Incorporating the Social–Ecological Approach in Protected Areas in the Anthropocene

IGNACIO PALOMO, CARLOS MONTES, BERTA MARTÍN-LÓPEZ, JOSÉ A. GONZÁLEZ, MARINA GARCÍA-LLORENTE, PALOMA ALCORLO, AND MARÍA ROSARIO GARCÍA MORA

Protected areas are the main strategy for the protection of ecosystems and biodiversity. However, changes occurring during the Anthropocene continue to threaten biodiversity and, therefore, the associated ecosystem services that maintain human well-being. Despite efforts to integrate protected areas into a wider landscape, most of these areas are still managed as islands within a matrix of degraded territory; there is no clear conceptual framework that integrates them into the surrounding landscape. We first review the evolution of the protected-area *concept. Then, we acknowledge the main limitations that protected areas face for long-term conservation. Next, we discuss how the ecosystem service approach could overcome some of these protected-area limitations. Finally, we propose a social–ecological approach for protected areas to maintain high biodiversity and its associated flow of ecosystem services in the context of uncertainty.*

Keywords: protected areas, landscape ecology, biodiversity, complex systems, land-use management

Protected areas are the most widely known and well-accepted strategy for conserving biodiversity in the face of the ecosystem fragmentation (Chape 2005). Since the creation of Yellowstone National Park in 1872, and especially in the second half of the twentieth century, the extent of protected areas has grown exponentially and has been associated with the great acceleration of the Anthropocene (Steffen et al. 2011). Protected areas have proven successful in protecting the habitats of the species inside them from land-use changes occurring outside (Andam et al. 2008) and have silently achieved one of the biggest global surface areas dedicated to a common goal (after areas used for food production). With the goal of preserving global biodiversity, in 2004, the Program of Work on Protected Areas of the Convention on Biological Diversity (CBD) advocated that, by 2010, at least 10% of each of the world's ecological regions should be effectively conserved. Today, protected areas cover 12.7% of global land and 1.6% of the world's oceans (Bertzky et al. 2012).

Nonetheless, biodiversity continues to decline (Butchart et al. 2010), and the capacity of ecosystems to maintain ecosystem services and human well-being over the long term is being undermined. The biodiversity 2010 target, which was intended "to achieve a significant reduction of the current rate of biodiversity loss," has not been accomplished (CBD 2010), and predictive models indicate that biodiversity will continue to decline during the twenty-first century. Therefore, the global protected-area network is far from fully achieving its main objective—that is, preserving biodiversity and ecosystem services (Rodrigues et al. 2004). Under this premise, the CBD has demanded that 17% of the global land surface and 10% of the seas be designated as protected areas by 2020 (CBD 2010), and scenarios suggest that protected areas could increase more in the next 20 years than they did in the previous 20, reaching 15%–29% of global land by 2030 (McDonald and Boucher 2011). Here, a question arises: How much land must we set apart to protect ecosystems, biodiversity, and the related ecosystem services for the long term?

Along with increasing protected area coverage, we need to revise the current model of protected areas to address the challenges that global change, characteristic of the Anthropocene, imposes on them (Lovejoy 2006). Moreover, several authors have indicated that conservation science should also address human well-being, a demand that protected areas could also incorporate (Kareiva and Marvier 2012). Our goal in the present study is to analyze the limitations of current protected-area strategies and to explore how the operationalization of one of the most important changes in conservation science in the last years—the emergence of the ecosystem services approach—could contribute to solutions (Armsworth et al. 2007). First, we analyze the historical evolution of the concept of *protected areas*. Second, we provide a review of the major acknowledged shortcomings of protected areas. Third, we argue

BioScience 64: 181–191.
doi:10.1093/biosci/bit033

Advance Access publication 22 January 2014

Table 1. Evolution of the protected-area concept: From islands to networks to landscapes to the social–ecological approach.

	Approach to protected areas			
Attributes	**Island approach (ca. 1872–1980s)**	**Network approach (1990s–mid-2000s)**	**Landscape approach (mid-2000s–today)**	**Social–ecological approach (today–?)**
Type of management	Static: Seeks to maintain the status quo	Dynamic: Some natural changes are considered necessary	Dynamic: Some natural changes are considered necessary	Adaptive: Natural and social changes should be incorporated into management
Conservation values considered	Intrinsic values of ecosystems, biodiversity and cultural values	Intrinsic values of ecosystems, biodiversity and cultural values	Intrinsic values of ecosystems, biodiversity, ecological processes (functions, ecological integrity), and cultural values	Intrinsic and instrumental values of ecosystems and biodiversity (ecosystem services)
Knowledge involved	Scientific and technical	Scientific and technical	Scientific and technical	Scientific, technical, and local ecological knowledge
Resilience against perturbations	Reduction of variability	Moderate resilience	Moderate–high resilience	High resilience
Competition against other land uses	Partly competitive because of low demand for the landscape	Competitive because of high demand for the landscape	Highly competitive because of the high demand for the landscape	Cooperative: Multifunctional landscapes
Local population involvement	Managed without the local population, which is seen as a threat; managed by researchers and environmental experts	The local population is included in some participatory management processes	The local population is included in participatory management processes	Truly managed with the local population
Landscape management	No integrated landscape management	No integrated landscape management	No integrated landscape management; the landscape is managed to avoid harming the protected area	Integrated landscape management; management of the landscape as a whole

that incorporating the ecosystem service framework into protected-area management could overcome many of the challenges that those areas face and propose a social–ecological approach for managing protected areas and the landscapes that surround them. Finally, we discuss the limitations of the approach.

The historical evolution of conservation strategies in protected areas for biodiversity conservation

In this section, we review the evolution of the *protected-area* concept, focusing on the social–ecological relationships between protected areas and their surrounding landscapes. Although the present study describes how the different conceptualizations of protected areas are often regarded as incomplete, the achievements of these conceptualizations have been of great importance, and they still have an essential role in the future of conservation. Although we provide durations for each period, these are only approximations, because periods overlap in some regions, and some regions need more time to evolve between periods than do others. Table 1 summarizes the main characteristics of each of the major conceptualizations of protected areas, which are illustrated graphically in figure 1.

Protected areas as islands (ca. 1872–1980s). The first protected areas were established under the paradigm that conservation is possible by setting apart certain areas from land-use transformation, and these areas were created after observing the extent of human impacts on natural ecosystems (Chape et al. 2005). The concept of protected areas as *islands* originated in 1872, with Yellowstone National Park, and is still found in current conservation conventions. For example, the CBD's definition of a *protected area*—"a geographically defined area [that] is designated or regulated and managed to achieve specific conservation objectives" (CBD 1992, art. 2)—does not include the territorial matrix or landscape dimension of protected areas. The reason for the creation of protected areas was to preserve the great values of nature from human transformation, and the main beneficiaries of these protected areas were usually conservationists, scientists, and nature tourists (Phillips 2003). In the second half of the twentieth century, protected areas shifted from conserving landscapes to conserving species and their habitats. Protected areas were designated mainly by biologists or environmental nongovernmental organizations with extensive knowledge about biodiversity and ecology but less experience with landscape planning. Transformations in the areas around protected areas made clear that the protected areas were partially threatened by their isolation and lack of connectivity with other protected areas (Janzen 1983). In the late 1970s, the United Nations Educational, Scientific, and Cultural Organization developed the concept of a

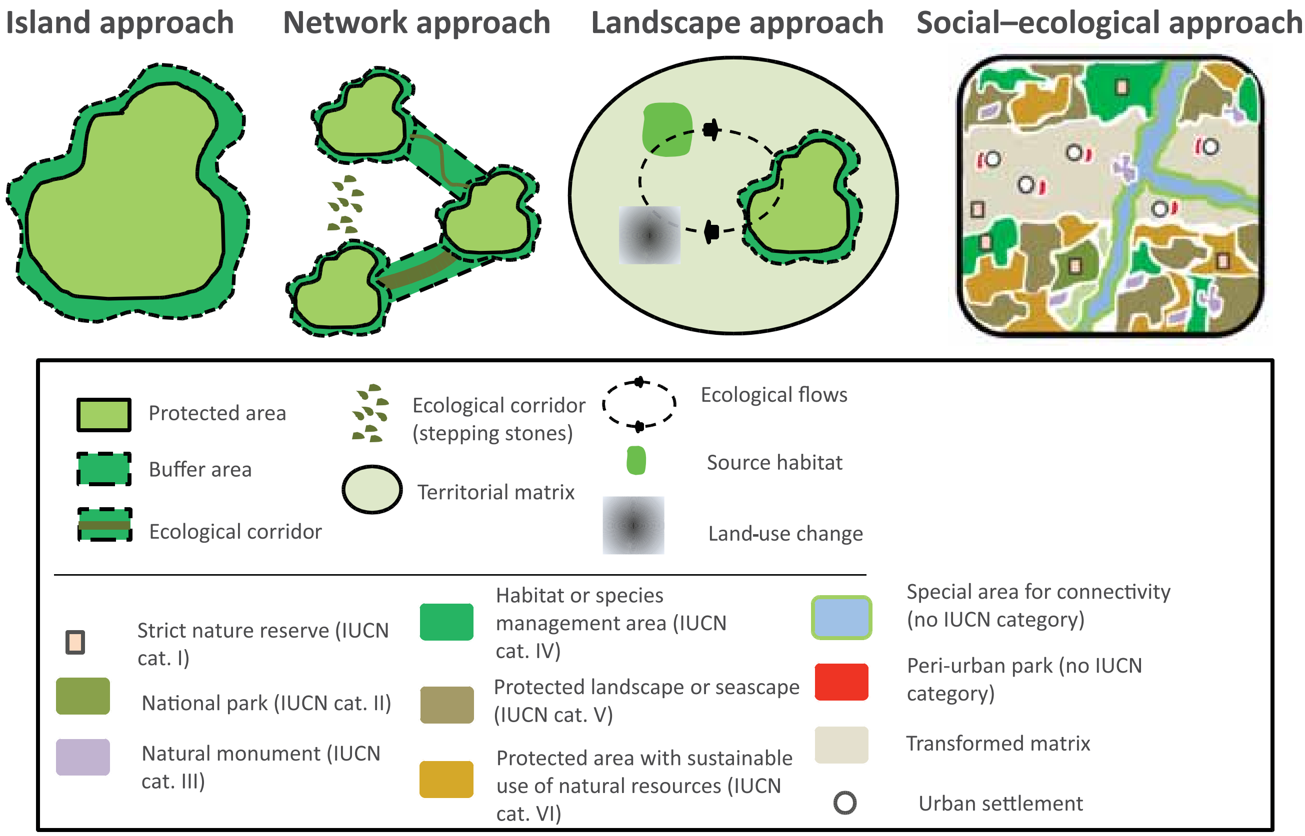

Figure 1. The evolution of the protected-area *concept—from islands to networks to the landscape approach—and the proposed social–ecological approach for protected areas. Abbreviation: IUCN cat., International Union for Conservation of Nature protected-area category.*

biosphere reserve, which advocated for managing the areas around protected areas along a gradient of decreasing use by including buffer zones (Batisse 1982). In the mid-1980s, conservation scientists introduced the concept of *biodiversity hotspots*, which shifted the focus to species-rich areas (Myers et al. 2000) and which later received some criticism as more attention was demanded to protect places with high land-conversion rates.

Ecological corridors and the creation of conservation networks (1990s–mid-2000s). Conservation shifted in the 1990s toward the protection of connections among protected areas and the creation of conservation networks. The concept and application of *ecological corridors* emerged to connect protected areas and foster the movement of species among them (Bennett 1990). This concept was included in the Caracas Action Plan for the Fourth World Congress on National Parks in 1992 (Phillips 2003). During this period, systematic conservation planning emerged, which consists of the use of specific protocols to identify priority areas and separate them from processes that threaten them (Margules and Pressey 2000).

At the global scale, under the ecoregional approach, which is focused on the concept of *representation*, the 200 most important places for conservation, including all of the major biomes, were mapped, and the need for a global network of protected areas was highlighted (Olson and Dinerstein 1998). Currently, many policies for conservation are still focused on spatial networking. The Natura 2000 reserve network, the cornerstone of the European Union's conservation policy, has nevertheless not stopped the loss of European biodiversity (Maiorano et al. 2007).

The landscape approach for protected areas (mid-2000s–today). During the fifth World Parks Congress, "Benefits beyond boundaries," held in Durban, South Africa, in 2003, a conservation approach that considered the surroundings of protected areas was proposed. Expanding conservation beyond the limits of protected areas was also previously advocated in the literature (Holdgate 1994, McNeely 1994). The area around protected areas needs to be managed along with the protected area so that the conservation objectives of the protected area will not be compromised (Hansen and DeFries 2007). Therefore, landscape ecology has been considered a "foundation for sustainable conservation" (Wiens 2009). The idea of buffer zones evolved to include concepts such as the *zone of interaction*, which is an area that "encompasses hydrologic, ecological, and socioeconomic interactions between a protected area and the surrounding landscape" (DeFries et al. 2010, p. 2870). Multiobjective landscape planning, which allows an integrated evaluation of different planning aims, is being developed as part of the *landscape* approach (Bryan and Crossman 2008).

In addition, between the mid-2000s and today, as in previous years, emphasis has been placed on including stakeholders' participation in the decisionmaking processes for protected areas (Phillips 2003), because of the problems of top-down approaches to conservation (Berkes 2004). Despite the important advance of the *landscape* approach, it still does not fully acknowledge that every socioeconomic system is embedded within an ecological system. Moreover, the main purpose of this approach is the maintenance of biodiversity, and it therefore does not fully incorporate the interests of local populations.

Limitations of the current model

In this section, we address the three main limitations of protected areas that we have identified for the long-term conservation of biodiversity and the associated ecosystem services: (1) the effect of drivers of change on protected areas in the context of their territorial isolation, (2) the bias of the location of current protected areas toward high and remote places, and (3) the disconnection between protected areas and society.

Drivers of change and the isolation of protected areas. The Anthropocene is the geological epoch in which the human impacts on the Earth are comparable to those of geological forces, and some authors associate its beginning with the Industrial Revolution (Crutzen 2002). During the Anthropocene, protected areas have been affected by direct drivers of change or pressures (e.g., land-use change, climate change, invasive alien species, overexploitation), which are, in turn, caused by indirect drivers (e.g., sociopolitical, economic, cultural). Land-use change has been especially intense since 1850, after which the surface area of cropland and pasture increased rapidly, greatly reducing the area of natural ecosystems (Foley et al. 2005). Intensive land-use transformations (e.g., the expansion of intensive agricultural production, urbanization) continue to occur around many protected areas (Joppa et al. 2008). As has been acknowledged in the scientific literature, human impacts on the surrounding lands may bleed into protected areas (Myers 1972, Laurence et al. 2012). The long-term conservation of biodiversity cannot be achieved if protected areas are surrounded by degraded habitats that limit the genetic, nutrient, and water flows to and from the outside (McNeely 1994). Moreover, in the long term and as a result of climate change, protected areas will not retain their present ecosystems, and some species will move beyond the protected areas' boundaries (Hannah et al. 2007)—a fact that shows the limitations of protected areas' rigid administrative boundaries.

In the face of global change, we need to navigate through anthropogenic change in order to avoid critical tipping points that would lead to more hostile states of the Earth system (Steffen et al. 2011), such as that caused by a reduction in the number or size of areas with the capacity to generate ecosystem services that maintain human well-being. Protected areas have been one of conservationists' solutions to prevent land-use change. However, anthropogenic change is a complex and nonlinear process, whereas protected areas are a linear answer to that problem (Ewers and Rodrigues 2008). In this sense, protected areas have addressed the effects (i.e., direct drivers, such as land-use change) more than they have the causes (i.e., indirect drivers) of transformation.

The bias in the location and size of current protected areas. A second limitation of protected areas is their bias toward places where they can barely prevent land conversion—for example, elevated areas with steeper slopes that are located far from roads and cities (Joppa and Pfaff 2009). Moreover, protected areas are often designated on the basis of factors other than ecological integrity, such as administrative barriers. These factors sometimes exclude a portion of the area that is needed to maintain ecological processes and populations of organisms (Western et al. 2009). Because protected areas are not the result of organized landscape planning, we have often protected areas that are not useful for economic development plans, such as mountain protected areas. In addition, protected areas are occasionally used in politics to hide the lack of a wider landscape-planning strategy; the political decisions ruling their declaration do not necessarily support ecological needs (Pyke 2007). Moreover, although large protected areas are of great importance to the preservation of key ecological processes, it is now difficult to create the very large protected areas that were declared in the past, because an economics-driven policy usually treats protected areas as restrictions to economic development. In this context, protected-area downgrading, downsizing, and degazettement also threaten their long-term conservation utility (Mascia and Pailler 2010). In fact, recent analyses of the growth of protected areas highlight the decline in the previously exponential rate of expansion of protected areas, especially in the Northern Hemisphere (figure 2).

The disconnection between protected areas and society. A third limitation is the social impact that some local populations experience because of the establishment of protected areas, which reduce societal support for them. Many protected areas were promoted by nature scientists and conservationists, who were governed by wilderness ideals and held the belief that biodiversity had to be safeguarded against the human threat. Protected areas were often imposed from city offices on the rural populations, and many development activities were prohibited. Even the term *protected area* explicitly creates a duality of what is and is not allowed. This exclusionary process of declaring protected areas established a fortress-conservation strategy (Brockington 2002) that separated people from nature. As a result, thousands of people have been displaced from their traditional lands

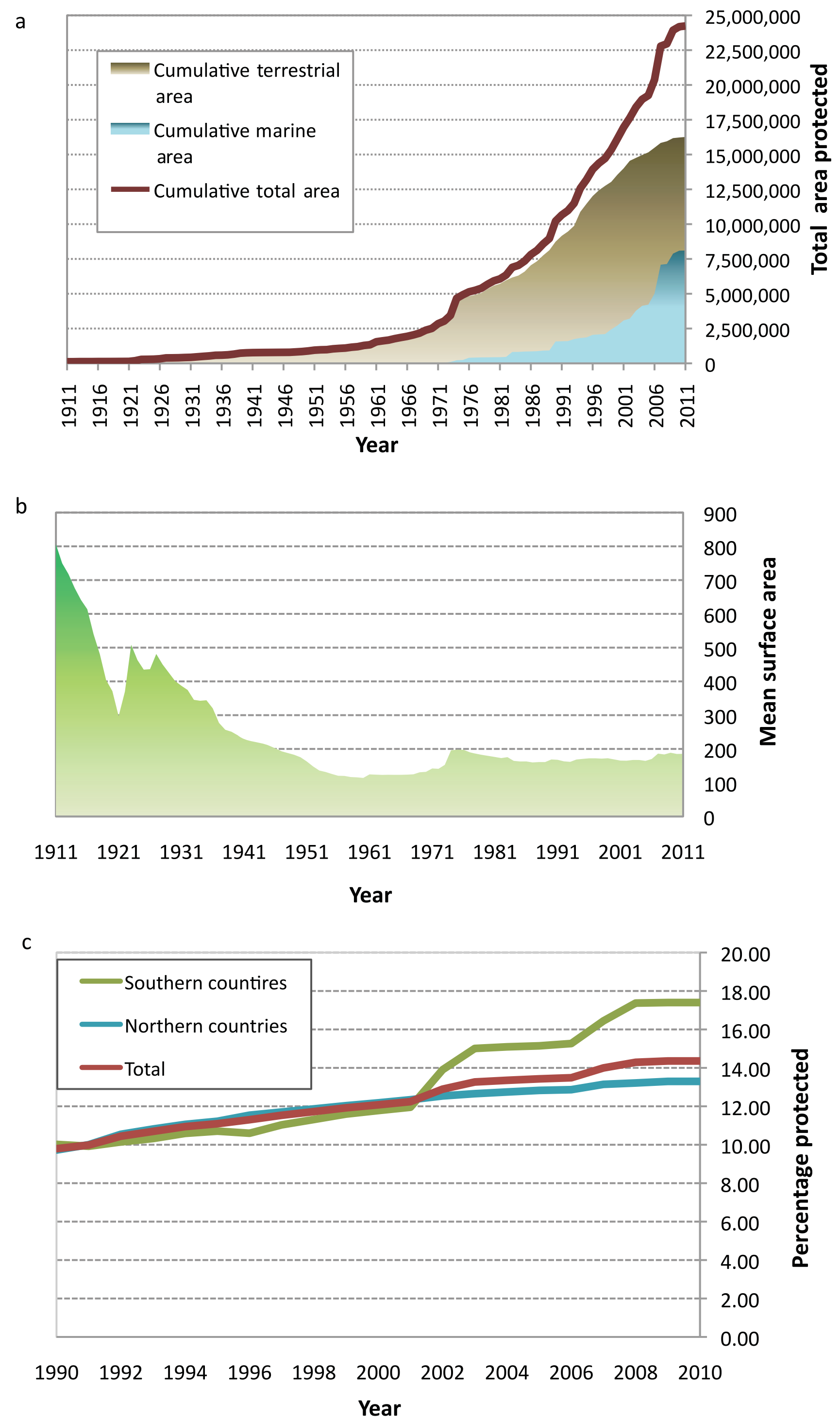

Figure 2. (a) Worldwide growth (in square kilometers) in nationally designated protected areas from 1911 to 2011. (b) Mean surface area (in square kilometers) of nationally designated protected areas from 1911 to 2011. (c) Evolution of the percentage of surface area protected in the Northern and Southern Hemispheres from 1990 to 2010. Source: Adapted with permission from UNEP-WCMC (2010).

(Dowie 2009). This process not only restricts access to several ecosystem services (mostly provisioning, such as gathering, hunting, or wood collection); it can also create poverty and social conflicts (West et al. 2006) and ignores the important role of local and indigenous communities in managing ecosystems and biodiversity.

A social–ecological approach for protected areas

In this section, we describe the concept of protected areas from a social–ecological approach in order to address how it can mitigate the limitations of the current model of protected areas, and we acknowledge the limitations of the approach for managing protected areas.

Protected areas integrated into social–ecological systems and social–ecological science. The concept of landscapes as complex, adaptive systems entails the recognition that human and biophysical components and processes are strongly linked across multiple scales and, therefore, that they should be considered coupled social–ecological systems (Cumming 2011). To fully integrate protected areas into the spatial planning of the landscape, they could be designed and managed by incorporating a social–ecological system approach, which broadens the ways in which we understand protected areas, because it acknowledges the complexity of the social–ecological interactions in every landscape. In this sense, complementing ecological with social analyses will provide a better conservation approach that incorporates the social processes that influence conservation decisions (Ban et al. 2013). To succeed in this challenge, deeper interactions between the social and natural sciences are needed. In box 1, we summarize a case study in the Galápagos Islands in which the social–ecological approach has been included in the management plan of the protected area and address the most important conservation challenges that the archipelago is facing.

Increasing social support for protected areas. The Millennium Ecosystem Assessment (MA) mainstreamed the idea of conservation for human well-being (MA 2005). The basis underlying this conservation concept is the need to reconnect nature and society by considering not only intrinsic but also instrumental values

Box 1. Applying a social–ecological approach in the Galápagos protected areas.

Unlike other oceanic archipelagos, the Galápagos Islands retain most of their original unique biodiversity and have become one of the most renowned natural sites in the world. However, in 2006, a joint mission of the United Nations Educational, Scientific, and Cultural Organization and the International Union for Conservation of Nature concluded that the Galápagos were shifting into an economic development model that was fundamentally at odds with long-term conservation and sustainability interests. The local population was increasing at an alarming rate, triggering imports from mainland Ecuador and increasing the number of alien species. An exponential increase in tourist visitation, the overexploitation of coastal fisheries, and a severe degradation of highland ecosystems were also signs that the islands were threatened.

Taking these negative trends into account, several efforts have been made during the last decade to change the traditional command-and-control management approach (i.e., conservation versus development) that has predominated during most of the Galápagos' recent history. There is no doubt that this paradigm has yielded some good results in the past, but it is failing to solve the present complex problems of the archipelago (González et al. 2008). A new management approach using a social–ecological framework and resilience theory has been put in practice since the approval of a new Galápagos National Park management plan, designed to give new insight into how to address the most important problems and how to cope with major drivers of change.

In the implementation of this new management approach, a transition from traditional management practices to a more-adaptive, resilience-based management model has begun. Participatory planning and decisionmaking has become a rule in the archipelago, particularly after the passing of the Galápagos Special Law in 1998, which notably reduced social conflict and achieved great institutional progress in fishery and tourism management (Heylings and Bravo 2007). Transdisciplinary research has also become a priority for local authorities, and we now have much more accurate information on the links between socioeconomic issues and ecosystem integrity. In this sense, the *ecosystem services* concept has been particularly useful in bridging the human–nature divide.

However, despite the partial success of this new approach, the conceptual landscape-planning proposal, which is the key to facilitating a transitional path toward a more sustainable regime, remains only on paper, as the result of a failure to match governance mechanisms and policy-planning instruments. Institutional weakness and the lack of an appropriate institutional architecture lie behind unsustainable trends that still persist in the archipelago. The tourism industry continues to fuel the economy, acting as the primary indirect driver of change through its direct effects on other drivers, such as population growth, the movement of goods and services, and resource consumption. The increasing array of invasive species and the associated erosion of native fauna and flora are also indirect consequences of the ongoing development of the tourism industry. Energy flows, growing economic opportunities, and positive economic feedback associated with the current model of tourism have the potential to accelerate major changes and to threaten the sustainability of the archipelago, which highlights the need for further efforts to implement the social–ecological approach.

(Folke et al. 2011). One of the principal recommendations of the MA for protected areas is to develop, through legal, policy, and other effective means, stronger societal support based on the benefits and values of the services the protected areas provide (MA 2005). The ecosystems of areas declared as protected provide important services, such as freshwater availability; harvesting; water purification; erosion control; habitat for species; climate regulation; protection from environmental hazards, such as floods or storms; and many other services, such as nature tourism and spiritual and intrinsic values. The ecosystem service approach helps us acknowledge that humans depend on and benefit from protected areas and, therefore, reveals the social benefits that are obtained from ecosystem services, which could enhance current conservation efforts (Haslett et al. 2010). Furthermore, conservation programs that are based on ecosystem services could attract more social support than strict biodiversity conservation programs do, because they are more likely to encompass working landscapes and the people in them (Goldman et al. 2008).

The incorporation of ecosystem services as a reason for the creation of new protected areas has already been suggested (Saunders et al. 2002, Pyke 2007). In fact, the definition of *protected area* has recently included the term *ecosystem services*. In 2008, the International Union for Conservation of Nature's (IUCN) definition of *protected area* was "a clearly defined geographical space, recognized, dedicated, and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008, p. 8). However, the inclusion of ecosystem services in this definition has been more a change in writing than a real incorporation of an ecosystem service framework. Moreover, this definition does not include the *landscape* dimension of protected areas.

Participatory processes and comanagement to reduce social conflict

In contrast to the command-and-control governance that characterizes protected areas as islands, to govern complex adaptive systems, we should incorporate the diverse views of stakeholders, their different value systems, and different knowledge sources (e.g., experimental or scientific knowledge, experiential or local ecological knowledge). To achieve this goal, we should foster the engagement of stakeholders in real participatory processes to enhance communication among decisionmakers, protected-area managers, and protected-area users. This decisionmaking process, which

Box 2. Social–ecological methods used in protected-area management.

There are several social–ecological methods that result in useful information for managing protected areas that include not only the ecological and social dimensions of protected areas but also the main links between the ecological and social systems—that is, the delivery of ecosystem services and the human actions affecting ecosystem integrity and biodiversity. Many of these techniques include social participation, which can foster the engagement of the stakeholders in the management of the protected areas, leading to reduced conflicts. Here, we summarize the most commonly used.

Surveys and interviews

Surveys and interviews are widely applied methods that allow the collection of diverse information from multiple stakeholders. In the Solomon Islands, data collected through these methods have served to combine scientific and indigenous knowledge for the creation of two marine protected areas to protect bumphead parrotfish (Aswani and Hamilton 2004).

The multidimensional assessment of ecosystem services

The multidimensional assessment, an integrated method, consists of assessing different value domains of ecosystem services, such as ecological, social, and monetary values, to better address the complexity of those services. This assessment has provided a complete picture of ecosystem services delivered by Doñana National Park, a protected area in Spain. The assessment shows that the value of a given service depends on the value dimension considered (Martín-López et al. 2013).

Participatory ecosystem-service mapping

This method allows mapping through interviews or workshops the spatial knowledge of stakeholders regarding ecosystem services. The method has been used in the Hinchinbrook Island National Park, in Australia, to identify priority areas for conservation and areas perceived as less worthy so that managers can increase public awareness of these areas (Van Riper et al. 2012) and in Doñana National Park to analyze trade-offs among ecosystem services delivered by protected areas and their surroundings (Palomo et al. 2013b).

Social network analysis

A social network analysis permits the identification of the interactions and relationships among stakeholders in a social–ecological system. In the Peak District National Park, in the United Kingdom, this method has served to inform stakeholder analysis and in the selection of stakeholders for participation in environmental management processes (Prell et al. 2009).

Participatory scenario planning

Participatory scenario planning allows the description of how the future might unfold on the basis of coherent assumptions about the relations among drivers of change and key aspects of the system. The method allows the participation of a great diversity of stakeholders. In Kalimantan, Indonesia, this method has been used to explore the consequences of establishing a new protected area and transforming that area into a palm plantation (Sandker et al. 2007).

involves shared learning, interactions among stakeholders, institutional diversity, and multiscale governance, is reflected in adaptive comanagement strategies. Community-based conservation and participatory approaches may reduce social conflict and may increase the effectiveness of conservation programs (Berkes 2004). Moreover, the *ecosystem service* concept may provide a common language that allows the inclusion of all stakeholder groups in the participatory decisionmaking process regarding the management of landscapes and protected areas. In box 2, we describe some of the most frequently used methodologies from a social–ecological system perspective that can contribute to protected-area management.

Institutions for managing protected areas

Ostrom (2005) recommended that all of the diverse institutions involved in the governance of complex adaptive systems, such as social–ecological systems, be considered and dispersed across multiple governing scales. Therefore, multiple institutions should be involved during the establishment of protected areas and during their management (Ludwig 2001). If the protected area provides the provisioning service of freshwater, the regional water agency might take part in its management; if tourism is delivered, tourism institutions (e.g., corporations, enterprises, local and regional government) might also be directly involved. This would reduce the pursuit of conflicting goals by different institutions and would increase the efficiency of landscape management. In order to do this, a strong coordination between the institutions related to different sectors (e.g., agriculture, water, tourism) and the protected-area institutions must be fostered.

Incorporating beneficiaries

Ecosystem service *beneficiaries* are the stakeholders who directly or indirectly value, use, or enjoy any ecosystem service, including those who will do so in the future. The incorporation of ecosystem service beneficiaries into protected-area planning could highlight the existing scale mismatch between the delivery of ecosystem services and their use and could promote the sustainable use of ecosystem services (Palomo et al. 2013a). Knowledge of this spatial mismatch provides useful information for protected-area management, because it demonstrates that management would improve if its focus were beyond the protected area's biophysical limits, including those places where beneficiaries live. Such knowledge could also help incorporate learning from the social sciences that could be used to understand the factors underlying the demand for ecosystem services and techniques that could be used to foster behavioral changes that would promote the sustainable use of ecosystem services. For example, citizens living in urban areas close to protected areas might enjoy the beauty of the landscape but might also influence soil erosion and vegetation cover by trampling the area. In addition, farming on surrounding lands might reduce the quality of water resources in the protected area. To prevent these problems, it might be adequate for protected areas to have a program that was focused on the beneficiaries of the services that the protected area provides and on the stakeholders who have an impact on ecosystem services delivered by the protected area. In this sense, the *ecosystem service* concept can be used as a principle for conservation beyond protected-area limits, considering the flow from ecosystem service providers to beneficiaries. Therefore, working on the connection between providers and beneficiaries could reduce the problem of isolation and could broaden the limits of protected areas to include the whole territory.

Understanding landscape disparity through ecosystem services

Land-use intensity around protected areas is growing, creating contrasting landscapes between the inside of the protected area and the surrounding area, with negative consequences for the interior of the protected area. Provisioning and cultural services (mainly tourism) normally provide higher economic returns than do regulating services. Therefore, the protected areas' surrounding matrices are sometimes managed mainly to provide services with high economic values, while regulating services are maintained only inside the protected areas (Martín-López et al. 2011). Over the long term, this conservation versus development approach could lead to the overall decline of regulating services, which could also affect the provisioning and cultural services that are maintained by them (Palomo et al. 2013b). The ecosystem service approach highlights these contrasting landscapes and their negative consequences for maintaining a long-term diverse flow of ecosystem services, which result from trade-offs among provisioning, regulating, and cultural services.

Avoiding location bias and the different roles of protected areas in multifunctional landscapes

To overcome location bias, the creation of protected areas in the middle and lower parts of a watershed or near cities should be seen as a priority. New protected areas could occupy areas that have been traditionally dedicated to other uses, such as nonintensive agriculture. This prioritization leads us to the protection of multifunctional landscapes. A diversity of land uses will often increase the resilience of a social–ecological system, because more options are available in a diverse system (Cumming 2011). Some studies have demonstrated that multifunctional landscapes can provide several ecosystem services and greater economic benefits than the conversion to a single use would if the environmental costs and ecosystem services are taken into account (Balmford et al. 2002). In addition, in landscape-aesthetic evaluations, stakeholders prefer multifunctional landscapes over intensively managed lands, because, to some extent, people perceive that multifunctional landscapes are able to deliver a broad range of ecosystem services (García-Llorente et al. 2012). In fact, stakeholders perceive that those protected areas that are embedded in multifunctional landscapes deliver more ecosystem services than do strictly conserved lands (such as IUCN protected-area categories I and II) or intensively managed lands (Martín-López et al. 2012). In this context, in the debate between *land sparing* (in which intensively managed lands and strict conservation areas coexist) and *land sharing* (in which multifunctional landscapes are promoted), land-sharing conservation schemes may help overcome the location bias of protected areas. Consequently, multiple-use protected areas (i.e., IUCN categories V and VI) will be common under future scenarios of increasing human population and demand for ecosystem services (McDonald and Boucher 2011).

The success of protected areas demands a dynamic approach in which the entire landscape is considered (Bengtsson et al. 2003). In a regional and social–ecological approach, protected areas could be a functional part of the system, playing several roles, such as (a) reservoirs of ecological memory, through the preservation of biodiversity and functional diversity; (b) providers of different ecosystem services if protected-area designation and management involves proactive analyses of the ecological and societal vocation of the territory; and (c) reservoirs of social memory (mainly for protected areas included in IUCN categories V and VI), through the preservation of local ecological knowledge.

With an integrative landscape-management strategy, such as the one proposed in the present article, we would not need to increase protected areas' coverage indefinitely. To avoid impacts to the protected area, we consider the biosphere reserve model to be the best, because it acknowledges that the matrix of the protected area can be transformed only along a gradient of uses. Following a social–ecological approach, a *protected area* can be defined as a geographical space managed through legal or other effective means under a social–ecological framework that acts as a functional unit of the landscape to maintain biodiversity and the associated ecosystem services that contribute to long-term human well-being.

Limitations of the social–ecological approach for managing protected areas

The ecosystem services framework is being developed to provide better outcomes for biodiversity and human well-being (Daily 1997). However, some scientists have acknowledged several limitations of current ecosystem services science and its implementation (Sagoff 2011). Therefore, the debate over whether the ecosystem services approach will provide better outcomes for biodiversity conservation is still open (Reyers et al. 2012). Protected areas have proven successful in most cases to protect ecosystems and biodiversity, so a change in strategy could be argued against. However, incorporating ecosystem services in the conservation discourse could foster the creation of new protected areas and could help cope with some of the challenges that protected areas will face in the future. Protected-area management will certainly face decisions in which scenarios are not win–win (for biodiversity and for ecosystem services and, therefore, for human well-being). Situations in which the strict protection of biodiversity might diminish ecosystem services beneficial to society should be examined on a case-by-case basis. Therefore, we do not advocate substituting biodiversity for ecosystem services as the primary goal of protected areas. However, we call for a broader and more adaptive social–ecological approach for protected areas and their surrounding landscapes that includes both the intrinsic value (i.e., biodiversity conservation) and the instrumental value (i.e., ecosystem services) of nature. Considering both types of values necessitates that both social and ecological information be taken into account, along with varied sources of knowledge (e.g., scientific, local ecological knowledge); more-informed decisions and better outcomes for biodiversity, the local population, and society might therefore be achieved.

Nonetheless, numerous circumstances complicate the management of protected areas beyond their boundaries, such as rigid conservation institutions; the conflicting needs, interests, or level of power that different stakeholders have concerning landscape planning; the different levels of information, communication, and trust among different social agents; and the influence of indirect drivers of change, such as the global economy or demographic trends, on conservation strategies at the regional scale (Schonewald-Cox et al. 1992, MA 2005). These represent real challenges for the success of conservation biology and protected areas.

Conclusions

The original concept of a *protected area* has strongly influenced the development of the subsequent protected-area models. In this sense, our response to the decline of biodiversity and ecosystem services has been to increase the supply side of conservation in terms of the number and size of protected areas, instead of managing the drivers of change and the demand for specific ecosystem services (which ultimately threatens biodiversity and ecosystem services). Protected areas are necessary in order to maintain biodiversity and ecosystem services in the long term, but they are not currently sufficient. Moreover, protected areas face several limitations, such as isolation, location bias, and lack of societal support. Because of the large expanses covered by protected areas, they represent a great opportunity for landscape management. In a changing world, we need new integrative and holistic approaches for the conservation of biodiversity, ecosystem services, and protected areas. These can be achieved by incorporating a social–ecological approach to protected areas, (a) because ecosystem services may bring protected areas more in line with the needs of society by highlighting all of the benefits that we derive from those areas; (b) because ecosystem services provide an integrative language for community-based management and offer a real opportunity for stakeholder involvement in the management of the protected area; (c) because ecosystem services demonstrate the spatial connections between the service-providing areas and the service-benefitting areas, thus promoting regional landscape planning beyond the limits of the protected area; and (d) because the social–ecological approach integrates the effects of drivers of change in ecosystems with social and ecological sciences that might improve the management of protected areas and their surrounding landscapes.

Acknowledgments

We thank Gretchen C. Daily for her useful comments on an early version of the manuscript and three anonymous reviewers for their suggestions. We also thank the Spanish Ministry of the Environment (project no. 018/2009) and the Ministry of Economy and Competitiveness (project no. CGL2011-30266) for the financial support that permitted the creation of this article and the Ministry of Education for a University Teacher Training fellowship.

References cited

Andam KS, Ferraro PJ, Pfaff A, Sanchez-Azofeifa GA, Robalino JA. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. Proceedings of the National Academy of Sciences 105: 16089–16094.

Armsworth PR, Chan KMA, Daily GC, Erhlich PR, Kremen C, Rickets TH, Sanjayan MA. 2007. Ecosystem-service science and the way forward for conservation. Conservation Biology 21: 1383–1384.

Aswani S, Hamilton RJ. 2004. Integrating indigenous ecological knowledge and customary sea tenure with marine and social science for conservation of bumphead parrotfish (*Bolbometopon muricatum*) in the Roviana Lagoon, Solomon Islands. Environmental Conservation 31: 69–83.

Balmford A, et al. 2002. Economic reasons for conserving wild nature. Science 297: 950–953.

Ban NC, et al. 2013. A social–ecological approach to conservation planning: Embedding social considerations. Frontiers in Ecology and the Environment 11: 194–202.

Batisse M. 1982. The biosphere reserve: A tool for environmental conservation and management. Environmental Conservation 9: 101–111.

Bengtsson J, Angelstam P, Elmqvist T, Emanuelsson U, Folke C, Ihse M, Moberg F, Nyström M. 2003. Reserves, resilience and dynamic landscapes. AMBIO 32: 389–396.

Bennett AF. 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecology 4: 109–122.

Berkes F. 2004. Rethinking community-based conservation. Conservation Biology 18: 621–630.

Bertzky B, Corrigan C, Kemsey J, Kenney S, Ravilious C, Besançon C, Burgess N. 2012. Protected Planet Report 2012: Tracking Progress towards Global Targets for Protected Areas. International Union for Conservation of Nature and the United Nations Environment Programme's World Conservation Monitoring Centre.

Brockington D. 2002. Fortress conservation: The preservation of the Mkomazi Game Reserve, Tanzania. International Journal of African Historical Studies 35: 594–596.

Bryan BA, Crossman ND. 2008. Systematic regional planning for multiple objective natural resource management. Journal of Environmental Management 88: 1175–1189.

Butchart SHM, et al. 2010. Global biodiversity: Indicators of recent declines. Science 328: 1164–1168.

[CBD] Convention on Biological Diversity. 1992. Convention on Biological Diversity. United Nations.

———. 2010. COP 10 Decision X/2: X/2. Strategic Plan for Biodiversity 2011–2020. United Nations. (31 October 2013; *www.cbd.int/decision/cop/default.shtml?id=12268*)

Chape S, Harrison J, Spalding M, Lysenko I. 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. Philosophical Transactions of the Royal Society B 360: 443–455.

Crutzen PJ. 2002. Geology of mankind. Nature 415: 23.

Cumming GS. 2011. Spatial Resilience in Social–Ecological Systems. Springer.

Daily GC, ed. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island.

DeFries R, Karanth KK, Pareeth S. 2010. Interactions between protected areas and their surroundings in human-dominated tropical landscapes. Biological Conservation 143: 2870–2880.

Dowie M. 2009. Conservation Refugees: The Hundred-Year Conflict between Global Conservation and Native Peoples. Massachusetts Institute of Technology.

Dudley N, ed. 2008. Guidelines for Applying Protected Areas Management Categories. International Union for Conservation of Nature.

Ewers RM, Rodrigues ASL. 2008. Estimates of reserve effectiveness are confounded by leakage. Trends in Ecology and Evolution 23: 113–116.

Foley JA, et al. 2005 Global consequences of land use. Science 309: 570–574.

Folke C, et al. 2011. Reconnecting to the biosphere. AMBIO 40: 719–738.

García-Llorente M, Martín-López B, Iniesta-Arandia I, López-Santiago CA, Aguilera PA, Montes C. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: An ecosystem service approach. Environmental Science and Policy 19–20: 136–146.

Goldman RL, Tallis H, Kareiva P, Daily GC. 2008. Field evidence that ecosystem service projects support biodiversity and diversify options. Proceedings of the National Academy of Sciences 105: 9445–9448.

González JA, Montes C, Rodríguez J, Tapia W. 2008. Rethinking the Galapagos Islands as a complex social–ecological system: Implications for conservation and management. Ecology and Society 13 (art. 13).

Hannah L, Midgley G, Andelman S, Araújo M, Hughes G, Martinez-Meyer E, Pearson R, Williams P. 2007. Protected area needs in a changing climate. Frontiers in Ecology and the Environment 5: 131–138.

Hansen AJ, DeFries R. 2007. Ecological mechanisms linking protected areas to surrounding lands. Ecological Applications 17: 974–988.

Haslett JR, Berry PM, Bela G, Jongman RHG, Pataki G, Samways MJ, Zobel M. 2010. Changing conservation strategies in Europe: A framework integrating ecosystem services and dynamics. Biodiversity and Conservation 19: 2963–2977.

Heylings P, Bravo M 2007. Evaluating governance: A process for understanding how co-management is functioning, and why, in the Galapagos Marine Reserve. Ocean and Coastal Management 50: 174–208.

Holdgate MW. 1994. Protected areas in the future: The implications of change, and the need for new policies. Biodiversity and Conservation 3: 406–410.

Janzen DH. 1983. No park is an island: Increase in interference from outside as park size decreases. Oikos 41: 402–410.

Joppa LN, Pfaff A. 2009. High and far: Biases in the location of protected areas. PLOS ONE 4 (art. e8273).

Joppa LN, Loarie SR, Pimm SL. 2008. On the protection of "protected areas." Proceedings of the National Academy of Sciences 105: 6673–6678.

Kareiva P, Marvier M. 2012. What is conservation science? BioScience 62: 962–969.

Laurence WF, et al. 2012. Averting biodiversity collapse in tropical forest protected areas. Nature 489: 290–294.

Lovejoy TE. 2006. Protected areas: A prism for a changing world. Trends in Ecology and Evolution 21: 329–333.

Ludwig D. 2001. The era of management is over. Ecosystems 4: 758–764.

[MA] Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute.

Maiorano L, Falcucci A, Garton EO, Boitani L. 2007. Contribution of the Natura 2000 network to biodiversity conservation in Italy. Conservation Biology 21: 1433–1444.

Margules CR, Pressey RL. 2000. Systematic conservation planning. Nature 405: 243–253.

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.

Martín-López B, et al. 2012. Uncovering ecosystem service bundles through social preferences. PLOS ONE 7 (art. e38970).

Martín-López B, Gómez-Baggethun E, García-Llorente M, Montes C. 2013. Trade-offs across value-domains in ecosystem services assessment. Ecological Indicators 37: 220–228.

Mascia MB, Pailler S. 2011. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conservation Letters 4: 9–20.

McDonald RI, Boucher TM. 2011. Global development and the future of the protected area strategy. Biological Conservation 144: 383–392.

McNeely JA. 1994. Protected areas for the 21st century: Working to provide benefits to society. Biodiversity and Conservation 3: 390–405.

Myers N. 1972. National parks in savannah Africa. Science 178: 1255–1263.

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J. 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853–858.

Olson DM, Dinerstein E. 1998. The global 200: A representation approach to conserving the Earth's most biologically valuable ecoregions. Conservation Biology 12: 502–515.

Ostrom E. 2005. Understanding Institutional Diversity. Princeton University Press.

Palomo I, Martín-López B, Potschin M, Haines-Young R, Montes C. 2013a. National parks, buffer zones and surrounding lands: Mapping ecosystem service flows. Ecosystem Services 4: 104–116.

Palomo I, Martín-López B, Zorrilla-Miras P, García del Amo D, Montes C. 2013b. Deliberative mapping of ecosystem services within and around Doñana National Park (SW Spain) in relation to land use change. Regional Environmental Change. (1 November 2013; *http://link.springer.com/article/10.1007%2Fs10113-013-0488-5*) doi:10.1007/s10113-013-0488-5

Phillips A. 2003. Turning ideas on their head: The new paradigm for protected areas. The George Wright Forum 20: 8–32.

Prell C, Hubacek K, Reed M. 2009. Stakeholder analysis and social network analysis in natural resource management. Society and Natural Resources 22: 501–518.

Pyke CR. 2007. The implications of global priorities for biodiversity and ecosystem services associated with protected areas. Ecology and Society 12: 4.

Reyers B, Polasky S, Tallis H, Mooney HA, Larigauderie A. 2012. Finding common ground for biodiversity and ecosystem services. BioScience 62: 503–507.

Rodrigues ASL, et al. 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428: 640–643.

Sagoff M. 2011. The quantification and valuation of ecosystem services. Ecological Economics 70: 497–502.

Sandker M, Suwarno A, Campbell BM. 2007. Will forests remain in the face of oil palm expansion? Simulating change in Malinau, Indonesia. Ecology and Society 12 (art. 37).

Saunders DL, Meeuwig JJ, Vincent ACJ. 2002. Freshwater protected areas: Strategies for conservation. Conservation Biology 16: 30–41.

Schonewald-Cox C, Buechner M, Sauvajot R, Wilcox BA. 1992. Cross-boundary management between national parks and surrounding lands: A review and discussion. Environmental Management 16: 273–282.

Steffen W, et al. 2011. The Anthropocene: From global change to planetary stewardship. AMBIO 40: 739–761.

[UNEP-WCMC] United Nations Environment Programme World Conservation Monitoring Centre. 2010. World Database on Protected Areas. UNEP-WCMC. (5 November 2013; *www.wdpa.org*)

Van Riper CJ, Kyle GT, Sutton SG, Barnes M, Sherrouse BC. 2012. Mapping outdoor recreationists' perceived social values for ecosystem services at Hinchinbrook Island National Park, Australia. Applied Geography 35: 164–173.

West P, Igoe J, Brockington D. 2006. Parks and peoples: The social impact of protected areas. Annual Review of Anthropology 35: 251–277.

Western D, Russell S, Cuthill I. 2009. The status of wildlife in protected areas compared to non-protected areas of Kenya. PLOS ONE 4 (art. e6140).

Wiens JA. 2009. Landscape ecology as a foundation for sustainable conservation. Landscape Ecology 24: 1053–1065.

Ignacio Palomo (ignacio.palomo@uam.es), Carlos Montes, Berta Martín-López, José A. González, Marina García-Llorente, and Paloma Alcorlo are affiliated with the Social–Ecological Systems Laboratory, in the Department of Ecology, at the Autonomous University of Madrid, Spain. MG-L is also affiliated with the Sociology of Climate Change and Sustainable Development research group, in the Department of Social Analysis, at the University Carlos III, also in Madrid. María Rosario García Mora is affiliated with the Ministry of the Environment of the government of Andalusia, in Seville, Spain.