphological types  
237 of fragmentation, focusing on the dynamics of habitat availability, and to recognize the variation  
238 in the time of the connective elements of the landscape.

239 In this research it is noticeable a high and sustained habitat decline in the period 1974-2011 and a  
240 slight recovery (in some indicators) in 2016. Monitoring the dynamics of these grasslands is  
241 necessary to contribute to their conservation.

242 This research opens up the need to think about a short-term research on the minimum size of  
243 pajonal fragments that preserve ecosystem services, and maintain in an acceptable status their  
244 biodiversity components and structure.

245

## 246 **CONCLUSIONS**

247 The variations in the size and connectivity of the patches of Pq were significant in the study  
248 period (1974-2016). the major effects were the habitat fragmentation and patch size reduction  
249 leading to connectivity loss. In terms of fragmentation, there was a true fragmentation, and also

250 habitat loss, determined by a high trend to lower the patch size and connectivity. Slight structural  
251 recovery in terms of area could be seen at the end of the study period (2016). This trend is clearer  
252 in connectivity issues measured by ECA%.

253 This work showed that the habitat patches of Pq, have undergone deep transformations since  
254 1974 interchanging the original landscape matrix by other subordinate community, the Short-  
255 Grasses Matrix (Lara & Gandini 2014). These changes could be evidenced through the  
256 application of a methodology originally used for the study of forest fragmentation, the  
257 morphology and connectivity analysis (Vogt et al 2007a).

258

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396

**Table 1** (on next page)

Classification Quality of Pq cover estimation, and numerical results of the morphological and connectivity analysis.

Most significant MSPA results (as percentages over the total) and connectivity analysis: Number of components, maximum, mean and median of connectors / component; and relative ECA

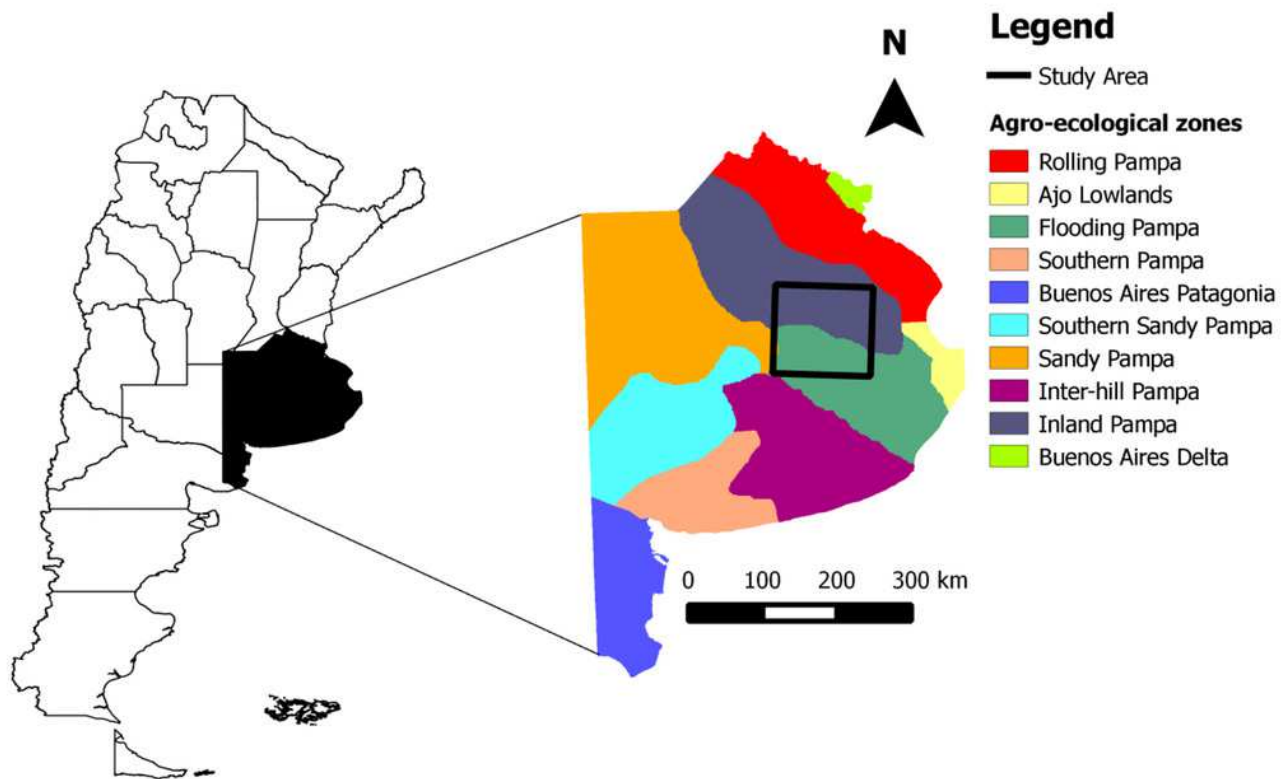
1

| Analysis                   | Parm/Form/year  | 1975   | 1988   | 2000    | 2011   | 2016   |
|----------------------------|-----------------|--------|--------|---------|--------|--------|
| Class. Disagreement % (PJ) | Quantity        | 1      | 5      | 5       | 4      | 2      |
|                            | Allocation      | 10     | 18     | 12      | 9      | 12     |
| MSPA                       | Core (small)    | 4,35   | 3,12   | 3,68    | 3,17   | 3,46   |
|                            | Core (medium)   | 4,65   | 2,08   | 3,12    | 2,85   | 2,17   |
|                            | Core (large)    | 11,94  | 4,58   | 6,06    | 3,73   | 3,71   |
|                            | Islet           | 1,05   | 2,53   | 1,11    | 1,46   | 1,75   |
|                            | Perforation     | 1,72   | 0,62   | 0,56    | 0,28   | 0,50   |
|                            | Edge            | 13,52  | 6,87   | 9,07    | 7,36   | 8,65   |
|                            | Loop            | 1,08   | 0,89   | 0,69    | 0,32   | 0,65   |
|                            | Branch          | 3,23   | 4,85   | 2,80    | 1,82   | 2,46   |
| Components                 | # total         | 2726   | 2061   | 2186    | 2852   | 3772   |
| Conectivity (links)        | Mean of links   | 1153   | 379    | 287     | 167    | 49     |
|                            | median of links | 55     | 49     | 41      | 21     | 43     |
|                            | max links       | 80557  | 28468  | 16738   | 15040  | 43912  |
|                            | ECA (pixels)    | 945277 | 894122 | 1041192 | 236552 | 374184 |
|                            | relative ECA %  | 28     | 57     | 50      | 15     | 25     |

2

# Figure 1

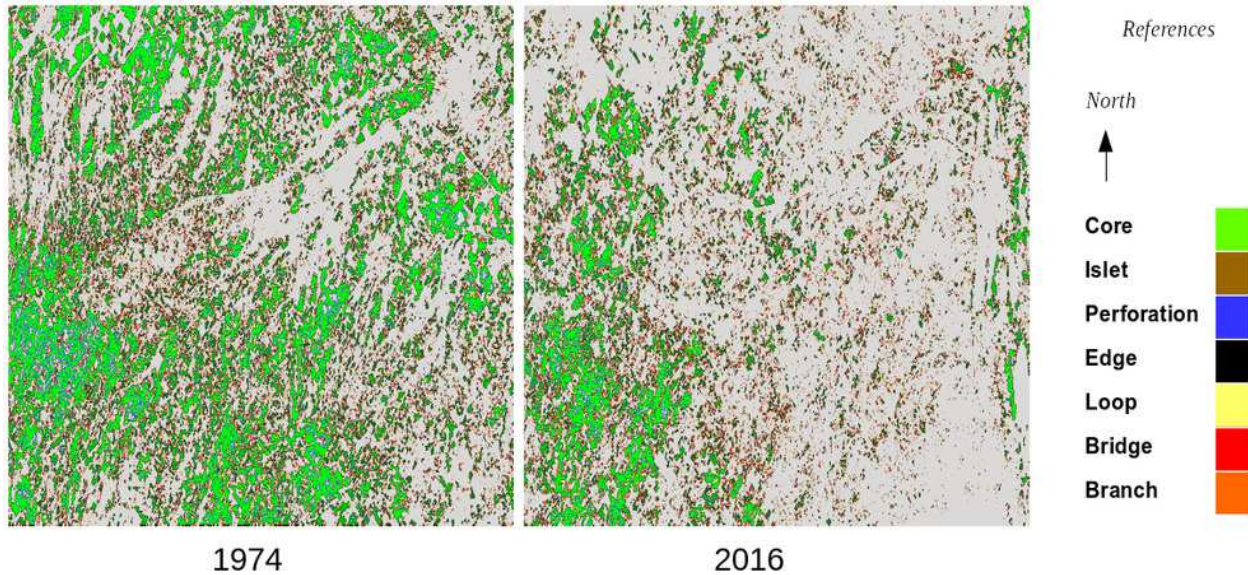
Study area and Agroecological zones





## Figure 2

Map of MSPA segmentation results of *Paspalum quadrifarium* presence-absence data (1974 vs. 2016)





## Figure 3

A visual comparison between 1974 and 2016 connectivity results

Different colors indicate patches belonging to different networks

