Effect of protected areas in reducing land development across geographic and climate conditions of a rapidly developing country, Spain

David Rodríguez-Rodríguez [1,2], Joana Sebastiao[2,3], Ángel Enrique Salvo Tierra[3], Javier Martínez-Vega[1]

[1] *Institute of Economy, Geography and Demography, Spanish National Research Council (IEGD-CSIC), Associated Unit GEOLAB, C/Albasanz, 26-28, 28037 Madrid, Spain*

[2] *European Topic Centre, University of Malaga, Campus de Teatinos s/n, 29010 Malaga, Spain*

[3] *Department of Plant Biology, Faculty of Sciences University of Malaga, Campus de Teatinos s/n, 29071 Malaga, Spain*

Abstract: Protected areas (PAs) aim at safeguarding biodiversity and ecosystem services in the long term. Despite the remarkable growth in area covered by PAs in recent years, bio- diversity trends continue to worsen as a result of serious global pressures such as habitat destruction and degradation. One main cause of habitat destruction and degradation is land development that implies the replacement of natural land uses–land covers (LULCs) with artificial ones. Here, we assessed the effectiveness of four PA networks at preventing land development in Spain, a biodiversity-rich country that has experienced recent rapid environmental transformations, using two models of increased validity: an original model and a biophysically enhanced model. We applied a before–after control–impact (BACI) design whereby absolute artificial area increase (AAI) and relative artificial area increase (RAI) were compared across PA categories (nature reserves [NRs], nature parks [NPs], Sites of Community Importance [SCIs], and Special Protection Areas [SPAs]), study zones (coastal and inland), and climates (Atlantic and Mediterranean) using CORINE Land Cover (CLC) data and two control zones: 1- and 5-km buffers around protected polygons. NRs prevented land development, whereas other categories reduced it moderately to very substantially in the assessed period. AAI was especially intense in inland SPAs and NPs. NRs and NPs were the most effective PA categories inland, whereas NRs and SPAs were the most effective ones on the coast. Land development was greater on the Spanish coast than inland inside and outside PAs, especially around Macaronesian and Mediterranean PAs. Atlantic PAs experienced similar or greater land development values than surrounding areas. Our results are intended to guide future conservation efforts in Spain, chiefly on its heavily pressured coastal environment.

Keywords: BACI design, land use–land cover, multiple use, Natura 2000, reserve

1. Introduction

Protected areas (PAs) are legally designated sites aimed at the long-term conservation of biodiversity and associated ecosystem services and cultural values (Dudley, 2008). They are the main policy to stop global biodiversity loss (Convention on Biological Diversity, 1992). Even though the number of PAs and the area covered by them have greatly expanded in the last decades (Bhola, Juffe-Bignoli, Burguess, Sandwith, & Kingston, 2016), biodiversity continues to be lost (Butchart et al., 2010), which raises the question of PA effectiveness to abate pressures on biodiversity. Among these, LULC changes leading to natural and seminatural habitat destruction and degradation are the main causes of biodiversity loss (World Wildlife Fund, 2016).

At global scale, urban development is expected to degrade biodiversity in most ecoregions, impacting rare species and PAs (McDonald, Kareiva, & Forman, 2008). Coastal areas are especially vulnerable to land development pressures from housing and tourist infra- structure building (Pan et al., 2016). New land uses alter important ecosystem services provided by coastal habitats of great ecological value such as mangroves or wetlands (Khamis, Kalliola, & Käyhkö, 2017), also affecting PAs (Dal & Baysan, 2011). In Europe, pressure from land development in and around PAs is a long-lasting concern (Brotherton, 1982). Actually, transformation of natural and seminatural habitats to artificial land covers is the dominant LULC change in the region (European Environment Agency [EEA], 2011). Land development is an especially concerning LULC change due to its serious and largely irreversible impacts on natural habitats and associated biodiversity (EEA, 2011; McKinney, 2002).

Spain is a Euro-Mediterranean country rich in biodiversity (Araújo, Lobo, & Moreno, 2007; Médail & Quézel, 1999; Montes Santos & Benayas, 2011) that expands across four biogeographic regions: Alpine, Atlantic, Mediterranean, and Macaronesian (EEA, 2015). Its geological, geographic, climatic, and relief diversity determines extraordinary biodiversity figures in the European context, with 75% of all European vertebrate species and 50% of all endemic plant species of the continent (Spanish Government, 2008). On the other hand, Spain has experienced enormous socio-economic changes in recent decades leading to intense LULC changes and landscape transformation (Jiménez, 2012; Montes et al., 2011; Stellmes, Röder, Udelhoven, & Hill, 2013). Residential, industrial, and infrastructural development along the 1990s and, especially, early 2000s has expanded the Countries' artificial areas hugely, with widespread impact on territorial sustainability and wildlife (García-Ayllón, 2015; Jiménez, 2012; Torres, Jaeger, & Alonso, 2016). Actually, LULC changes towards rural abandonment, agricultural intensification and land development, especially along the coast, are the main pressures leading to biodiversity loss in Spain (Custodio et al., 2016; de Andrés, Barragán, & García Sanabria, 2017; Montes et al., 2011; Rey Benayas, Martins, Nicolau, & Schulz, 2007).

LULC change has been largely studied in Spain, across its whole territory (Jiménez, 2012; Stellmes et al., 2013), inside and around PAs (Hewitt, Pera, & Escobar, 2016; Martínez-Fernández, Ruiz-Benito, & Zavala, 2015; Rodríguez-Rodríguez & Martínez-Vega, 2018a), across biogeographic regions (Martínez-Fernández et al., 2015), and on specific environments such as the coast (Jiménez, 2007, 2012; Jiménez, Prieto, Riechmann, & Gómez, 2005). Recent studies have delved into the influence of PA management, territorial regulations, and biophysical characteristics on LULC changes in and around Spanish PAs using semi-experimental designs (Martínez-Fernández et al., 2015; Rodríguez-Rodríguez & Martínez-Vega, 2017, 2018a, 2018b) in order to help to fill the gap in PA effectiveness studies based on evidence (Juffe-Bignoli et al., 2014). In their comprehensive studies Martínez- Fernández et al. (2015) and Rodríguez-Rodríguez and Martínez-Vega (2018a) shed light on a number of unresolved issues to understand the influence of PAs of different legal, managerial, and environmental contexts on land development. However, the fact that country-wide censuses of PAs were used in an environmentally diverse country such as Spain (Spanish Government, 2008) made it likely that the influence of relevant land development factors such as climate (Barbero-Sierra, Marques, & Ruíz-Pérez, 2013; Jiménez et al., 2005) and proximity to the coast (Jiménez, 2007, 2012; Jiménez et al, 2005) was not sufficiently considered, thus providing improvable results in terms of accuracy.

According to Jiménez (2007), coastal municipalities in Spain experienced an average land development increase of 28% between 1987 and 2000. As a result of such processes, the density of residential areas on the Spanish coast in more than 10-fold the density of those areas inland, chiefly as a result of them being popular tourist destinations (Instituto Nacional de Estadística [INE], 2017). Jiménez (2007) and Martínez-Fernández et al. (2015) also showed differential values of land development across Mediterranean and Atlantic regions, largely different on climatic grounds (EEA, 2002; Spanish Government, 2008). Between 1987 and 2000, residential population on the Spanish Mediterranean coast increased by more than 7%, whereas on the Atlantic coast, it decreased by 1.6%, the percentage of artificial areas being nearly double in the Mediterranean (Jiménez, 2007). Tourism's total contribution to Spanish GDP accounted for 14.2% in 2016 and is forecasted to rise by nearly 2% per annum, whereas the sector's total contribution to employment was 14.5% of total employment that year, with similar increase forecasts (World Travel and Tourism Council, 2017). Moreover, 91% of foreign tourists' overnight stays in Spain occur on the coast (INE, 2017), which makes coasts an essential asset to the Spanish economy. As a result, Spanish coastal ecosystems are among the most threatened ones and with worst recent environmental trends (Montes et al., 2011). Therefore, assessing the effectiveness of coastal sustainability policies is paramount if coastal biodiversity and socio-economic development are to coexist.

Thus, in this article, we wanted to expand and refine previous findings by testing a number of research hypotheses: (a) Land development pressure on the coast is greater than inland in Spain; (b) land development pressure on the Mediterranean and Macaronesian coasts is greater than on the Atlantic coast due to better climate conditions; (c) PAs are an effective territorial policy against land development regardless of the study zone: coastal or inland; and (d) PA categories have different effectiveness against land development according to their legal stringency and management.

2. Materials and methods

2.1. Study zones

Our study zones included (a) the coastal zone encompassing the 10-km inland from the sea line (Instituto Geográfico Nacional [IGN], 2015) extending over 39,800 km2 where coastal influence is maximal. That distance has been used in a number of LULC studies in Spain (Chica-Ruíz et al., 2014; Jiménez, 2010, 2012; Romano, 2011), and (b) the inland zone covering the area lying 100 km from the sea line (281,200 km2), where coastal influence was considered non-existent. We included both Spanish archipelagos: the Balearic Islands, in the Mediterranean Sea, and the Canary Islands, in the Atlantic Ocean (Figure 1). We used official biogeographic regions' cartography as a proxy for climatic conditions across Spanish coastal areas, as biogeographic regions are largely defined by climate (EEA, 2008). Climate- wise, all north-western coastal PAs belong to the Atlantic biogeographical region with mild temperatures and high rainfall throughout the year, whereas the remainder of PAs are in the Mediterranean region and the Macaronesian region (the Canary Islands), both with Mediterranean climate, characterised by high temperatures and low precipitation during the summer leading to droughts. Seasonal temperature contrast is, however, notably greater in the Mediterranean region than in the Macaronesian region except near the coast, where Mediterranean areas have very similar climate conditions to Macaronesian areas (EEA, 2008).

2.2. Studied networks

We assessed the effects of four PA networks with clear legal and managerial characteristics across Spain: NRs (IUCN's Management Category I), NPs (IUCN's Category V), SCIs (IUCN's Category IV), and SPAs (IUCN's Category IV; Atauri, Múgica, Gómez-Limón, & de Lucio, 2008). NRs and NPs are nationally designated PAs, NRs being reserves and NPs, multiple use PAs (Spanish Government, 1989). SCIs and SPAs and Special Areas of Conservation make the European-wide Natura 2000 network (European Economic Community, 1992). According to the protection framework by Rodríguez-Rodríguez, Rodríguez, and Abdul Malak (2016), those PA categories represent a gradient of protection: high legal stringency and active management (NRs) > moderate legal stringency and active management (NPs) > moderate legal stringency and no active management (SCIs and SPAs).

2.3. Methods

We operationally defined 'land development' as the replacement of natural and seminatural LULCs with artificial ones leading to complete or almost complete land degradation as defined by the United Nations (United Nations Convention to Combat Desertification, 1994). LULCs were obtained from CLC data for 1987 and 2006 (IGN, 2016). We considered 'artificial areas' the following environmentally unsustainable level 2 CLC artificial subclasses: 1.1, urban fabric; 1.2, industrial, commercial, and transport units; and 1.3, mine, dump, and construction sites. Some CLC class 1 area (chiefly, many existing transport infrastructures by 2006) did not appear in the official Spanish CLC-2006 version (IGN, 2016), so we created a new CLC-2006 class 1 layer that included all CLC-1990 class 1 covers plus those 'new' class 1 covers in CLC-2006, despite a slight excess of artificial covers that might have been restored in that period. Our definition of land development is similar to that of 'land take' by the EEA (2018). It differs in the fact that we excluded green urban areas, that is, CLC subclass 1.4: 'artificial, non-agricultural vegetated areas,' as they normally entail limited soil sealing, therefore causing substantially less land degradation. They are thus more biodiversity friendly and prone to ecological restoration than the remainder LULCs in CLC subclass 1.

Official census data boundaries of all PAs belonging to the four categories (N = 2,369) were downloaded from the Spanish Ministry of Environment's website (Spanish Government, 2015) and unioned in a single protected polygon (PP) layer, in order to determine legal overlaps among the four PA categories and the initial designation date of each area (PP) exactly. PPs have been used in a number of recent assessments on PA effectiveness for being a more discriminating and valid approach than traditional approaches that assessed PAs' official boundaries regardless of total or partial legal overlaps (Rodríguez-Rodríguez, Rodríguez, Blanco, & Abdul Malak, 2016; Rodríguez-Rodríguez & Martínez-Vega, 2018a, 2018b). In the case of legal overlap, the oldest designation category was assigned to each PP, for being the initial date of the site's protection. Then, the complete PA layer was clipped against the two study zones (inland and coastal), and PPs equal to or bigger than 100 ha that were designated between 1987 (t1) and 2000 (t2) in both zones were retained. We obtained 724 PPs in total, 511 in the inland zone and 213 in the coastal zone. We made sure that each PP had been designated 'after' the LULC data (satellite images) in their area were taken (after t1), and at least 3 years 'before' the first available comparison time (t2) to allow enough time for the PA to show some effectiveness. We used a before–after control–impact research design (Smith, 2002) whereby we compared land development figures inside PAs (cases) and in 'control' areas 'before' and 'after' the 'impact' we aimed to measure: PA's protection determined by each PP's designation date. Then, 1-and 5-km buffer areas surrounding each PP were created as 'controls' and assigned to that PP's designation category. Buffer areas smaller than 100 ha were also discarded due to CLC's relatively broad spatial resolution and to minimise boundary alignment errors.

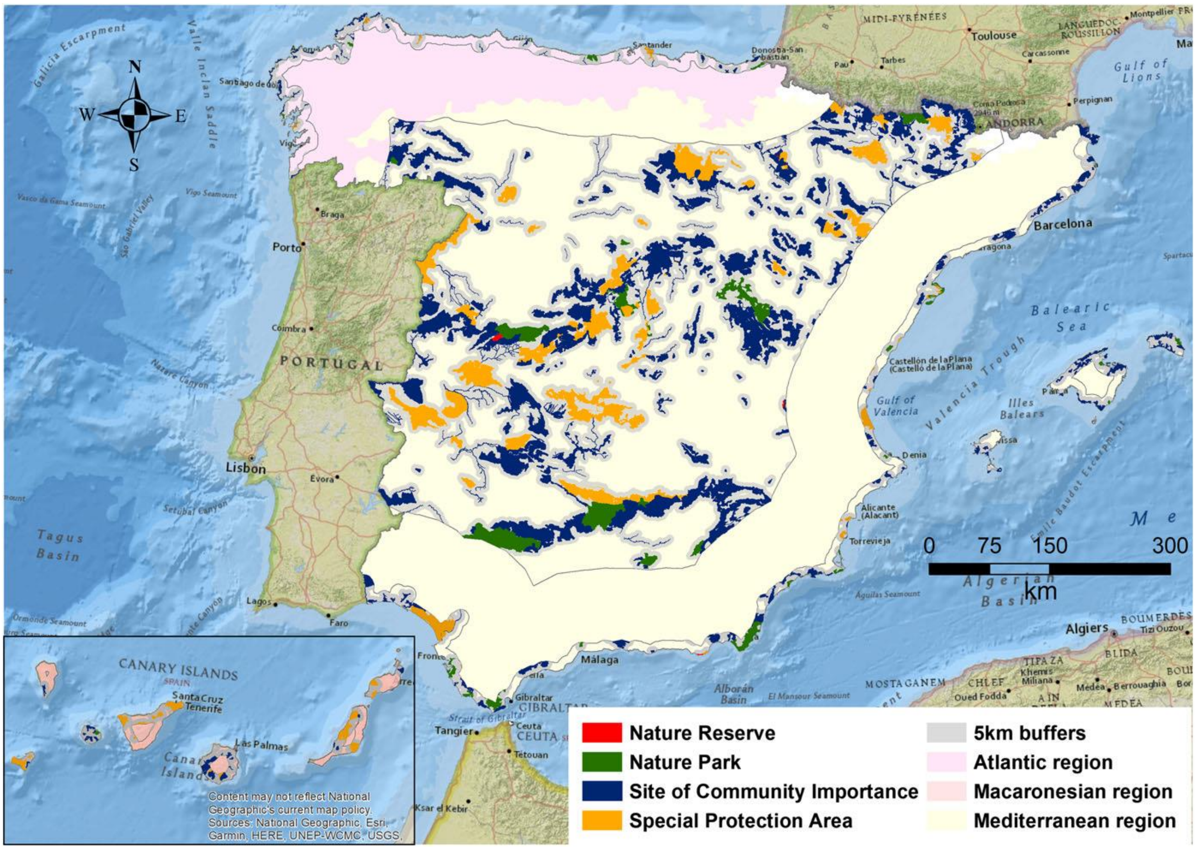

FIGURE 1. Protected area categories and study zones.

We assessed land development using two models: (a) an original model and (b) a biophysically enhanced model (BEM). To create the original model, we deleted the areas affected by additional territorial regulations limiting land development that confer additional protection from the assessed areas, following methodological recommendations (Rodríguez-Rodríguez & Martínez-Vega, 2018a). We produced a legal 'exclusion layer' including public hydraulic domain areas (Spanish Government, 2001), public coastal domain areas (Spanish Government, 1988), and other PAs designated until 2008 (Spanish Government, 2007; only for controls), from cases (PPs) and/or controls (Appendix A). We did an initial appraisal of the biophysical similarity between cases and controls of the same network for each study zone in order to estimate the validity of case–control comparisons, as previously suggested (Andam, Ferraro, Pfaff, Sánchez- Azofeifa, & Robalino, 2008; Mas, 2005). For this, we considered the following five biophysical covariates: altitude, slope, distance to main cities, distance to major transport infrastructures, and degree of initial treeless cover (Appendix B). All those variables have been shown to affect LULC changes (Andam et al., 2008; Rodríguez-Rodríguez & Martínez-Vega, 2018a).

For biophysical similarity calculations, we used a similarity index based on the normalised Manhattan similarity coefficient (Cha, 2007), according to the following formula:

$$S(X,X') = 1 - \frac{\sum_{i=1}^{k} |X_i - X'_i| \, / Range\ (Xi)}{K}$$

where Xi is the median value of group X for variable i; Range is the amplitude of measurement Xi in the study area; and K is the number of variables used to assess groups X and X′. The Manhattan similarity coefficient ranges between 0 (complete difference between compared group values) and 1 (complete similarity).

In order to increase validity of comparisons of the original model, we created a BEM by making cases and controls of each PA network biophysically similar according to the aforementioned covariates. To validly compare cases and controls, we made the ranges of the five covariates equivalent between cases and controls of each PA network and study zone by excluding covariates' values of controls (or cases) that were outside the ranges of their cases (or controls; e.g. greater slope; lesser altitude). Thus, we finally compared PPs and buffer areas within the same ranges of altitude, slope, distance to cities and infrastructures, and degree of treeless cover. 'Altitude' was excluded from the BEM due to its high correlation (>0.8) with "slope" in both study zones (Appendix C). Once covariate homogenous PPs and buffers were produced for each PA network, we calculated the biophysical similarity of the resulting cases and controls (S) as shown above. We then compared case–control similarity results between the original model and the BEM to check whether the latter had enhanced the validity of case–control comparisons. For this, a global similarity index was computed for the original model and the BEM by averaging S for the four remaining covariates (Appendix C).

Absolute artificial area increase (AAI) and relative artificial area increase (RAI) were computed for each PP and buffer network:

$$\mathrm{AAIx} = \frac{(\mathrm{ARTx}_{(t2)} - \mathrm{ARTx}_{(t1)})}{\mathrm{AREAx}} \text{ x } 10^2$$

$$\mathrm{RAIx} = \frac{(\mathrm{ARTx}_{(t2)} - \mathrm{ARTx}_{(t1)})}{\mathrm{ARTx}_{(t1)}} \text{ x } 10^2$$

where $\mathrm{ARTx}_{(t1)}$ is the sum of artificial areas in PP or buffer network x around 1987 (in hectare), $\mathrm{ARTx}_{(t2)}$ is that sum around 2006, and AREAx is the total area of the PP or buffer network x. Finally, land development results on the coastal zone were disaggregated by biogeographic region: Atlantic and Mediterranean, as proxies of climate. Figure 2 summarises the methods used in the study. All GIS calculations were done using Arc-GIS 10.3 (Environmental Systems Resource Institute, 2014) and QGIS (2017) 2.18.7. Statistical calculations were performed in SPSS and Microsoft Excel.

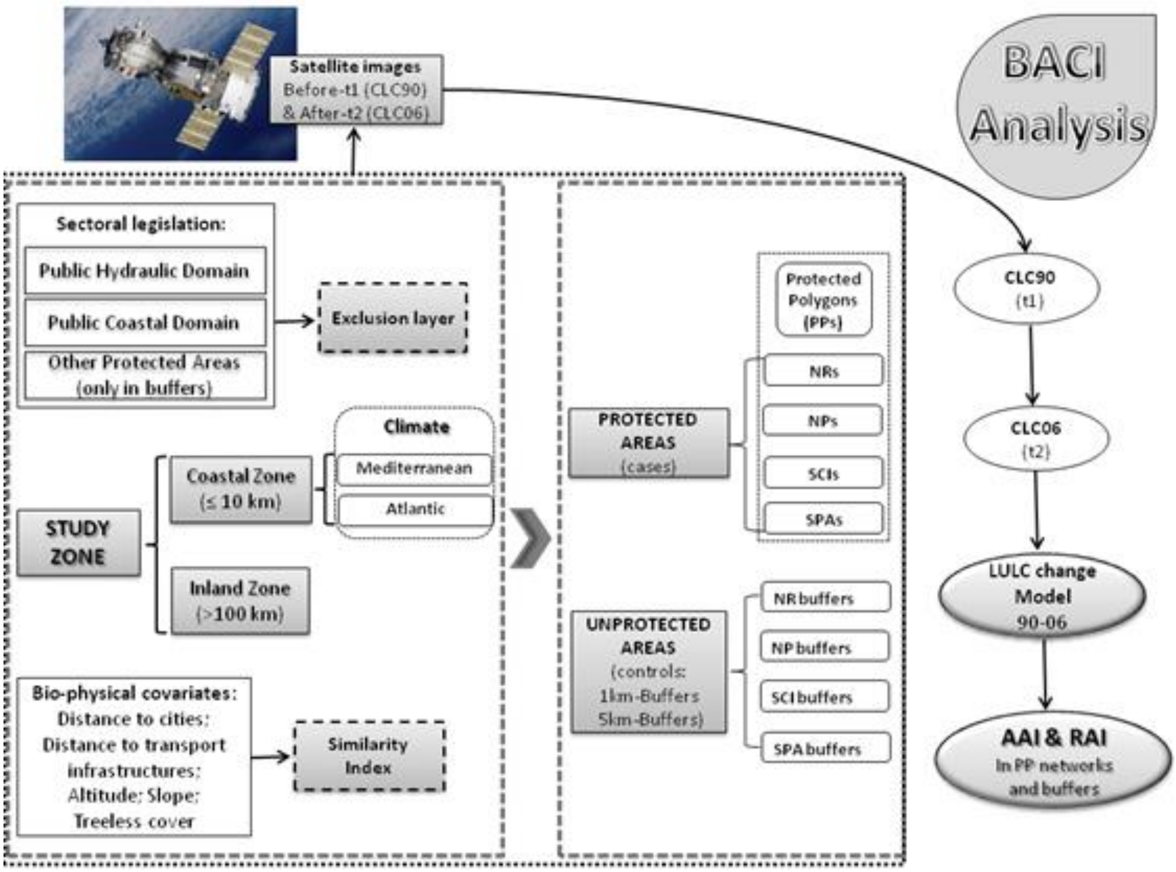

FIGURE 2 Methodological outline of the study. NRs: nature reserves; NPs: nature parks; SCIs: sites of community importance; SPAs: special protection areas; BACI: before–after control–impact; CLC: CORINE Land Cover; LULC: land use–land cover; AAI: absolute artificial area increase; RAI: relative artificial area increase

3. RESULTS

3.1. *Case–control similarity*

In the original model, S was greater than 0.82 for all pair-wise comparisons except between NRs and their 5-km buffers inland (S = 0.75) and on the coast, where similarity was the lowest (S = 0.65; Appendix C). Biophysical similarity change in the BEM was inconsistent. S generally increased in the BEM for both controls in the inland zone, but it mostly decreased for 1- and 5-km controls in the coastal zone (Appendix C). Thus, here, we present the results of the original model. PA effectiveness results of the BEM can be consulted in Appendix D.

3.2. | *PA effectiveness*

Table 1 shows AAI and RAI values of the original model for each PP and buffer network by study zone, whereas Table 2 shows land development values on the Spanish coasts. According to both of them, our initial hypotheses can be validated:

1. PAs are an effective territorial policy against land development.

This hypothesis is largely supported by evidence. AAI and RAI values were smaller in PAs than in buffer areas in both zones, except inland SPAs' RAI and coastal SCIs' RAI that were greater than their control areas.

2. PA categories have different effectiveness against land development according to legal stringency and management.

This hypothesis is supported by evidence. NRs were the most effective PA category in both study areas, whereas SPAs and SCIs were the least effective categories in inland areas and coastal areas, respectively.

3. Land development pressure on the coast is greater than inland.

This hypothesis is largely supported by evidence. AAI was nearly three times greater on the coast outside PAs and 1.33 times greater inside coastal PAs. However, RAI values were systematically greater inland.

TABLE 1 Land development results of the original model

INLAND ZONE												
Protected Area Network	AAI	RAI	Total network area (ha)	New artificial area (ha)	1KM AAI	1KM RAI	Total 1km buffer area (ha)	New artificial area in 1km buffer (ha)	5KM AAI	5KM RAI	Total 5km buffer area (ha)	New artificial area in 5km buffer (ha)
NRs	0	0	9,254	0	0.44	0	1,499	6.66	0.50	92.82	10,797	53.89
NPs	0.22	44.96	561,367	1237.84	4.47	79.08	59,235	2,648	3.53	67.35	380,266	13426.88
SCIs	0.04	32.36	3,487,177	1273.77	0.70	50.15	1,910,642	13,456	0.67	54.58	9,321,780	62369.4
SPAs	0.26	81.95	1,624,293	4193.27	1.41	69.33	406,383	5,714	1.53	74.39	2,033,625	31206.26
ALL PAs	0.12	56.79	5,682,091	6704.87	0.92	56.80	2,377,759	21,824	0.91	60.75	11,746,468	107056.43
COASTAL ZONE												
Protected Area Network	AAI	RAI	Total network area (ha)	New artificial area (ha)	1KM AAI	1KM RAI	Total 1km buffer area (ha)	New artificial area in 1km buffer (ha)	5KM AAI	5KM RAI	Total 5km buffer area (ha)	New artificial area in 5km buffer (ha)
NRs	0	0	531	0	4.67	80.26	635	30	2.53	67.73	6,104	154
NPs	0.15	16.48	99,783	149	2.63	29.39	53,378	1,402	2.68	32.00	266,206	7,139
SCIs	0.20	38.87	361,491	735	1.96	25.03	279,229	5,488	2.33	33.05	1,341,370	31,328
SPAs	0.09	22.58	193,751	172	5.47	74.75	55,592	3,043	4.14	62.88	338,165	14,000
ALL PAs	0.16	29.66	656,713	1,056	2.56	32.34	388,833	9,964	2.69	37.70	1,951,845	52,621

TABLE 2 Land development results in the coastal zone by biogeographic region

ZONE	TOTAL AREA (ha.)	ARTIFICIAL AREA 1990 (ha.)	ARTIFICIAL AREA 2006 (ha.)	NEW ARTIFICIAL AREA (ha.)	RAI	AAI
Protected Areas - Atlantic	50,942	460	640	180	39.13	0.35
Protected Areas - Mediterranean	494,028	3,397	4,246	849	24.99	0.17
Protected Areas - Macaronesian	129,040	63	153	89	141.07	0.07
1Km Buffer - Atlantic	298,829	4,187	5,194	1,008	24.06	0.34
1Km Buffer - Mediterranean	946,014	28,734	36,706	7,972	27.74	0.84
1Km Buffer Macaronesian	208,562	2,605	4,515	1,909	73.28	0.91
5Km Buffer - Atlantic	3,617,539	18,931	23,803	4,872	25.74	0.13
5Km Buffer - Mediterranean	12,579,362	122,510	163,450	40,941	33.42	0.33
5Km Buffer Macaronesian	5,002,219	19,097	31,672	12,574	65.84	0.25

4. Land development pressure on the Mediterranean and Macaronesian coasts is greater than on the Atlantic coast.

This hypothesis is moderately supported by evidence (Table 2). There was differential land development pressure inside and around PAs depending on the coastal zone. The AAIs of Mediterranean and Macaronesian buffer areas were substantially greater than those of the Atlantic region. RAIs were greater too. However, AAI in Atlantic PAs was five times greater than in Macaronesian PAs and twice as large as AAI in Mediterranean PAs. The proportion of PA categories was similar in the Atlantic and Mediterranean coastal zones. So was the proportion of wetland area, around 14%. However, the proportion of legally overlapping PAs was substantially higher in the Mediterranean zone compared with the Atlantic zone (74.5% vs. 47.3%). Of the 52.7% non-overlapping area in Atlantic PAs, 82% corresponded to SCIs and 18% to NPs. In contrast, all Atlantic SPAs and 99.6% of Mediterranean SPAs overlapped with other legal categories assessed here.

4. DISCUSSION

PAs reduced land development inland and on coastal areas of Spain between 1987 and 2006. NRs and SPAs were especially effective in coastal areas, whereas SPAs were just moderately effective inland. The high effectiveness of Spanish NRs to prevent land development has been previously stated as resulting from high legal and managerial protection and legal overlaps (Rodríguez-Rodríguez & Martínez-Vega, 2018a). The very different performance of coastal and inland SPAs is probably influenced by the larger proportion of legal overlap in coastal SPAs and/or the types of ecosystems that SPAs protect in both zones. In Spanish inland areas, many SPAs have an agrarian character (Araújo et al., 2007; Martínez-Fernández et al., 2015) with some biophysical characteristics that make them more prone to land development than other PA categories, such as greater initial percentage of treeless cover (Rodríguez-Rodríguez & Martínez-Vega, 2018a). In turn, coastal SPAs likely include a large proportion of wetland area, unsuitable to land development, and may overlap with other legal PA categories more easily. For instance, in Andalusia, southern Spain, all the regional 25 Ramsar sites are SPAs (20 of them), and/or other nationally designated PAs (all of them), additionally increasing protection and reducing land development pressure in these biologically important wetlands (Andalusian Government, 2016).

Land development pressure on the coast was clear and generally greater than inland, underpinning previous claims in Spain (de Andrés et al., 2017; Jiménez, 2012; Jiménez et al., 2005; Pons & Rullán, 2014) and abroad (Burak, Dogan, & Gazioglu, 2004; Creel, 2003; Iglesias-Campos, Meiner, Bowen, & Onwona Ansong, 2015; Zhang & Song, 2003). It was similarly intense in 1- and 5-km buffers around PAs, both inland and on the coast, although land development buffer values on the coast nearly doubled those inland. This trend suggests PA isolation and connectivity issues on the Spanish coast, particularly for SPAs and NRs, the categories with the highest AAI values in proximal buffer areas (Capdepón, 2016). Parallel issues arise in the United States where Radeloff et al. (2010) point to increasing isolation of Pas by new urban uses around the Country's PAs that may compromise ecological connectivity if recent developmental trends continue (Wade & Theobald, 2010).

Coastal PAs also experienced a 25% greater AAI than inland Pas confirming that land development has been a serious pressure to Spanish coastal PAs despite legal protection (Greenpeace, 2018). Coastal development mostly affected SCIs and, to a lesser extent, NPs. Land development was particularly intense on the Macaronesian and Mediterranean coasts, more than doubling land development figures on the Atlantic coast outside PAs, similar to previous claims (Jiménez, 2007). Tolerance with or even promotion of massive development proposals by competent local authorities, which obtain substantial income through housing permits and taxes, and insufficient supervision by regional authorities have resulted in overdevelopment of the Spanish Mediterranean coast (Jiménez et al., 2005). In other Mediterranean countries such as Italy, rapid coastal urbanisation is similarly eased by administrative fragmentation, low landscape legislation effectiveness, and weakness of sanctions to illegal developments (Falco, 2017). In contrast, biodiversity conservation guides land development along coastal areas in other countries to a large extent (Najafinasab, Karbassi, & Ghoddousi, 2015). In city-states such as Singapore, a system of PAs and corridors seeks to preserve biodiversity and ensure ecological connectivity in a rapidly increasing urbanisation context (Tan, 2006).

In contrast to the comparatively low land development around them, PAs on the Atlantic coast doubled land development figures of PAs on the Mediterranean coast and experienced five times more land development than Macaronesian PAs. Greater environmental and landscape quality of Atlantic PAs is likely to have driven holiday home construction by high-income tourists at discomfort with mass tourism and high facility concentration typical of the Spanish Mediterranean coast (García-Ayllón, 2015; Rico-Amorós, Olcina-Cantos, & Sauri, 2009). Other explaining factors being similar, the substantially lower degree of legal overlap in Atlantic PAs, most of which corresponded to legally lenient, unmanaged SCIs in the assessed period, may additionally explain this finding. Previous studies did not find a correlation between the number of legal overlaps and land development in Spanish PAs but did find differences in effectiveness in particular PA categories with and without legal overlaps (Rodríguez-Rodríguez & Martínez-Vega, 2018a), which suggests some influence of this variable on land development at PA category level in Spain. The greatest relative effectiveness of coastal Macaronesian PAs is encouraging to long-term coastal conservation in a small biogeographic region that is rich in endemic biodiversity (Santamarta, Naranjo, & Arraiza, 2014) and especially important in the Canary Islands, where over 75% of the species of conservation concern are in unfavourable or unknown conservation status (Prieto, 2016) and where ecosystem services are deteriorating (Montes et al., 2011).

Legal reinforcement in two target areas emerges as a priority of this analysis. On the one hand, legal upgrading of single-designation Atlantic coastal PAs and/or addition of further legal designations to them. On the other, severely restricting further development on the Spanish Macaronesian and Mediterranean coasts by promoting more environmentally friendly and still socio-economically sustainable LULCs that may include new PAs and other protected zones of environmental and cultural value (Rodríguez-Rodríguez, 2012). Tourism is heavily driven by climatic conditions and available services (Hadwen, Arthington, Boon, Taylor, & Fellows, 2011; Hein, Metzger, & Moreno, 2009; Roca, Riera, Villares, Fragell, & Junyent, 2008). Notwithstanding land development pressure on Atlantic PAs, this coast has so far ridden out the land development issues of the Mediterranean coast by being naturally protected against residential development by its rainy climate (Spanish Government, 2008) that substantially reduces its attractiveness as a tourist destination (Collet, 2010). Jiménez (2012) showed that the amount of artificial area in the 10-km stripe from the Mediterranean coastline of Spain more than doubled that of the (north) Atlantic coast, at 13.5% and 5.8%, respectively, in 2006, with RAIs of 21.3% versus 9.3% in the 1987–2000 period. By 2000, the Mediterranean global biodiversity hotspot was the second one with the greatest urban area, and artificial LULC increase prospects were as large as 160% by 2030 (Seto, Guneralp, & Hutyra, 2012). Long-lasting attractiveness of Spain as a tourist destination and current instability in a number of competing countries around the Mediterranean Sea are driving increasing tourist numbers in the country (INE, 2018; Tourspain, 2018; Vera & Ivars, 2009) and rising space, water, and energy demands on its already heavily pressured Mediterranean coastal ecosystems (Bramwell, 2004; Jiménez, 2012; Montes et al., 2011). Exhaustion of natural areas, isolation of existing PAs, and saturation of coastal destinations are likely outcomes of the continuation of recent trends (Jurado, Dantas, & da Silva, 2009; Martín-López, García-Llorente, Palomo, & Montes, 2011; Palomo, Martín-López, Potschin, Haines-Young, & Montes, 2013; Roca et al., 2008).

There was no consistent pattern of developmental pressure across proximal and distant buffers, in contrast to what was shown by Rodríguez-Rodríguez and Martínez-Vega (2018a). Proximal SPA and NR buffers on coastal zones experienced the greatest AAI values, which suggests strong pressures around these areas, many of them coastal wetlands (Capdepón, 2016). Inland, the greatest AAI occurred around NPs, many of them peri-urban sites in Spain (Rodríguez-Rodríguez & Martínez-Vega, 2018a).

Biophysical similarity between cases and controls assessed by S was high but lower than in other previous assessments (Rodríguez-Rodríguez & Martínez-Vega, 2018b), suggesting that improvements to validity and uncertainty of results could be attained. The lowest similarity between cases and controls was obtained for NRs, many of them wetland ecosystems in Spain, which poses increased difficulty at finding similar surrounding controls. Even though we tried to follow best practice at assessing the environmental effects of PAs in terms of consideration of relevant covariates such as additional territorial regulations, biophysical constraints to development, geographic location and climate, our covariate control technique used for the BEM did not render consistent results and should be improved. Further splitting PA samples according to relevant biophysical covariates might result in a simple, consistent, and highly valid case–control comparison option, as we sought here, but this point deserves further study.

5. CONCLUSIONS

PAs reduced land development differentially in Spain inland and on the coast in the 1987–2006 period. NRs were the most effective PA category in both zones, whereas Natura 2000 SPAs and SCIs were the least effective categories inland and on the coast, respectively. Even if reducing land development when compared with controls, the fact that all PA categories except NRs experienced some land development in the assessed period is worrisome to long-term biodiversity conservation in Spain. Land development was a greater pressure on the coast than inland in the assessed period. Land development figures on the Macaronesian and Mediterranean coasts were greater than those of the Atlantic coast, most likely as a result of more benign climate conditions attracting larger number of residents and visitors (Jiménez, 2007; INE, 2018). However, Atlantic coastal PAs were ineffective to reduce land development, with similar or greater AAI and RAI values than their controls. Greater proportion of single designation SCIs may help to explain this surprising and concerning finding. Thus, legal improvements in coastal Atlantic Pas and around coastal Macaronesian and Mediterranean PAs arise as priorities of this assessment. Biophysical similarity between cases and control was high but could be improved. A different covariate control technique should be used in future studies.

ACKNOWLEDGMENTS

This paper recognises contributions through the sequence-determines- credit approach. D. R. R. and J. S. contributed equally. D. R. R. was funded for this study by the Spanish Ministry of Economy, Industry and Competitiveness in the framework of the SOSTPARK project (CSO2014-54611-JIN).

ORCID

David Rodríguez-Rodríguez https://orcid.org/0000-0001-9662-4210
Ángel Enrique Salvo Tierra https://orcid.org/0000-0002-1464-9770
Javier Martínez-Vega https://orcid.org/0000-0002-8519-120X

REFERENCES

Andalusian Government. (2016). Consejería de Medio Ambiente y Ordenación del Territorio. Información Ambiental. Espacios Protegidos. Red de Espacios Naturales Protegidos de Andalucía. Sitios Ramsar. Humedales de Andalucía en la Lista Ramsar (Andalusian Ministry of Environment and Territorial Planning. Environmental Information. Protected Areas. Andalusian Protected Areas Network. Ramsar sites. Andalusian wetlands on the Ramsar list). Retrieved from: http://www.juntadeandalucia.es/medioambiente/site/portalweb/menuitem.7e1cf46ddf59bb227a9ebe205510e1ca/?vgnextoid=4efabfe08a7a5010VgnVCM1000000624e50aRCRD&vgnextchannel=a22ca0d0851f4310VgnVCM2000000624e50aRCRD

Andam, K. S., Ferraro, P. J., Pfaff, A., Sánchez-Azofeifa, G. A., & Robalino, J.A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. PNAS, 105(42), 16089–16094. https://doi.org/10.1073/pnas.0800437105

Araújo, M. B., Lobo, J. M., & Moreno, J. C. (2007). The effectiveness of Iberian protected areas in conserving terrestrial biodiversity. Conservation Biology, 21(6), 1423–1432. https://doi.org/10.1111/j.1523-1739.2007.00827.x

Atauri, J. A., Múgica, M., Gómez-Limón, J., & de Lucio, J. V. (2008). Procedimiento para la asignación de las categorías internacionales de manejo de áreas protegidas de la UICN. Madrid: Ed. Fundación Fernando González Bernáldez.

Barbero-Sierra, C., Marques, M. J., & Ruíz-Pérez, M. (2013). The case of urban sprawl in Spain as an active and irreversible driving force for desertification. Journal of Arid Environments, 90, 95–102. https://doi org/10.1016/j.jaridenv.2012.10.014

Bhola, N., Juffe-Bignoli, D., Burguess, N., Sandwith, T., & Kingston, N. (Eds.)(2016). Protected planet report 2016. How protected areas contribute to achieving global targets for biodiversity. Cambridge and Gland: UNEP-WCMC and IUCN.

Bramwell, B. (2004). Coastal mass tourism diversification and sustainabledevelopment in southern Europe (Vol. 12) (p. 1). Clevedon: Channel ViewPublications. https://doi.org/10.21832/9781873150702

Brotherton, I. (1982). Development pressures and control in the National Parks, 1966–1981. The Town Planning Review, 53(4), 439–459. https://www.jstor.org/stable/40111902?seq=1#page_scan_tab_contents. https://doi.org/10.3828/tpr.53.4.n6375742882p2576

Burak, S., Dogan, B., & Gazioglu, C. (2004). Impact of urbanization and tourism on coastal environment. Ocean and Coastal Management, 47,515–527. https://doi.org/10.1016/j.ocecoaman.2004.07.007

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Schalermann, J. P.W., Almond, R. E. A., … Watson, R. (2010). Global biodiversity: Indicators of recent declines. Science, 328(5982), 1164–1168. https://doi.org/10.1126/science.1187512

Capdepón, M. (2016). Conflictos ambientales derivados de la urbanización turístico-residencial. Un caso aplicado al litoral alicantino. Boletín de la Asociación de Geógrafos Españoles, 71, 31–57. https://doi.org/10.21138/bage.2273

CBD, Convention on Biological Diversity. (1992). Text of the convention. Retrieved from: https://www.cbd.int/convention/text/

Cha, S. H. (2007). Comprehensive survey on distance/similarity measures between probability density functions. International Journal of Mathematical Models and Methods in Applied Science, 4(1), 300–307. Retrieved from: http://users.uom.gr/~kouiruki/sung.pdf

Chica-Ruiz, J.A., Pérez-Cayeiro, M.L. & Barragán Muñoz, J. M. (2014). Aproximación a los Impulsores Directos de Cambio en la Evaluación de los Ecosistemas del Milenio del litoral de Andalucía. Retrieved from: http://rodin.uca.es/xmlui/bitstream/handle/10498/16864/OT%20%28Chica%29.pdf?sequence=1&isAllowed=y

Collet, I. (2010). Portrait of EU coastal regions. Eurostat 38/2010. Retrieved from http://edz.bib.uni-mannheim.de/www-edz/pdf/statinf/10/KS-SF-10-038-EN.PDF

Creel, L. (2003) Ripple effects: Population and coastal regions. Population Reference Bureau, 1–8. Retrieved from: http://152.46.13.240/ Moodle Content/HumanGeographyAP/Week_7/RippleEffects_Eng.pdf

Custodio, E., Andreu-Rodes, J. M., Aragón, R., Estrela, T., Ferrer, J., García-Aróstegui, J. L., … del Villar, A. (2016). Groundwater intensive use and mining in south-eastern peninsular Spain: Hydrogeological, economic and social aspects. Science of the Total Environment, 559, 302–316. https://doi.org/10.1016/j.scitotenv.2016.02.107

Dal, N., & Baysan, S. (2011). Land use alterations in Kusadasi coastal area. Procedia-Social and Behavioral Sciences, 19, 331–338. https://doi.org/10.1016/j.sbspro.2011.05.139

de Andrés, M., Barragán, J. M., & García Sanabria, J. (2017). Relationships between coastal urbanization and ecosystems in Spain. Cities, 68,8–17. https://doi.org/10.1016/j.cities.2017.05.004

Dudley, N. (Ed.) (2008). Guidelines for applying protected area management categories. Gland: IUCN. https://doi.org/10.2305/IUCN.CH.2008. PAPS.2.en

EEA, European Environment Agency. (1995). Publications. CORINE Land Cover—Part 2: Nomenclature. Available online from: http://www.eea.europa.eu/publications/COR0-part2 [Accessed 26/01/2017]

EEA, European Environment Agency. (2002). Publications. Europe's biodiversity—Biogeographical regions and seas. Retrieved from: https://www.eea.europa.eu/publications/report_2002_0524_154909

EEA, European Environment Agency. (2008). Europe's biodiversity—Biogeographical regions and seas. Biogeographical regions in Europe. Retrieved from: https://www.eea.europa.eu/publications/report_2002_0524_154909/biogeographical-regions in-europe

EEA, European Environment Agency. (2011). Landscape fragmentation in Europe. A joint EEA-FOEN report. EEA report no 2/2011. Luxembourg: Publications Office of the European Union.

EEA, European Environment Agency. (2015). Data and maps. Datasets. Biogeographical regions. Retrieved from: http://www.eea.europa.eu/data-and-maps/data/biogeographical-regions-europe

EEA, European Environment Agency. (2018). Data and maps. Indicators. Land take. Retrieved from: https://www.eea.europa.eu/data-and-maps/indicators/land-take-2

EEC, European Economic Community. (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal L., 206, 0007–0050. Retrieved from: http://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX%3A31992L0043

ESRI, Environmental Systems Resource Institute. (2014). ArcInfo, version 10.3. Redlands, California: ESRI.

Falco, E. (2017). Protection of coastal areas in Italy: Where do national landscape and urban planning legislation fail? Land Use Policy, 66,80–89. https://doi.org/10.1016/j.landusepol.2017.04.038

García-Ayllón, S. (2015). La Manga case study: Consequences from short-term urban planning in a tourism mass destiny of the Spanish Mediterranean coast. Cities, 43, 141–151. https://doi.org/10.1016/j.cities.2014.12.001

Greenpeace. (2018). Costas. Retrieved from https://es.greenpeace.org/es/trabajamos-en/oceanos/costas/

Hadwen, W. L., Arthington, A. H., Boon, P. I., Taylor, B., & Fellows, C. S.(2011). Do climatic or institutional factors drive seasonal patterns of tourism visitation to protected areas across diverse climate zones in eastern Australia? Tourism Geographies, 13(2), 187–208. https://doi.org/10.1080/14616688.2011.569568

Hein, L., Metzger, J., & Moreno, A. (2009). Potential impacts of climate change on tourism; a case study for Spain. Current Opinion in Environmental Sustainability, 1(2), 170–178. https://doi.org/10.1016/j.cosust.2009.10.011

Hewitt, R., Pera, F., & Escobar, F. (2016). Cambios recientes en la ocupación del suelo de los parques nacionales españoles y su entorno. Cuadernos Geográficos, 55(2), 46–84. https://doi.org/10.30827/cuadgeo.v55i2.3130

Iglesias-Campos, A., Meiner, A., Bowen, K., & Onwona Ansong, J. (2015).Coastal population and land use changes in Europe: Challenges for a sustainable future. In J. Baztan, O. Chouinard, B. Jorgensen, et al.(Eds.), Coastal zones, solutions for the 21st century (pp. 29–49). Ontario, Canada: Elsevier.

Retrieved from: https://www.sciencedirect.com/book/9780128027486/coastal-zones

IGN, Instituto Geográfico Nacional. (2011). Centro de descargas. Centro Nacional de Información geográfica. Catálogo de productos. Base Cartográfica Nacional 1:200.000. BCN 1:200. Versión 4.0. Available online from: http://centrodedescargas.cnig.es/CentroDescargas/catalogo.do;jsessionid=76B6BFF30E1DBAB4B3993F1F95C28954#selectedSerie [Accessed 21/03/2017]

IGN, Instituto Geográfico Nacional. (2012a). Centro de descargas. Centro Nacional de Información geográfica. Catálogo de productos. Base Cartográfica Nacional 1:500.000. BCN 1:500. Versión 2.3. Available online from: http://centrodedescargas.cnig.es/CentroDescargas/catalogo.do;jsessionid=76B6BFF30E1DBAB4B3993F1F95C28954#selectedSerie [Accessed 21/03/2017]

IGN, Instituto Geográfico Nacional. (2012b). Centro de descargas. Centro Nacional de Información geográfica. Catálogo de productos. MDT200. Available online from: http://centrodedescargas.cnig.es/CentroDescargas/catalogo.do;jsessionid=76B6BFF30E1DBAB4B3993F1F95C28954#selectedSerie [Accessed 21/03/2017]

IGN, Instituto Geográfico Nacional. (2015). Servicio de descarga Inspire de acceso directo conforme con ISO 19142 Web Feature Service e ISO 19143 Filter Encoding. Base de datos de límites jurisdiccionales de España. Retrieved from: http://www.ign.es/wfs-inspire/unidades-administrativas [Accessed 15/03/2017]

IGN, Instituto Geográfico Nacional. (2016). Centro Nacional de Información Geográfica. Centro de Descargas. Catálogo de productos. Corine Land Cover. Retrieved from: http://centrodedescargas.cnig.es/CentroDescargas/buscar.do

INE, Instituto Nacional de Estadística. (2016). Censos de población y viviendas 1991. Resultados definitivos. Municipios. Available online from:http://www.ine.es/censo91/es/seleccion_municipio.jsp [Accessed 14/03/2017]

INE, Instituto Nacional de Estadística (2017). INEBase. Encuesta de ocupación hotelera. Retrieved from. http://www.ine.es/dynt3/inebase/es/index.htm?padre=238&dh=1

INE, Instituto Nacional de Estadística. (2018). Movimientos turísticos en fronteras. Resultados nacionales. Retrieved from: http://www.ine.es/jaxiT3/Datos.htm?t=23982

Jiménez, L., Prieto, F., Riechmann, J., & Gómez, A. (coord.) (2005).Sostenibilidad en España 2005. Informe de primavera. Alcalá de Henares:Observatorio de la Sostenibilidad en España.

Jiménez, L. M. (dir.).(2007). Sostenibilidad en España, 2007. Alcalá de Henares: Observatorio de la Sostenibilidad en España.

Jiménez, L. M. (dir.).(2010). Sostenibilidad en España 2010. Alcalá de Henares: Observatorio de la Sostenibilidad en España.

Jiménez, L. M. (dir.).(2012). Sostenibilidad en España 2012. Capítulo especial energía sostenible para todos (2012 Año Internacional de la Energía). Madrid: Ministerio de Agricultura, Alimentación y Medio Ambiente.

Juffe-Bignoli, D., Burguess, N. D., Bingham, H., Belle, E. M. S., de Lima, M.G., Deguignet, M., Kingston, N. (2014). Protected planet report 2014.Tracking progress towards global targets for protected areas. Cambridge:UNEP-WCMC. Retrieved from: https://www.unep-wcmc.org/system/dataset_file_fields/files/000/000/289/original/Protected_Planet_Report_2014_01122014_EN_web.pdf?1420549522

Jurado, E., Dantas, A. G., & da Silva, C. P. (2009). Coastal zone management:Tools for establishing a set of indicators to assess beach carrying capacity (Costa del Sol - Spain). Journal of Coastal Research, SI, 56, 1125–1129. Retrieved from: http://www.cerf-jcr.org/images/stories/1125.1129_E.Jurado_ICS2009.pdf

Khamis, Z. A., Kalliola, R., & Käyhkö, N. (2017). Geographical characterization of the Zanzibar coastal zone and its management perspectives. Ocean and Coastal Management, 149, 116–134. https://doi.org/10.1016/j.ocecoaman.2017.10.003

MAGRAMA, Ministerio de Agricultura, Alimentación y Medio Ambiente.2015. Cartografía y SIG. Infraestructura de Datos Espaciales-IDE.Descargas. Biodiversidad. Available online from: http://www.mapama.gob.es/es/cartografia-y-sig/ide/descargas/biodiversidad/default.aspx [Accessed 14/03/2017]

Martínez-Fernández, J., Ruiz-Benito, P., & Zavala, M. A. (2015). Recent land cover changes in Spain across biogeographical regions and protection levels: Implications for conservation policies. Land Use Policy, 44,62–75. https://doi.org/10.1016/j.landusepol.2014.11.021

Martín-López, B., García-Llorente, M., Palomo, I., & Montes, C. (2011). The conservation against development paradigm in protected areas: Valuation of ecosystem services in the Doñana social–ecological system (southwestern Spain). Ecological Economics, 70, 1481–1491. https://doi.org/10.1016/j.ecolecon.2011.03.009

Mas, J. F. (2005). Assessing protected areas effectiveness using surrounding (buffer) areas environmentally similar to the target area.Environmental Monitoring and Assessment, 105, 69–80. https://doi.org/10.1007/s10661-005-3156-5

McDonald, R. I., Kareiva, P., & Forman, R. T. T. (2008). The implications of current and future urbanization

for global protected areas and biodiversity conservation. Biological Conservation, 141, 1695–1703.https://doi.org/10.1016/j.biocon.2008.04.025
McKinney, M. L. (2002). Urbanization, biodiversity and conservation. Bioscience, 52(10), 883–890.https://doi.org/10.1641/0006-3568(2002)052[0883:UBAC]2.0.CO;2
Médail, F., & Quézel, P. (1999). Biodiversity hotspots in the Mediterranean Basin: Setting global conservation priorities. Conservation Biology, 13(6), 1510–1513. https://doi.org/10.1046/j.1523-1739.1999.98467.x
Montes, C., Santos, F., & Benayas, J. (coord.) (2011). Ecosistemas y biodiversidad para el bienestar humano. Evaluación de los Ecosistemas del Milenio en España. Síntesis de resultados. Madrid: Fundación Biodiversidad.
Najafinasab, F., Karbassi, A. B., & Ghoddousi, J. (2015). Fuzzy analytic network process approach to evaluate land and sea criteria for land use planning in coastal areas. Ocean and Coastal Management, 116,368–381. https://doi.org/10.1016/j.ocecoaman.2015.07.023
Palomo, I., Martín-López, B., Potschin, M., Haines-Young, R., & Montes, C.(2013). National Parks, buffer zones and surrounding landscape: Mapping ecosystem services flows. Ecosystem Services, 4, 104–116. https://doi.org/10.1016/j.ecoser.2012.09.001
Pan, Y., Zhai, M., Lin, L., Lin, Y., Cai, J., Deng, J. S., & Wang, K. (2016). Characterizing the spatiotemporal evolutions and impact of rapid urbanization on island sustainable development. Habitat International, 53, 215–227. https://doi.org/10.1016/j.habitatint.2015.11.030
Pons, A., & Rullán, O. (2014). Artificialization and islandness on the Spanish tourist coast. Miscellanea Geographica-Regional Studies on Development,18(1), 5–16. https://doi.org/10.2478/mgrsd-2014-0010
Prieto, F. (Coord.). (2016). Sostenibilidad en España 2016. SOS.Observatorio de la Sostenibilidad. Available online from: http://www.observatoriosostenibilidad.com/
QGIS. (2017). For users. Download QGIS. Retrieved from: https://www.qgis.org/en/site/forusers/index.html
Radeloff, V. C., Stewart, S. I., Hawbaker, T. J., Gimmi, U., Pidgeon, A. M., Flather, C. H., … Helmers, D. P. (2010). Housing growth in and near United States protected areas limits their conservation value. PNAS, 107(2), 940–945. https://doi.org/10.1073/pnas.0911131107
Rey Benayas, J.M., Martins, A., Nicolau, J.M., & Schulz, J.J. (2007). Abandonment of agricultural land: An overview of drivers and consequences. CAB Reviews Perspectives in Agriculture Veterinary Science Nutrition and Natural Resources, 2, 057, https://doi.org/10.1079/PAVSNNR20072057
Rico-Amorós, A. M., Olcina-Cantos, J., & Sauri, D. (2009). Tourist land use patterns and water demand: Evidence from the Western Mediterranean. Land Use Policy, 26(2), 493–501. https://doi.org/10.1016/j. landusepol.2008.07.002
Roca, E., Riera, C., Villares, M., Fragell, R., & Junyent, R. (2008). A combined assessment of beach occupancy and public perceptions of beach quality:A case study in the Costa Brava, Spain. Ocean and Coastal Management, 51(12), 839–846. https://doi.org/10.1016/j. ocecoaman.2008.08.005
Rodríguez-Rodríguez, D. (2012). Integrated networks. A territorial planning proposal for biodiversity conservation in urban, densely populated regions. The case of the Autonomous Region of Madrid, Spain. Journal of Environmental Planning and Management, 55(5), 667–683. https://doi.org/10.1080/09640568.2011.620391
Rodríguez-Rodríguez, D., & Martínez-Vega, J. (2017). Assessing recent environmental sustainability in the Spanish network of National Parks and their statutory peripheral areas. Applied Geography, 89, 22–31. https://doi.org/10.1016/j.apgeog.2017.09.008
Rodríguez-Rodríguez, D., & Martínez-Vega, J. (2018a). Protected area effectiveness against land development in Spain. Journal of Environmen-tal Management, 215, 345–357. https://doi.org/10.1016/j.jenvman.2018.03.011
Rodríguez-Rodríguez, D., & Martínez-Vega, J. (2018b). Effect of legal protection and management of protected areas at preventing land development: A Spanish case study. Regional Environmental Change, 18, 2483–2494. https://doi.org/10.1007/s10113-018-1369-8
Rodríguez-Rodríguez, D., Rodríguez, J., & Abdul Malak, D. (2016).Development and testing of a new framework for rapidly assessing legal and managerial protection afforded by marine protected areas: Mediterranean Sea case study. Journal of Environmental Management, 167, 29–37. https://doi.org/10.1016/j.jenvman.2015.11.016
Rodríguez-Rodríguez, D., Rodríguez, J., Blanco, J. M., & Abdul Malak, D.(2016). Marine protected area design patterns in the Mediterranean Sea: Implications for conservation. Marine Pollution Bulletin, 110(1), 335–342. https://doi.org/10.1016/j.marpolbul.2016.06.044
Romano, R. (2011). Potencial del suelo artificializado en la costa Catalana. Centre de Política de Sòl i Valoracions, Universitat Politècnica de Catalunya. Retrieved from: https://upcommons.upc.edu/handle/2117/15457
Santamarta, J. C., Naranjo, J., & Arraiza, M. P. (2014). Challenges for future of natural spaces of Canary Islands, Spain. IERI Procedia, 8, 176–181 https://doi.org/10.1016/j.ieri.2014.09.029
Seto, K. C., Guneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools.PNAS, 109(40), 16083–16088. https://doi.org/10.1073/pnas.1211658109
Smith, E. P. (2002). BACI design. In A. H. El-Shaarawi, & W. W. Piegorsch (Eds.), Encyclopedia of

environmetrics (Vol. 1). Vol. (pp. 141–148). Chichester: John Wiley & Sons.
Spanish Government. (1988). Ley 22/1988, de 28 de Julio, de Costas. Boletín Oficial del Estado, 181, 23386–23401. Retrieved from: https://www.boe.es/boe/dias/1988/07/29/pdfs/A23386-23401.pdf
Spanish Government. (1989). Ley 4/1989, de 27 de marzo, de Conservación de los Espacios Naturales y de la Flora y Fauna Silvestres. Boletín Oficial del Estado, 74, 8262–8269. Retrieved from: https://www.boe.es/buscar/doc.php?id=BOE-A-1989-6881
Spanish Government. (2001). Real Decreto Legislativo 1/2001, de 20 dejulio,por el queseaprueba el textorefundido de la Leyde Aguas. Boletín Oficial del Estado, 176, 26791–26817. Retrieved from: https://www.boe.es/boe/dias/2001/07/24/pdfs/A26791-26817.pdf
Spanish Government. (2007). Ley 42/2007, de 13 de diciembre, del Patrimonio Natural y de la Biodiversidad. Boletín Oficial del Estado, 299, 51275–51327. Retrieved from: https://www.boe.es/boe/dias/2007/12/14/pdfs/A51275-51327.pdf
Spanish Government. (2008). Atlas Nacional de España. El Medio Terrestre.Biogeografía, flora y fauna. Espacios naturales protegidos. Retrieved from: http://www.ign.es/ane/ane1986-2008/
Spanish Government. (2015). Cartografía y SIG. Infraestructura de Datos Espaciales-IDE. Descargas. Biodiversidad. 2015. Retrieved from: http://www.mapama.gob.es/es/cartografia-y-sig/ide/descargas/biodiversidad/default.aspx
Stellmes, M., Röder, A., Udelhoven, T., & Hill, J. (2013). Mapping syndromes of land change in Spain with remote sensing time series, demographic and climatic data. Land Use Policy, 30(1), 685–702. https://doi.org/10.1016/j.landusepol.2012.05.007
Tan, K. W. (2006). A greenway network for Singapore. Landscape and Urban Planning, 76, 45–66. https://doi.org/10.1016/j.landurbplan.2004.09.040
Torres, A., Jaeger, J. A. G., & Alonso, J. C. (2016). Assessing large-scale wildlife responses to human infrastructure development. PNAS, 113(30), 8472–8477. https://doi.org/10.1073/pnas.1522488113
Tourspain. (2018). Tourism statistics. Retrieved from: http://estadisticas.tourspain.es/en-EN/turismobase/Paginas/default.aspx
UNCCD, United Nations Convention to Combat Desertification. (1994).Text of the convention. Retrieved from: https://www.unccd.int/sites/default/files/relevant-links/2017-01/UNCCD_Convention_ENG_0.pdf
Vera, J. F., & Ivars, J. A. (2009). Spread of low-cost carriers: Tourism and regional policy effects in Spain. Regional Studies, 43(4), 559–570. https://doi.org/10.1080/00343400701874164
Wade, A. A., & Theobald, D. M. (2010). Residential encroachment on U.S.protected areas. Conservation Biology, 24(1), 151–161. https://doi.org/10.1111/j.1523-1739.2009.01296.x
WTTC, World Travel and Tourism Council. (2017). Travel & Tourism Economic Impact 2017 Spain. Retrieved from: https://www.wttc.org/-/media/files/reports/economic-impact-research/countries-2017/spain2017.pdf
WWF, World Wildlife Fund. (2016). Living planet report 2016. Risk and resilience in a new era. Gland: WWF International. Retrieved from: https://wwf.panda.org/knowledge_hub/all_publications/lpr_2016/
Zhang, K. H., & Song, S. (2003). Rural–urban migration and urbanization in China: Evidence from time-series and cross-section analyses. China Economic Review, 14, 386–400. https://doi.org/10.1016/j.chieco.2003.09.018

APPENDIX A

Legal exclusion layer

The exclusion layer used to exclude areas where land development if legally restricted in Spain from the spatial analysis was created by merging the following partial digital layers:

1. Coastal public domain: A 100-m inland buffer from the Spanish coastline (Instituto Geográfico Nacional [IGN], 2015) was created.
2. Public hydraulic domain: A 100-m buffer along both sides of main rivers and around dams, lakes, and lagoons was created from the National Cartographic Base 1:500,000 (IGN, 2012a) by selecting classes 01, 02, 03 (over 25 km long, level 1, 2, or 3 rivers), 06,07, 08, or 09 (dams, lakes, and lagoons).
3. All existing protected areas (PAs) designated until December 31,2006, including biosphere reserves (MAGRAMA, 2015). This date was selected to exclude all possible effect of PA coverage, whichever their designation category, on buffers until the assessment date. This layer was only excluded from buffer areas, not from cases.

Resulting protected polygons (PPs) and buffer polygons smaller than 100 ha were deleted from each case and buffer layer to avoid alignment and overlap errors. Other important areas where land development is limited by territorial legislation such as cattle pathways or public utility forest could not be added to the legal exclusion layer because of data incompleteness or inconsistency for the whole country.

APPENDIX B

Biophysical similarity analysis

We analysed the environmental characteristics of each PA and PP network for five biophysical covariates that are thought to influence land use–land cover change: ‘degree of initial treeless cover,’ ‘altitude,’slope,’ ‘distance to main cities,’ and ‘distance to main transportinfrastructures.’

The degree of initial treeless cover layer was produced by merging the following CORINE Land Cover-1990 subclasses that are completely or mostly treeless: 211, 212, 213, 221, 222, 231, 241,242, 243, 321, 322, 331, 332, 333, and 334 (European Environment Agency, 1995). Then, the percentage of each PP's area covered by those subclasses was computed. The altitude layer was created by interpolating altitude isolines for Spain (IGN, 2011) to raster format at 1-km^2 resolution to create a 31-class, 100-m discrete altitude model for the whole terrestrial territory of the Country. A discrete, 1-km^2 resolution raster slope map (in degrees) was created from a digital elevation model for Spain (IGN, 2012b). Distances of PPs' boundaries and buffers' boundaries to main transport infrastructures (CORINE Land Cover-1990 subclass 122) and to major cities in 1991 were calculated using the Near function in Arc-GIS (Environmental Systems Resource Institute, 2014). For this, the main infrastructure layer was decomposed to their forming points. Centroids of the cities having more than 50,000 inhabitants according to the 1991 national population census (Instituto Nacional de Estadística, 2016) were calculated from the Spanish main city polygon areas (IGN, 2012a) using the ‘Feature to point’ (inside unchecked) Arc-GIS tool (Environmental Systems Resource Institute, 2014).

Each of the five biophysical layers (distance to main cities, altitude, slope, initial degree of artificial covers, and initial degree of treeless covers) was intersected with the PP layer and with their respective 1- and 5-km buffer areas. For each PA and PP network, an analysis of the percentage of the total network area represented by each rank of each of the covariates was calculated in each study zone (e.g., the area of the nature reserve network covered by each 100 m, discrete altitude range inland and on the coast). Then, buffer area values outside the range of values of each covariate for each specific PA (PP or buffer) network were excluded for not being representative of that network in each study zone. Then, those values were excluded for the same PA networks between study zones (i.e., different range values of altitude between nature reserve cases and controls inland and on the coast), in order to increase validity of within-zone and interzone comparisons. In the cases of distance to main cities and degree of initial treeless cover, entire polygon buffers were excluded if meeting any exclusion criterion. In the cases of altitude and slope, average (mean) values were computed for each individual PP and control and excluded if they were outside the cases'ranges. All five partial environmental exclusion layers (for PPs and controls) were merged in a ‘biophysical exclusion layer,’ which was later on extracted from the PP, 1- and 5-km buffer layers to create control areas that were environmentally similar to their respective networks. Finally, resulting PPs and buffer polygons smaller than 100 ha were deleted from each case and buffer layer to avoid alignment and overlap errors and produce results for the biophysically similar model.

APPENDIX C

Biophysical similarity results (median covariate values)

Original model

Covariate	NRs	NR-1km	NR-5km	***S* (NRs-1km buffer)**	***S* (NRs-5km buffer)**	NPs	NPs-1km	NPs-5km	***S* (NPs-1km buffer)**	***S* (NPs-5km buffer)**	SCIs	SCI-1km	SCI-5km	***S* (SCIs-1km buffer)**	***S* (SCIs-5km buffer)**	SPAs	SPA-1km	SPA-5km	***S* (SPAs-1km buffer)**	***S* (SPAs-5km buffer)**
INLAND Zone Similarity																				
Altitude (m)	1100	800	800	0.90	0.90	1200	900	1000	0.90	0.94	1100	900	900	0.94	0.94	1000	800	800	0.94	0.94
Slope (º)	6.50	11.50	8.00	0.94	0.98	7.00	5.00	4.00	0.98	0.97	5.00	4.00	4.00	0.99	0.99	6.00	3.00	4.00	0.97	0.98
Treeless cover (%)	25.88	87.47	81.91	0.38	0.44	9.10	37.77	55.18	0.71	0.54	24.46	53.73	64.89	0.71	0.60	34.35	65.31	64.77	0.69	0.70
Distance to populations (km)	72.31	79.36	77.25	0.44	0.46	18.45	30.20	42.53	0.94	0.89	63.96	64.94	67.93	1.00	0.98	36.663	46.050	43.262	0.96	0.97
Distance to infrastructures (km)	3.89	3.88	4.14	1.00	0.98	1.52	4.55	1.44	0.93	1.00	2.51	4.64	3.69	0.96	0.98	1.28	4.13	3.09	0.90	0.94
Global Similarity Index				**0.83**	**0.75**				**0.89**	**0.87**				**0.92**	**0.90**				**0.89**	**0.90**
COASTAL Zone Similarity																				
Altitude (m)	0	0	0	1.00	1.00	200	100	100	0.97	0.97	300	200	300	0.97	1.00	500	200	200	0.90	0.90
Slope (º)	0.00	0.00	0.00	1.00	1.00	3.00	3.00	3.00	1.00	1.00	5.00	4.00	4.00	0.99	0.99	6.00	3.00	4.00	0.97	0.98
Treeless cover (%)	4.19	90.45	90.88	0.14	0.13	17.32	63.61	71.82	0.54	0.46	17.37	52.85	58.30	0.65	0.59	39.55	75.24	72.66	0.64	0.67
Distance to populations (km)	27.79	28.24	25.35	0.52	0.49	30.33	35.52	36.67	0.98	0.97	31.93	29.68	27.55	0.99	0.98	29.525	32.237	26.631	0.99	0.99
Distance to infrastructures (km)	9.14	9.18	5.19	1.00	0.63	1.44	2.91	2.02	0.97	0.99	1.66	1.69	0.91	1.00	0.99	3.52	5.91	3.05	0.92	0.98
Global Similarity Index				**0.83**	**0.65**				**0.89**	**0.88**				**0.92**	**0.91**				**0.88**	**0.90**

Note: NRs: nature reserves; NPs: nature parks; SCIs: sites community importance; SPAs: special protection areas; S: similarity index

Biophysically enhanced model

Covariate	NRs	NR-1km	NR-5km	***S* (NRs-1km buffer)**	***S* (NRs-5km buffer)**	NPs	NPs-1km	NPs-5km	***S* (NPs-1km buffer)**	***S* (NPs-5km buffer)**	SCIs	SCI-1km	SCI-5km	***S* (SCIs-1km buffer)**	***S* (SCIs-5km buffer)**	SPAs	SPA-1km	SPA-5km	***S* (SPAs-1km buffer)**	***S* (SPAs-5km buffer)**
INLAND Zone Similarity																				
Slope (º)	6.50	11.00	8.00	0.95	0.98	7.00	5.00	4.00	0.98	0.97	5.00	4.00	4.00	0.99	0.99	5.00	3.00	4.00	0.98	0.99
Treeless cover (%)	14.16	12.24	29.48	0.98	0.85	32.00	33.58	44.52	0.98	0.87	24.46	53.73	64.89	0.71	0.60	34.35	65.31	47.57	0.69	0.87
Distance to populations (km)	6.51	79.36	74.12	0.85	0.90	18.45	29.23	40.26	0.95	0.90	63.96	64.94	67.89	1.00	0.96	36.66	45.55	39.55	0.96	0.99
Distance to infrastructures (km)	3.89	3.09	3.42	0.93	0.96	1.52	4.42	1.43	0.94	1.00	2.51	4.63	3.69	0.96	0.99	1.28	4.00	3.05	0.91	0.94
Global Similarity Index				**0.93**	**0.92**				**0.96**	**0.93**				**0.91**	**1.00**				**0.88**	**0.95**
COASTAL Zone Similarity																				
Slope (º)	0.00	0.00	0.00	1.00	1.00	3.00	3.00	3.00	0.98	0.98	5.00	4.00	4.00	0.99	0.99	6.00	3.00	4.00	0.97	0.98
Treeless cover (%)	4.19	90.45	90.88	0.14	0.13	17.32	63.61	71.82	0.54	0.46	17.37	52.85	58.30	0.65	0.59	29.525	44.43	45.29	0.85	0.84
Distance to populations (km)	27.79	13.41	25.35	0.36	0.49	24.66	35.52	30.41	0.95	0.97	31.07	29.29	36.67	0.99	0.98	22.92	32.11	26.63	0.95	0.98
Distance to infrastructures (km)	9.14	8.14	5.19	0.91	0.64	1.44	2.87	1.66	0.97	1.00	1.66	1.69	0.90	1.00	0.99	3.52	4.56	2.82	0.97	0.98
Global Similarity Index				**0.72**	**0.57**				**0.86**	**0.85**				**0.91**	**0.89**				**0.93**	**0.94**

Note: NRs: nature reserves; NPs: nature parks; SCIs: sites community importance; SPAs: special protection areas; S: similarity index

APPENDIX D

Land development results of the biophysically enhanced model

Protected Area Network	AAI	RAI	Total network area (ha)	New artificial area (ha)	1km AAI	1km RAI	Total 1km buffer area (ha)	New artificial area in 1km buffer (ha)	5km AAI	5km RAI	Total 5km buffer area (ha)	New artificial area in 5km buffer (ha)
INLAND ZONE												
NRs	0	0	8,753	0	0	0	0	0	0	0	0	0
NPs	0.02	13.49	474,291	85	0.14	21.08	21,578	31	0.16	18.48	35,044	55
SCIs	0.04	33.36	2,771,585	1,201	0.24	39.04	1,021,831	2,466	0.43	63.45	4,175,398	17,821
SPAs	0.26	83.57	1,550,345	4,099	0.22	36.29	324,111	731	0.86	84.59	1,386,349	11,876
ALL PAs	**0.11**	**58.93**	4,804,973	5,385	**0.24**	**38.08**	1,367,520	3,228	**0.53**	**70.13**	5,596,791	29,753
COASTAL ZONE												
NRs	**0**	**0**	**0**	**0**	**0**	**0**	0	0	0	**0**	0	0
NPs	0	0	23,588	0	0.20	37.63	5,554	11	0	0	0	0
SCIs	0.16	23.65	212,912	344	1.15	54.89	111,881	1,283	0.88	35.20	424,092	3,750
SPAs	0.04	11.60	135,802	66	1.54	57.53	16,589	256	1.24	61.26	73,122	906
ALL PAs	**0.11**	**19.40**	373,460	411	**1.16**	**55.13**	134,024	1,550	**0.94**	**38.38**	497,214	4,655

Note: AAI: absolute artificial area increase; RAI: relative artificial area increase; NRs: nature reserves; NPs: nature parks; SCIs: sites community importance; SPAs: special protection areas; PAs: protected areas.