Biological Conservation 159 (2013) 507–513

Contents lists available at SciVerse ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

The conflict between agricultural expansion and priority conservation areas: Making the right decisions before it is too late

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A R T I C L E   I N F O

Article history:
Received 15 August 2012
Received in revised form 5 November 2012
Accepted 11 November 2012

Keywords:
Deforestation
Conservation areas
Amphibians
Reptiles
ConsNet

A B S T R A C T

Agricultural expansion generates problems to ecosystems and hinders conservation efforts. Although several researchers worldwide are working to develop efficient approaches for conserving biodiversity adequately, certain decisions should be taken urgently. Here, by using two entire classes of vertebrates (amphibians and reptiles), we calculated optimal conservation area networks (CANs) under nine different scenarios (including three different degrees of land-use and three different degrees of ambition in conservation objectives), in a highly deforested province of central Argentina. We show that if conservation decisions are delayed further, not only will more land be necessary to meet a given conservation objective, but also that optimal CANs will have to have a much greater perimeter, a larger number of fragments and a less appropriate shape. Therefore, as agricultural frontiers continue to advance without designating specific sites for conservation, we are losing the possibility to conserve biodiversity adequately.

1. Introduction

The transformation of natural ecosystems to croplands represents one of the greatest threats to global biodiversity (Sala, 2000). Human population and per capita expectations are growing, and as human consumption increases (Ewers et al., 2009), consequently so does the pressure on natural resources. Humans, in order to satisfy their necessities, modify natural habitats through changes in land cover and land use, mostly by expanding areas for agriculture (Green et al., 2005). Despite a rapid increase in crop yields, croplands have continued to expand around the world (Donald, 2004; Green et al., 2005), with this expansion most likely continuing in the near future (Balmford et al., 2005; Dobrovolski et al., 2011). Agricultural expansion, which is very large in some areas (Zak et al., 2008; Gasparri and Grau, 2009; Dobrovolski et al., 2011; Hoyos et al., 2012), generates clear problems for ecosystems and biodiversity, as well as hinders conservation efforts (Fuller et al., 2007; Gasparri and Grau, 2009). Given these problems, the primary response from conservation biologists has been to develop approaches for the establishment of protected areas that adequately protect biodiversity (Margules and Pressey, 2000; Margules and Sarkar, 2006; Sarkar and Illoldi-Rangel, 2010; Arzamendia and Giraudo, 2012).

Protected areas throughout the world play key roles in the conservation of global biodiversity (Margules and Pressey, 2000; Margules and Sarkar, 2006) and have become of crucial importance in regions that are occupied by intensive human land-uses (Luck, 2007; Dobrovolski et al., 2011). The identification of conservation units based on biodiversity patterns that include and connect several ecosystems is also key in keeping biological processes and ecosystem services operating on broad spatial scales (Margules and Sarkar, 2006). Systematic conservation planning is a protocol that can be used to identify areas for conservation according to different criteria (Sarkar and Illoldi-Rangel, 2010; Illoldi-Rangel et al., 2012). This protocol aims to select areas that can be used to establish conservation area networks (CANs) based on species distributions and by using algorithms that seek to maximize biodiversity representation in the smallest area of land possible (minimum area problem) by incorporating the concepts of complementarity, representativeness, persistence and other spatial criteria, such as the size or compactness (economy) of each individual area (Margules and Sarkar, 2006; Sarkar and Illoldi-Rangel, 2010).

Systematic conservation planning recognizes that species' ranges change in response to anthropogenic disturbances such as

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0006-3207/$ - see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.biocon.2012.11.020

deforestation (Soh et al., 2006) or the spread of agriculture (Bomhard et al., 2005). More specifically, the irreversible conversion of land to intensive agricultural uses can generate unsuitable habitats for many life forms, deeming these areas inadequate for designation as conservation areas and excluding them from conservation planning analyses (Margules and Sarkar, 2006, Illoldi-Rangel et al., 2012). The impossibility of using certain places for the conservation of biodiversity generates important economic and opportunity costs in optimal CANs (Fuller et al., 2007). In a recent paper, Fuller et al. (2007), analyzed the surface of priority conservation areas for endemic mammals in Mexico in relation to different land-use time periods and pinpointed that in 30 years, 90% more land will be necessary to meet a given conservation objective. However, the relationship between agricultural expansion and its impact on optimal CANs remains poorly known.

The establishment of priority conservation areas for amphibians and reptiles could result in an important tool for taking conservation actions. First, because these groups represent an important portion of all terrestrial vertebrates. Also, because amphibians are undergoing an unprecedented global conservation crisis (Houlahan et al., 2000; Stuart et al., 2004), with forecasts for reptiles not very encouraging either (Sinervo et al., 2010), being the habitat loss and fragmentation the most important agents responsible for these crises (Gibbons et al., 2000; Cushman, 2006; Gardner et al., 2007). In addition, as seen in other studies (and places) as well (e.g. Pawar et al., 2007; Urbina-Cardona and Flores-Villela, 2010; Corbalán et al., 2010; De Pous et al., 2010; Arzamendia and Giraudo, 2012), these groups represent excellent surrogates in our study area because they: (a) include representative species that inhabit all of the ecoregions of the province, with at least one threatened or charismatic species in each one; (b) they have an important taxonomical diversity: 106 species recorded in the province, comprising 2 classes (Amphibia and Reptilia), 23 families and 55 genera (DiTada, 1999; Cabrera, 2009; Nori et al., 2011, pers. obs); (c) include three endangered, 11 vulnerable and four poorly known species (Vaira et al., in press; Giraudo et al., in press; Abdala et al., 2012) (Supplementary Table A1). (d) lastly, the most threatened of the environments of the province (Chaco Forest region), showed the greatest diversity and the highest number of threatened or range restricted species of amphibians and reptiles.

Based on the hypothesis proposed by Fuller et al. (2007), the aim of this study is, to provide new knowledge to help understand how the most important physical characteristics of CANs (area, perimeter, shape and fragmentation) (Margules and Sarkar, 2006) change with the progressive conversion of ecosystems to intensive human land-uses. In other words, we studied the influence of agricultural expansion on the size and spatial configuration of CANs. We used two entire classes of threatened vertebrates (amphibians and reptiles) and calculated the priority CANs under nine different scenarios in Córdoba, a mostly deforested province in Argentina. In this province, the deforestation rate is more than 12 times the global average, with more than 80% of its closed forest lost in the last 20 years (Zak et al., 2008; Hoyos et al., 2012). Only approximately 44% of the province in 2009 was neither urbanized nor occupied by intensive agriculture (INTA, 2009), and therefore could be potentially included in systematic conservation planning.

2. Materials and methods

2.1. Study area

Our analysis took place in the Cordoba Province, located in central Argentina. Predominant ecoregions in the province are: Great Chaco Forest, located in the north and west. Chaco Serrano and Grassland Highlands, a north-to-south mountain range located in the west of the province. Pampas, in the southeast, and Espinal, a transitional zone located between the Chaco and Pampas ecoregions. See Appendix A and Supplementary Fig. A1 for details.

2.2. Data and analyses

Our study began with a database of 2453 individual records for 99 species comprising data from 1979 to 2011. The records were obtained from herpetological collections belonging to different institutions (CZA: Centro de Zoología Aplicada, FML: Fundación Miguel Lillo, MACN: Museo Argentino de Ciencias Naturales, MLP: Museo de Ciencias Naturales de La Plata, UNRC: Universidad Nacional de Río Cuarto, AC: Diagnostic collection of MRC, housed at Universidad Nacional de Cordoba), relevant literature (e.g. di Tada and Bucher, 1996; di Tada, 1999; Leynaud and Bucher, 1999) and field trips made in the last 25 years by the authors. In order to estimate the potential distribution of each species, we generated species distributional models (SDMs) by implementing MaxEnt 3.3.3e (Phillips et al., 2006, 2009). The goal of this methodology is to calculate the potential geographic distribution for a species over a given geographical space, using information from the observed association of the localities of the species with other information, in this case, environmental and topographic variables. This algorithm has been shown to be a robust method for presence-only datasets (Elith et al., 2006; Wisz et al., 2008). We used 19 bioclimatic layers and one topographic layer as predictors (Hijmans et al., 2005; available at http://www.worldclim.org). We ran MaxEnt under the "autofeatures" as suggested by Phillips and Dudík (2008).

Of the original 99 species, we generated SDMs for 77 species that had more than four unique localities. We assessed model performance using 25% of the records of each species as "test data" in order to calculate the area under the receiver operating characteristic curve (AUC/ROC). The models to be included in the CAN prioritization were chosen according to the following criteria: (a) higher area-under-the-curve test values (models with values of AUC less than 0.70 were discarded) (Pawar et al., 2007) and (b) expert opinions for each species regarding the potential distribution in the study area. We included 70 of the for the area prioritization analysis (Supplementary Table A1); the seven remaining models did not satisfy the mentioned criteria and were eliminated. We classified each potential distribution map as either presence or absence. Following Pearson et al. (2006), we selected the 10 percentile training presence as the threshold value in order to not overestimate the potential distributional range of the species.

The study area was divided into 225,645 square cells of 0.0083° (0.8 km^2), of which 18,992 cells were occupied by current conservation areas and permanently included in the results of all analyses. We performed area prioritization analyses by implementing ConsNet v 2.0 to determine the smallest set of cells (with the best shape possible) so that each species could meet predetermined representation targets in different land use scenarios (Ciarleglio et al., 2010). This is a comprehensive software package used for the design and analysis of conservation area networks (CANs) representing biodiversity by implementing multiple criteria and based on specific goals (see Ciarleglio et al., 2009, 2010 for details). ConsNet runs were initialized based on the RF4 adjacency algorithm (Ciarleglio et al., 2009, 2010), in which cells that contain the rarest species that do not meet the representation target are chosen first. In the event of a tie, cells are chosen based on complementarity. ConsNet was run with the "advanced: escape with spatial neighborhood" selection strategy for 800,000 iterations. The solutions generated were then improved by applying a basic selection strategy (using large neighborhoods only) as recommended by Ciarleglio et al. (2009, 2010).