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Beyond parks and reserves: The ethics and politics of conservation with a case study from Perú

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ABSTRACT

This paper elaborates, analyzes, and partly defends the normative and empirical foundations of a "social ecology" model for natural habitat and resource management. This model treats human societies as being irreducibly integrated with the natural systems in which they are embedded. It argues that any concept of biodiversity necessarily embodies cultural values for it to be operationalized for conservation decisions. It accepts the legitimacy of tradeoffs between biodiversity conservation and other values including human resource development. It prioritizes local control of decisions in regions which are often targeted by Northern conservationists: areas of the South, where cultural choices have led to the persistence of high biodiversity. This model is used to analyze an ongoing dispute over biodiversity conservation and natural resource control in Perú: the conflict over Kandozi territory in the Abanico del Pastaza. What the Kandozi want is more than just a rejection of what has been criticized as the national park/fortress model of conservation. It consists of an assertion of local institutionalized control over traditional lands which goes well beyond the purview of the usual alternative of the fortress model: the biosphere reserve model of external and internal joint control of natural resources. To the remarkable extent that Kandozi resource management practices have succeeded in maintaining biotic richness and variety in spite of multiple encroachments, the empirical evidence demands that today's conservationists pay adequate attention to these practices and the social institutions in which they are embedded. But, beyond such prudential concerns, the social ecology model of habitat conservation accepts the normative claims that resident communities should have control over their lands and livelihoods and that they should be allowed to maintain their habitats as cultural landscapes of their choice.

1. Introduction

The last decade has seen a remarkable convergence of methods used in some parts of conservation science, especially on how networks of protected areas should be selected to represent biodiversity (Margules and Pressey, 2000; Sarkar et al., 2006; Margules and Sarkar, 2007; Moilanen et al., 2009). However, there has not been a similar convergence on how they should be managed to ensure the persistence of the biodiversity within them: whether as national parks which exclude human habitation and use (hereafter the "fortress" model [Brockington, 2002]), or as reserves with (partly) externally controlled human habitation and use (the "biosphere reserve [BR]" model), or by local communities which set their own goals and management protocols (the "social ecology [SE]" model). Both the SE and BR models should be seen as belonging to a wide spectrum of possibilities, all to be distinguished from the fortress model because they endorse the ethical salience and legitimacy of human needs and aspirations in the context of biodiversity conservation (Sarkar, 2003, 2005).

The growing popularity of the BR model in the 1990s marked a return to an anthropocentric program for biodiversity conservation that was prompted by a variety of factors, especially the promotion of sustainable development, starting with the 1987 Brundtland Commission Report (World Commission on Environment and Development, 1987), as emphasized by Miller et al. (this issue) in their discussion of the "new conservation debate". While the extent of the influence of each factor is difficult to disambiguate without the benefit of historical distance, two other factors were also important: (i) ethically based Southern critiques of the forced expulsion of local residents to create parks, as endorsed by the fortress model (Guha, 1989; Sarkar, 1998, 1999); and (ii) a general perception that conservation cannot be successful without local support (Brechin et al., 2002). In particular, there was the striking case of India's Project Tiger, once hailed as a storybook success story (Panwar, 1982), which collapsed to a large extent because villagers expelled to create Tiger Reserves collaborated with poachers

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to decimate tiger populations across the region (Damodaran, 2007).

However, the last decade has seen a stridently vocal group of Northern conservationists demanding a return to the fortress model (Kramer et al., 1997; Brandon et al., 1998; Oates, 1999; Terborgh, 1999). While there has been little theoretical discussion of their agenda [hence, the contribution by Miller et al. (this issue) is timely], this retrenchment has been motivated by both philosophical assumptions and empirical judgments. The philosophical foundation has been provided by a reassertion of non-anthropocentric values as the basis for conservation of biodiversity. The empirical judgment has been that integrated conservation and development projects (ICDPs) (Alpert, 1996) do not work. These projects, which are one way in which protocols for sustainable development have been implemented, fall under the rubric of the BR model distinguished earlier. The debate between the fortress and BR models has been summarized in the paper by Miller et al. (this issue). (The terms ''North'' and ''South'' are used here to distinguish traditions of environmentalism loosely tied to dominant European and neo-European cultures of nature which view humans as categorically distinct from the natural world [the ''North''] from worldviews, generally associated with post–colonial social movements, which view nature as a resource [the ''South]. The terms only have well-defined geographical referents to this limited extent but that is all that is required for the arguments of this paper.)

The aim of this paper is to elaborate a third model for biodiversity conservation, which we earlier called the ''social ecology [SE]'' model, and which goes beyond the BR model (Guha, 1994; Sarkar, 1998), the latter being essentially what Miller et al. (this issue) call the ''social conservationist'' model. Box 1 provides one straightforward protocol by which the SE model can be implemented in practice through appropriate methodology being introduced into the framework of systematic conservation planning. The SE model is supposed to be applicable at least in those Southern contexts in which communities have lived in and controlled a habitat from pre-colonial times while the flora and fauna of the habitat have continued to flourish. We will call these social groups ''resident communities''. (The terminology is a bit misleading: we do not intend to exclude nomadic groups; rather, we regard them as resident over a wider geographical area.) Indigenous communities obviously fall under the rubric of this definition but we are not distinguishing between indigeneity and long historical presence and resource control. In the absence of any obvious better alternative, the choice of the pre-colonial starting time was made for convenience. To avoid later misunderstanding, even though, from our perspective, habitat conservation does not trump social justice, we want to emphasize that: (i) cases in which there is historical or ongoing habitat degradation fall outside the intended domain of the SE model as formulated here; (ii) we are assuming that there is no ongoing drastic change in (human) population size or social structure; and (iii) that we can ignore the introduction of new environmentally destructive technologies. For lack of adequate space, we leave for another occasion a discussion of these issues.

Section 2 elaborates a normative framework to distinguish the fortress, BR, and SE models. We emphasize the normative philosophical basis for the positions adopted here but also note how pragmatic prudential considerations may also encourage a shift from the fortress to at least the BR model. Turning to the SE model, in Section 3 we provide a characterization with sufficient detail to analyze our case study. We turn to the case study, Kandozi community activism in Perú, in Section 4. The case study is particularly interesting because it shows that the Kandozi—justifiably—do not accept that a transition from the fortress model to a BR model is sufficient to protect the natural resources of their lands. We further discuss the SE model and draw some conclusions in Section 5.

Social ecology in systematic conservation planning.

The following are the stages of systematic conservation planning, as elaborated by Margules and Sarkar (2007), with a first and a seventh stage added to conform to the requirements of the SE model. All stages except the sixth require modification from what would be done under the fortress model. The stages are not meant to be followed linearly and may influence each other, for instance, the first two stages.

- *Delimit boundaries of the study area* (*or planning region*): These boundaries must be set by local residents and may, for instance, incorporate traditional land use patterns. If local control is accepted, the boundaries may well be such that the study area crosses national borders. This is an additional virtue of the SE model insofar as political boundaries have often made biodiversity conservation planning ecologically sub-optimal.
- *Identify stakeholders for the planning region*: Local residents are privileged stakeholders. The SE model requires that they determine who else is a legitimate stakeholder. National and trans-national entities must negotiate with local residents to obtain such status. This is perhaps the most radical aspect of the SE model: it does not automatically guarantee stakeholder roles even to *conservation* NGOs.
- *Compile, assess, and refine natural feature and socio-cultural information for the region*: The only innovation in the SE model is its emphasis on socio-cultural information beyond data on biodiversity and other natural features.
- *Identify constituents of biodiversity*: Given that not every biotic feature can reasonably be targeted for protection, what we decide to protect must be a cultural choice. In practice this has always been true; the SE model accepts it in principle. This point is elaborated in detail in Section 2.
- *Establish conservation and social goals and targets*: The SE model explicitly accepts that attainment of desired social goals must be part of a conservation plan. As such, it is similar to programs of sustainable development.
- *Identify surrogates for biodiversity*: These are the features used to obtain quantitative estimates of the biodiversity constituents of a region. Technical protocols have been developed for this purpose (Sarkar et al., 2005; Margules and Sarkar, 2007).
- *Establish measures for socio-cultural goals*: The SE model requires a parallel effort to assess the performance of a plan in satisfying the socio-cultural goals for a region. It presumes that these decisions will ultimately be made by local residents.
- *Review existing conservation areas*: If some areas are already protected, the SE model requires that they be reviewed not only with respect to biodiversity representation but with respect to the socio-cultural goals for the region.
- *Prioritize areas for potential conservation action*: Priority areas must be selected for the achievement of both adequate biodiversity representation and the socio-cultural goals. All criteria that are relevant to these

decisions may be incorporated at this stage or, alternatively, some of them may be incorporated using multi-criteria analysis (as discussed below).

- *Assess threats to biodiversity and socio-cultural goals*: Part of the motivation for the SE model was the realization that the same factors often threaten biodiversity and the well-being of residents of many areas of the South. However, where they are different, the SE model requires explicit attention to the threats to human well-being. The purpose of the threat assessment is to use the results to refine the prioritization of areas for conservation management.
- *Use multi–criteria analysis to incorporate socio-cultural goals*: If these goals were not already incorporated during the prioritization stage, they can subsequently be included through multi-criteria analysis. Techniques for this have been developed by the decision theory community over the last four decades (Moffett and Sarkar, 2006).
- *Implement plan*: Because the SE model sees the planning process as "community driven" implementation of the plan becomes the task of the planners themselves. Here, the SE model has a major advantage over the fortress and even the BR model which envision planners from outside who are typically disengaged from the political and social life of a region. It is expected that the SE model will avoid a major pitfall of the fortress model, namely, that plan implementation is immensely more difficult compared to plan formulation (and often never achieved in complete).
- *Periodically reassess success*: Like traditional conservation planning, the SE model envisions adaptive management with periodic reviews and refinements over the life of a plan.

2. Normative framework

We will use three normative positions (principles or claims) to try to provide a more nuanced explication of the conceptual relationships between the fortress, BR, and SE models than has so far appeared in the literature (Sarkar, 2005). Part of the novelty of the framework is that it distinguishes sharply between the BR and SE models. Consequently, the debate over managing habitats for biodiversity should not be seen as simply one between those who advocate defending fortresses and those who accept human use of natural habitats. The most important feature of the SE model is its emphasis on local control (that is, control by resident communities) of natural resources.

The first of the three positions is well-known and is perhaps the central philosophical claim within non-anthropocentric ideologies (worldviews with a normative component) of conservation: biodiversity has intrinsic value. The second is an assumption that is implicitly made in much of conservation science but is open to empirical questioning: that biodiversity has a universal definition. The third is a pervasive operative assumption of conservation biologists and non-governmental organizations (NGOs) from the North: that biodiversity is a global resource and responsibility—thus, the interests and values of these Northern agents are relevant to habitat management decisions affecting resident communities in the South and, indeed, can trump the interests and values of these communities:

- *Biodiversity has intrinsic value*: The relevant contrast is with the position that biodiversity has value because it answers human needs and aspirations, including spiritual, emotional, and intellectual needs. Within environmental ethics, the debate over whether non-human entities, particularly classes of individuals (such as species and higher taxa), can have intrinsic value erupted in the 1970s and a transformed—and, arguably, tired—version continues to this day [see Jamieson (2008) for a summary of recent work]. This debate does not admit short summary and will not be recounted here [extensive reviews are widely available (Norton, 1987; Sarkar, 2005; Jamieson, 2008; Sarkar, in preparation)]. Since the SE model rejects intrinsic value attributions to biodiversity, suffice it here to note that most [though not all—see, *e.g.*, McShane (2007)] *professional* philosophers hold that, to the extent that any philosophical issue is ever fully resolved, intrinsic value arguments for biodiversity conservation cannot be sustained.
- *Biodiversity has a universal definition*: What is biodiversity? By this we do not mean the question how, given ubiquitous resource and data limitations, do we choose surrogates for biodiversity (Sarkar and Margules, 2002), for instance, to select priority areas for conservation. Before we can judge whether such surrogates (*e.g.*, classes of taxa or habitat types) are adequate, we must ask what are they supposed to be surrogates for Sarkar (2002). Those more fundamental foci of attention are the "constituents" of biodiversity [what Margules and Sarkar (2007), Sarkar (2002), Sarkar and Margules (2002) called "true surrogates"]. But what are these constituents? There is one obvious candidate answer: all diversity at all levels of biological structural, functional, and taxonomic organization. Unfortunately this answer is useless. It identifies all of biology with biodiversity (Takacs, 1996). Consequently, it cannot be operationalized even to the limited extent required to determine what are adequate surrogates for the constituents (Sarkar, 2002; Sarkar and Margules, 2002). If we were to use this definition in conservation planning, protecting biodiversity would commit us to protect every biotic entity. In practice, different entities make different choices (Margules and Sarkar, 2007). In the United States, most governmental agencies use endangered and threatened species but that is because much of conservation policy is set in the context of the legal requirements of the Endangered Species Act (ESA) (1973). In contrast, The Nature Conservancy (TNC) uses habitat types defined by characteristic ecological communities. Conservation International (CI) uses both globally threatened and geographically concentrated species. Some such choice is necessary in order to provide the minimal precision required to devise conservation policy. Each of these choices reflects cultural values. For instance, US governmental agencies and CI implicitly presume that species are the bearers of value. Moreover, they presume that the extinction of every species is equally (normatively) undesirable. (However, US governmental agencies are allowed to devalue insect pests because these are excluded from the ESA.) TNC implicitly presumes that ecological communities are the bearers of value. The critical point is that these definitions embody cultural norms even though they are often presented as if they are universal purely scientific definitions (Sarkar, 2008). As noted earlier, there is no single veridical scientific candidate for a definition of biodiversity that can be operationalized for conservation decisions (Sarkar, 2002, 2008). Moreover, there are many other equally defensible choices. Cloud forest resident communities of the Eastern Himalayas of India value sacred groves, patches of forest from which (sometimes) not even deadwood can be removed (for details, see Sarkar, in preparation). Many cultures around the world value individual species. But this diversity of views need

not lead to a vapid cultural relativism in which anything can count as biodiversity. We leave ample room for disagreement which may potentially be resolved: for instance, within a culture we may debate what we value most, whether we value every endangered species as much as we value selected endemic or charismatic ones (species of symbolic and other cultural value). Moreover, cultural values evolve and there can be cross-cultural dialectics of engagement, disagreement, and change. Finally, we may impose some adequacy conditions that delimit which forms of valuing natural entities may count as valuing biodiversity. As an example, if we impose a condition that an adequate definition must value entities that cover a large portion of the taxonomic spectrum, valuing totemic species would not count as valuing biodiversity (Sarkar, in preparation). But even these adequacy conditions have to be culturally debated. The most important conclusion to draw from this discussion is that any claim that there is a single universal definition of biodiversity is incoherent.

- *Biodiversity is a global resource and responsibility*: It is commonplace for Northern conservationists to propose policies for distant lands in the South and to demand action. In the 1980s the British parliament debated sending British troops to Kenya, Tanzania, and Mozambique to protect elephants (Neumann, 2004). In the Central African Republic, in the 1990s, Bruce Hayes (a co-founder of Earth First! in the United States), hired mercenaries to shoot at alleged poachers with no semblance of a trial, let alone a fair trial (Neumann, 2004). Even when military threats are not used—unlike these African examples—economic power is often deployed against people living near or below the subsistence level if they do not conform to the demands of Northern conservationists Dowie (2009).
Now, consider a hypothetical example from central Texas which is home to a suite of endangered and endemic species including birds, salamanders, and arthropods (Beatley, 1994; Beatley et al., 1995). In central Texas, attempts to list species under the ESA, delineate critical habitat for them, and develop habitat conservation plans, have long been controversial, often leading to ugly confrontations between landowners and conservationists. Now, imagine that an environmentalist from Mongolia decides to come to Texas, claim expertise on desert landscapes and cave ecology (perhaps justifiably), and demands that prime real estate around Austin be converted into a national park. It is intriguing to speculate on the reactions from gun-toting Texans. But, is there a salient *ethical* difference between this hypothetical situation and the one in which Oates (1999) (among others) demands more and better-policed national parks in west Africa? Or is it simply a question of power relations? From an ethical perspective, in both situations either we are violating basic human rights or we are not. We are either accepting the privilege of local residents on the use of habitat or not. If we are forced to conclude that all that differentiates the two situations are power relations, it is hard to avoid the conclusion that Northern conservationism is a continuation of colonial attitudes and policies in the South (Guha, 1997). We will leave further discussion of this controversial question for another occasion [but see Dowie (2009)]. The normative issue here is that of parity. What one community whether it be Northern conservationists or Mongolian desert experts) values cannot be transferred without consent to the habitats of other communities. When we couple this with the realization that any definition of biodiversity is context-dependent (Escobar, 1996), then there is no longer plausible to globalize biodiversity rights and responsibilities. Northern conservationists have no privileged status in the South.

These three positions allow a useful differentiation of the fortress, BR, and SE conservation models. We will introduce some necessary qualifications later because there are differences amongst proponents of each model. Nevertheless, for a start: the fortress model endorses all three positions; the BR model endorses the second, rejects the third, and is neutral with respect to the first; and the SE model rejects all three. However, a proponent of the fortress model need not endorse intrinsic value arguments. Moreover, the extent to which the BR model rejects the third position is a matter of degree. A proponent of this model may well believe in the global value of biodiversity and global responsibility for it, but yet also believe in working with local groups (especially resident communities). In recent years global NGOs, including the WorldWide Fund for Nature (WWF, except for the United States and Canada, where it remains the World Wildlife Fund), CI, and TNC have moved in that direction, thus making transitions from the fortress model to a BR model.

However, some caution should be exercised in interpreting these transitions. The normative framework presented earlier in this Section is based on foundational arguments about what we should do and believe, not on what works in practical contexts of devising conservation policy. The extent to which these norms can be translated into practice depends on many constraints no matter how well-justified the foundations may be. When conservationists adopt a particular policy because of these constraints, they would be acting because of prudential reasons, not normative ones. It is quite possible that global NGOs abandoned the fortress model because of a realization—buttressed by cases such as India's Project Tiger—that, in the post-colonial environment of the South, fortresses are no longer defensible (Dowie, 2009). In the case of WWF, in 2002, Jeanrenaud (2002) argued that it had changed its language to conform to the BR model without concomitant changes in institutional organization and power relations. However, WWF is widely credited to have made some progress since that time (Dowie, 2009) (as our case study will also illustrate).

3. The social ecology model

We turn to a more detailed exposition of the SE model and our focus continues to be normative rather than prudential. For brevity, we will restrict attention to the most basic assumptions of the model including those that are relevant to the case study of Section 4. There are eight such assumptions:

1. *Integration of humanity with nature*: There is continuity between human society and the ecological communities in which they are embedded (Seixas Simao and Berkes, 2003). Any distinction between cultural and natural values, goals, *etc.*, must be context-dependent, relying on operational needs of an analysis. In a context of malnutrition, enhanced productivity of an ecological community is a cultural value; in a context of restoration of bird breeding grounds, it may well be a natural value. What is more important is that the SE model denies the fortress model's implied dissociation between humanity and the rest of nature (Alessa et al., 2009). As such, the SE model is consistent with recent attempts to view environmental planning as an attempt to achieve resilience of social–ecological systems (Berkes et al., 2000; Folke et al., 2002; Walker et al., 2002; Walker and Salt, 2006; Chapin, 2009).
2. *Human welfare*: The basis for conserving natural values, including biodiversity, is human welfare including the welfare of future generations. Welfare means meeting not only the achievement of material needs (Blann et al., 2003) but also intellectual and spiritual needs (if these are distinguished, which depends on cultural context). Plans based on achieving resilience also accept human welfare as a goal (Walker and Salt, 2006).

3. *Contextualism of social institutions and values*: The institutions according to which communities are organized depend on local context, as do cultural and natural values (Folke et al., 2003; Chapin et al., 2009). In the case of concepts of biodiversity, this point was already illustrated in Section 2 when we emphasized the non-universality of what different groups choose to designate as bearers of natural value (and thus worth protection).
4. *Resident community privilege*: In decisions about the future of a habitat, resident communities have privileged status. They are not "mere" stakeholders: they are owners of the land. This is the most practically relevant foundational assumption of the SE model. Its basis is a claim of parity between the North and the South. We will use the United States to illustrate the point. Consider private property. The ESA is still applicable on such habitats but, nevertheless, this does not deny the owners' primary claim on the land and there is no question that the owners cannot be forcibly relocated. The SE model demands (by parity) that the same principle be applied to lands belonging to resident communities anywhere (with the important proviso that it recognizes the existence of much more complicated land tenure structures than simple private property, including communal ownership).
5. *Participatory decisions*: Contributing to group decisions is the privilege of all members of a community. This point is not controversial but deserves explicit enunciation because, in the context of Southern resident communities, it is typical for external agents to claim acquiescence from the community by reaching some "accommodation" with a real or perceived elite. Elite–based power structures often emerge within communities, especially in the presence of external economic incentives and, ironically, these structures sometimes mirror the power structures that keep these communities at the bottom of regional, national, and trans-national power structures.
6. *Pluralism about natural values*: As an operational definition for our context, we take as a natural value anything that promotes the enhancement of non-living environmental features or the promotion of the welfare of biota other than (but not necessarily excluding) humans. Resident communities of the South utilize natural resources in many ways and are often dependent on resources—such as food and firewood—directly drawn from the natural environment. Consequently, they require the conservation and enhancement of many natural values that cannot credibly be thought of as biodiversity: food, timber, and firewood productivity; erosion, flood, and storm control through vegetation enhancement; *etc*. The SE model embraces all of these.
7. *Legitimacy of cultural values*: Given the emphasis on the integration of humanity and nature, the SE model endorses the enhancement of cultural values as part of the goal of habitat management. This does not preclude criticism of particular cultural structures, for instance, discrimination based on gender or caste (genetic sub-group). The point is that there is no presumed sharp separation between nature and culture and there is none of the deification of nature that is an essential feature of the fortress model.
8. *Empiricism with respect to habitat management*: How habitats are best managed to promote natural values is an empirical question to be answered through data collection and analysis; *a priori* commitments are illegitimate (Clark, 2002). This is a factual assumption rather than a normative one but important to emphasize because the fortress model assumes without argument and evidence that human disengagement is the best—and perhaps the only (Terborgh, 1999)—method for biodiversity conservation no matter, where it is. The point is that this is an empirical claim and exceptions abound (Sarkar, 1999, 2005). The SE model asks for the question to be addressed empirically. Proponents of the BR model typically agree with the SE model on this issue.

Except for the last, each of these assumptions has a normative component; the last assumption captures perhaps the one point of scientific—rather than normative—disagreement between the fortress model and the SE model as well as, probably, between the former and the BR model.

There is much in common between the SE model and views promoted in the diverse array of studies that fall under the rubric of political ecology. We have not—in the interest of brevity, as noted earlier—listed assumptions of the SE model such as the importance of the analysis of power relations to understand the causes of habitat degradation which are also emphasized within political ecology. However, unlike most studies in political ecology, we acknowledge the importance of sound conservation science and our analysis of normativity draws on ethics rather than anthropology. Typically proponents of the SE model will also introduce other goals, most notably distributive justice. For some, what is distinctive about social ecology is that it adds explicit social justice considerations to political ecology's analysis of power structures. We do not disagree; from such perspectives, our treatment of social ecology will seem incomplete. But we have space limitations in a journal paper.

4. Case study: Kandozi activism in the Peruvian Amazon

4.1. Background

During the 1990s, national and international conservation experts, together with Perú's governmental authorities and civil society, extensively discussed a new conceptual framework for the management of protected areas (PAs) in the country. The result was a new legal framework which promotes private and local participation in PA management, incorporates PAs into economic and development contexts, and establishes a new relationship between local stakeholders (indigenous or not) and the state (Instituto Nacional de Recursos Naturales and Sociedad Peruana de Derecho Ambiental, 2002; Galvin and Thorndahl, 2005).

Strategic alliances between the state and local communities were incorporated as a necessary component of PA management; this is particularly relevant for the recognition of communal and indigenous rights and for the reinforcement of traditional local power (Chicchon, 2000; Galvin and Thorndahl, 2005; Instituto Nacional de Recursos Naturales and Sociedad Peruana de Derecho Ambiental, 2002). Under this new framework, the PA system included Communal Reserves as a PA category that aims to conserve biodiversity for the benefit of local communities. Natural resources in Communal Reserves were not under strict protection but their use was supposed to be regulated through explicit management plans. These plans were to be formulated and administered by local communities but only after prior approval by state authorities (Rodríguez and Young, 2000; Galvin and Thorndahl, 2005).

Although local communities were legally entitled to manage Communal Reserves, they did not have a legal title to their territories within them (or within any other category of PAs). Six of Perú's seven Communal Reserves were created after the new PA law was instituted in 1997 but only five of its 12 National Parks (with strict protection) were established after 1997. This is evidence that the new regulatory regime favored local participation and use of natural resources in PAs (Naughton-Treves et al., 1987). In principle, this development already marks a partial transition from the fortress model to a BR model based on Communal Reserves.

Indigenous groups were consulted about the creation of not only Communal Reserves but also of National Parks that would overlap their territories or be adjacent to them. The process of consultation had varied outcomes, depending on context. For example, after years of meetings and negotiations, the Ashaninka, Awajum, Ese'eja, Huambisa, and Matsiguenka indigenous groups agreed to establish PAs that overlapped their territories. The rationale for this was that they believed that PAs would protect their territories from large external development pressures, such as those for hydrocarbon or mining extraction (Chicchon, 2000). In contrast, the Kandozi and Achuar groups were against any type of PA because they want to secure their territories by acquiring legal titles without any form of state control.

This promising, though belated, recognition of traditional indigenous territorial rights was negated when the Peruvian government opened the Amazon and the Andean Highlands for hydrocarbon and mining exploitation by trans-national companies (Chicchon, 2000). The government began granting concessions for subsoil resources without prior consultations with resident communities and without taking into account whether the land was part of a PA. Only National Parks were exempt, and could not be exploited for subsoil resources. Moreover, in 2007, the García government attempted to reduce the area of the Bahuaja Sonene National Park to open the sector for hydrocarbon exploration. Consequently, local people no longer trust the government and have realized that living within a PA does not guarantee conservation of natural resources or their territories.

In recent years, this distrust has led to violent conflicts. A state of emergency was declared in Perú in June 2009 after thousands of Amazonian indigenous people protested against legislation introduced as part of Perú's Free Trade Agreement with the United States (Anaya, 2009). Indigenous peoples argued that these laws threaten their territories and their way of life because they facilitate investors to occupy their lands to extract hydrocarbons. Indigenous groups such as the Aguarunas and Huambisas took up arms in the city of Bagua to protect their territories and livelihoods from extractive companies and the situation remains fluid.

The case of the Kandozi is even more complex because they have been exposed to state control of their fishing resources, to contamination from oil exploration, and to conservation initiatives that aimed to alienate the natural resources that constitute their livelihoods. This case study details how they have resisted external interventions, have opposed PAs, and have decided to sustain and manage their own resources. It demonstrates the limitations of the BR model and brings into relief what the SE model can offer. The case study does not involve all the stages in Box 1, partly because it did not emerge from any attempt to apply systematic methods. However, the first five stages are clearly seen, as are the tenth (threat assessment) and thirteenth (implementation) stages.

4.2. Study area

The Pastaza fan, also known as the "Abanico del Pastaza" (Ramsar Convention, 2006), is the largest humid tropical alluvial fan in the world. It encompasses 60,000 sq km (54,000 sq km in Perú) and is situated on the eastern catchment of the Amazon River basin, draining from the Ecuadorian Andes to the Peruvian northwestern Amazon (Räsänen et al., 2006; Bes de Berc et al., 2005). In Perú, it has its boundaries roughly delineated by the Ecuadorian border to the north, the Marañón River in the south, the Pastaza River to the west and the Tigre River to the east (see Fig. 1).

The lower portion of the Abanico del Pastaza is a geological depression that floods for several month of the year. It is a combination of Andean rivers carrying volcanoclastic sediments with Amazonian black water rivers and ria lakes, that together form a patchy matrix that favor the development of diverse habitats (Kalliola et al., 1998, 1992; Kalliola and Flores-Paitán, 1998; Kvist and Nebel, 2001). The Peruvian portion (38,000 sq. km) of the Abanico del Pastaza has been internationally recognized by the Ramsar Convention because of its diverse wetland types (Ramsar Convention, 2006). Seven of the 20 types of continental wetlands in the Ramsar Convention's Wetland Classification System have been identified in the region.

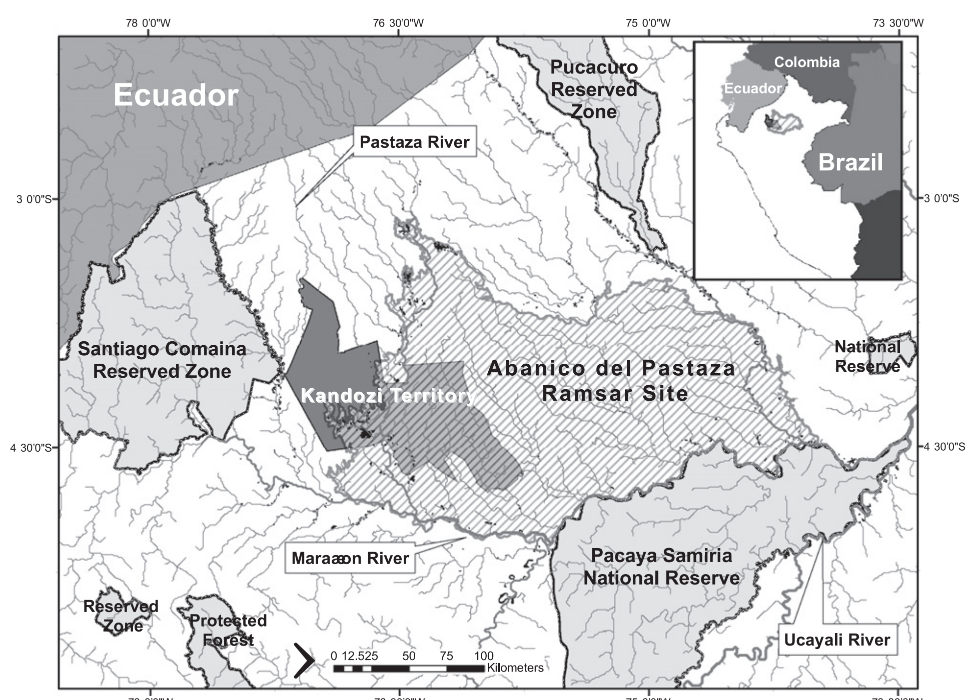

Fig. 1. *The Kandozi Area in Its Geographical Context.* Within Peru, the PAs are indicated by shading.

4.3. Biological and cultural diversity

The area has extremely high biodiversity, with a large number of threatened and endangered species including the black caiman (*Melanosuchus niger*), manatee (*Trichechus inunguis*), "charapa" turtle (*Podocnemis expansa*), spider monkey (*Ateles belzebuth*), giant river otter (*Pteronura brasiliensis*), and "piur" (*Crax globulosa*). There are some 261 bird species, 66 mammal species, 57 amphibian species, 38 reptile species, 292 fish species, 45 palm species, and at least 804 other tree and shrub species and forest types. Its forests include extensive and relatively well-conserved mixed palm forests and "mauritia" palm forest (*aguajales*), and such diverse flora species as cedar (Cedrela spp.), Ceiba species, and mahogany (*Swietenia macrophylla*). Many of these species are covered by CITES (19 listed in Appendix 1, 70 in Appendix 2) and some are included in the IUCN Red List (17 species) (Centro de Datos para La Conservación and WorldWide Fund for Nature–Perú, 2001).

The region is also important for its cultural diversity as it is home to a variety of distinct human groups, including the Achuar, Kandozi, Kichwa, Urarinas, and Cocama-Cocamillas. The different groups have historically struggled for rights over their traditional territories. Currently, their titles do not cover the totality of the traditional indigenous lands. In practical terms, this occurs because communities usually settle along the main waterways, leaving a large portion of their traditional territories in limbo. Indigenous people living in and around the Ramsar site have historically not been included in Perú's natural resource extraction policy and decision-making process. Current Peruvian legislation permits the concession of (untitled and titled) indigenous lands to oil companies or other extractive industries without consultation or compensation. Fees paid by oil companies to the federal government seldom benefit local communities directly (Greenspan, 2006).

4.4. Threats

The northern portion of the Abanico del Pastaza including the Achuar, Urarinas and Kichwa indigenous communities have been seriously impacted by petroleum extraction. Both governmental and non-governmental studies have provided ample evidence of severe pollution problems. These found that high proportions of indigenous people in the Corrientes River area have lead and cadmium concentration levels in their blood that are far above World Health Organization guidelines (Dirección de Ecologfa y Protección del Ambiente, 2006). For more than three decades, more than 60% of Perú's national oil production came from this area with environmentally unsound oil exploration and exploitation practices, outdated infrastructure and technology, and lack of adequate environmental regulation. The absence of proper environmental mitigation strategies continues to pose serious threats. Several freshwater ecosystems have been severely contaminated, necessitating the routine removal and appropriate disposal of contaminated soils and other severe restoration measures (Goldman et al., 2007).

The Kandozi occupy the southwestern part of the Abanico del Pastaza, where the Peruvian Government granted Block 4 with 900,000 h to Occidental Petroleum Corporation (OXY) in 1993. Though the concession significantly overlapped traditional Kandozi territory, resident communities were not informed about it. Granting concessions without informing communities or getting their consent was and continues to be the *modus operandi* of the Peruvian Government. In 1994, Kandozi communities only learned about OXY when the company entered the Chapuli river to start drilling an exploratory well. Despite Kandozi communities' opposition, OXY continued with its activities. The results of the exploration were negative and OXY stopped activity in the region in 1995. However, some time after it left, the first cases of Hepatitis B and Delta appeared in the area (Surralles, 2007). In 2003, the Ministry of Health declared the Pastaza district and the Kandozi territory as the area in the country with the highest risk for Hepatitis B and Delta (United Nations Children's Fund [UNICEF], 2005). (One critical difference with the northern area of the Abanico del Pastaza was that OXY's presence in the area was short and the environment was not damaged as had happened with Achuar lands.)

4.5. Kandozi history and habitat

The Kandozi indigenous group consists of approximately 3000 people, dispersed in about 40 communities. They were first encountered by colonialists in 1744 inhabiting the Huasaga River. Viewed as a belligerent group, they dominated both sides of the Pastaza river throughout the nineteenth century while living in isolation (Soriano, 2007). In 1748 and 1754 the Kandozi expelled putative Jesuit expeditions. In the early nineteenth century, they frequently attacked other explorers on the Pastaza river. During the 1920s they began to have less hostile but sporadic contacts with mestizos who traded rubber. However, they never allowed the mestizos to enter their territory; instead, they traded with them outside.

The Kandozi indigenous territory, which encompasses 9 116 sq. km, is located in the lower portion of the Pastaza river basin (Fig. 1) (Soriano, 2007). Lake Rimachi, inside Kandozi territory, is part of a series of interconnected lotic and lentic blackwater systems. It covers a surface area between 30 sq. km and approximately 79 sq. km depending on the season (Briceño, 2005; Surralles, 2007; Anderson et al., 2009). Its black waters mix with white waters from the Pastaza to create the conditions necessary for successful spawning of many fish species. As a result, the diversity of the ecosystems, flooding patterns, and the geochemical characteristics of this area are all relevant to Kandozi livelihoods and for ecological processes such as fish migration and reproduction. Lake Rimachi has a long history of active intervention by the Kandozi to manage fisheries because their communities rely on fishing not only for subsistence but also for monetary income generation (García, 2007). According to Kandozi fishermen, 21 commercial fish species are available in Lake Rimachi.

4.6. Conflict and resolution

In 1945, the Peruvian government established a Fishery Reserve in Lake Rimachi as a measure to conserve fish stocks, in particular, stocks of paiche (*Arapaima gigas*) and gamitana (*Colossoma macropomum*). However, instead of acting to prevent stock decline, state officials allowed mestizo commercial fishermen with large ships into the Lake and, for a period, restricted fishing only for consumption in military bases. In addition, officials prohibited the Kandozi from commercial fishing. State control of Lake Rimachi lasted until 1991 (see below) (Surralles, 2007). By 1990, populations of several fish species, including paiche, had noticeably declined (United Nations Children's Fund [UNICEF], 2005; Surralles, 2007; Anderson et al., 2009). Declining fish populations alarmed and seriously affected Kandozi communities because of their fisheries-based livelihoods and income generation. Consequently, in August 1991, Lake Rimachi became the center of a confrontation between the Ministry of Fishery and the Kandozi people who gathered in the community of Musa Karusha, at the mouth of Lake Rimachi.

Kandozi fishermen, in coordination with the Kandozi Federation (FECONAKADIP), and with legal assistance from the National Indigenous Organization (AIDESEP), took control of the Fishery Reserve's base office of operations in the Lake with the goal of forcing the Ministry of Fishery out of the region. They succeeded in that goal and from 1991 until 2004, the Kandozi did not have any direct contact with Ministry of Fishery personnel. After fishing activities in the Lake reverted to their control, the Kandozi succeeded in ensuring the recovery fish stocks by an internal agreement to stop commercial fishing for three years (United Nations Children's Fund [UNICEF], 2005; Surralles, 2007; Anderson et al., 2009).

Meanwhile, the Peruvian Freshwater Program of what was then called the World Wildlife Fund (WWF)–Perú Program started developing conservation plans for the Abanico del Pastaza after it was identified as a priority area for conserving the biodiversity of the Amazon River and Floodplains Ecoregion (World Wildlife Fund [WWF], 2005). Initially, the WWF's plan was to create PAs in Kandozi territory to conserve freshwater ecosystems. The Kandozi initially accepted the idea until they realized that they would not be able to continue acquiring legal land titles for their communities within official PAs. Subsequently, WWF–Perú and its donors (including the MacArthur Foundation), showing considerable flexibility, reconceptualized the project together with the Kandozi, and focused on natural resource management as the way to fulfil the original conservation goals.

As late as 2002, the Kandozi fished in the Rimachi without any control, external or internal. Commercial mestizo fishermen took advantage of them by buying fish at low prices and often trading back very expensive merchandise rather than paying money. The Kandozi needed to improve commercialization conditions of fish and forest resources, and needed to have some legal security about control of the Lake. Thus, a formal partnership was created between the Kandozi, WWF, and local NGOs to work towards formalizing Kandozi fishing activities and protecting freshwater resources through the creation and implementation of a fisheries management plan for Lake Rimachi (among other activities). The same year, the WWF promoted the designation of the area as a Ramsar site with support of resident indigenous communities. The Ministry of Foreign Affairs and the National Institute of Natural Resources requested the international recognition under the Ramsar Convention.

The Kandozi formed a fishing association which, with technical support from NGOs and the Ministry of Fishery, developed a management plan which has since been formally approved by all relevant governmental agencies. With that plan, the Kandozi can exclude mestizos from fishing in the Lake and can commercially harvest some fish species sustainably. They can also continue subsistence fishing for a greater number of species. This way of conserving fish stocks has so far been more effective than a PA in at least three ways: (i) the Kandozi have continued to acquire or increase the size of their land titles which, in turn, has increased their commitment to protect fish stocks over increasingly larger areas; (ii) the area of land controlled by the Kandozi is by now well beyond the capacity of the state to manage effectively as a PA; and (iii) since 2004, there has been a reduction in the volume of fish eggs collected and commercialized by the Kandozi and by mestizo fishermen in the region (Kalliola and Flores-Paitán, 1998; Anderson et al., 2009). The plan has made it possible that the Kandozi can "secure" their Lake while generating cash income and continuing with traditional fishing. It has also allowed the Ministry of Fishery to delegate control of the area to a local group which is almost certainly a more viable management plan in the long term than external control (Anderson et al., 2009).

4.7. Oil concessions and renewed conflict

Yet, while the Ministry of Fishery was promoting indigenous development based on the sustainable use of fish resources, and the Ministry of Foreign Affairs was committing the country to comply with the conservation of a new Ramsar site, the Ministry of Energy and Mining was promoting oil exploration and exploitation in the exact same area. The Ministry of Fishery delegated the control of fish resources to the Kandozi and formally recognized their right to the Lake Rimachi and their territory for fishing as a means of securing livelihoods for them. Almost simultaneously, the Ministry of Energy and Mining established and promoted petroleum blocks, and negotiated and signed contracts with petroleum companies before any consultation with the Kandozi communities that inhabit the overlapping area with the block (Greenspan, 2006).

At present, the Kandozi continue to reject petroleum exploration and exploitation in their territory. However, at least one oil block that overlaps their territory has already been granted under a exploitation contract. According to current Peruvian law, the government can grant concessions in any area, if the activity that will develop there is in the "national interest". For the government as a whole, petroleum extraction is apparently of greater interest than conserving a Ramsar site or conserving a healthy environment that sustains Kandozi livelihoods.

4.8. Reflections

In Perú designating a land parcel as a PA is no guarantee of real or sustained protection. Protected areas that are not National Parks may be developed for any purpose that is perceived to be in the national interest by state actors. But even National Parks are subject to manipulation if a government is sufficiently enthralled by financial incentives to delist or change the size of a National Park as the case of the Bahuaja Sonene National Park shows. Moreover, for all PAs, different governmental agencies had conflicting agendas some of which were inimical to conservation. This is not to say that such jurisdictional conflicts cannot be systematically adjudicated as they have in many other areas of the world. The point is that the fate of a habitat, if it is a PA, depends on externalities that change with political and economic trends.

In sharp contrast, local activism in support of habitat conservation and natural resources has had a record of success in Perú. We dealt with the case of the Kandozi here but the recent record of other indigenous groups in the Peruvian Amazon, and their successful resistance (so far) of the potential destruction of their habitats under the aegis of a Free Trade Agreement, suggests that *all* biodiversity conservationists—including proponents of the fortress model—should support these initiatives.

However, ultimately, these are prudential issues. The more important point is that the demands articulated by the Kandozi are normatively (ethically) legitimate: (i) control of their traditional territories; (ii) continuation of traditional livelihoods; (iii) sustainable use of their own resources while entering the regionally dominant market economy on their own terms; (iv) freedom from severe externally imposted health risks; and (v) maintenance of their habitats as cultural landscapes of their choice. We will turn next to the implications for the SE model.

5. Discussion and conclusions

Our analysis is consistent with the distinction drawn by Miller et al. (this issue) between the fortress model and those models that recognize the ethical salience of human needs and aspirations. However, we draw an additional distinction between the BR and SE models without which cases such as that of the Kandozi cannot be adequately analyzed: we would not have had a framework which could put the focus on the one issue that matter most to the Kandozi, legal titles to their territory. The BR model is neutral about power relations between external agents and resident communities in the control and management of habitat; the SE model explicitly privileges resident communities. The SE model captures, and perhaps even valorizes, what the Kandozi have demanded about the control and management of their territory.

As noted in the last Section, the basis for the SE model's endorsement of such local control is ultimately ethical. However, its prudential advantages should also be noted and, here, our case study supports suggestions for compromise and tradeoffs (McShane et al., this issue; Robinson this issue). Resident communities such as the Kandozi have a long history of resistance to degradation of their habitats by external agents, and have also demonstrated a willingness and the capacity to restore their habitat (the fisheries in the case of the Kandozi) through self-regulation after they had been degraded by external agents. It is only reasonable to expect—though it is by no means guaranteed—that continued control of habitats by such groups, and legal recognition of their tenure, would lead to persistent sustainable management and restoration of the habitats. This conclusion has general relevance beyond the case of Perú. Internationally, 85% of PAs are already occupied by resident communities (typically indigenous groups) and most new areas coveted by Northern conservationists have the same land tenure (Dowie, 2009). The SE model suggest that, in all such cases, *if* there is evidence that the habitat is flourishing under the present tenure regimes, local control should be legally institutionalized. The conditional nature of this recommendation should be emphasized: we are making no claim about any "ecologically noble savage" (Redford, 1990). If habitats are not flourishing under present tenures, we leave it open as to whether a BR model may be more appropriate. However, even in such cases, on the normative (ethical) ground of not denying fundamental human rights, we reject the fortress model.

How often do resident communities have the desire to manage their territories for natural values or the capacity to do so? This is contested territory. The first point to note is that this is an empirical question to be answered through systematic data collection and analysis, not *a priori* pronouncements of the value of wilderness or anecdotal impressions, especially those reported by individuals with vested interests in one or other model of natural habitat management (Sarkar, 1999). However, it is hard to deny the growing evidence of a clear correlation between cultural and biological diversity because different cultures manage their habitats according to different values and rules, and have to do so sustainably for either the culture or the habitat to persist (Nabhan, 1997; Sarkar, 1999; Colchester, 2000; Langridge et al., 2006; Alessa et al., 2009; Dowie, 2009). The SE model may have a large domain of application that we are only beginning to recognize.

Our description of the expressed goals of the Kandozi did not use the term, "biodiversity". As Nabhan (1995) has pointed out, while examples of sustainable resource management by resident communities are ubiquitous, there is no clear example of such a community managing "biodiversity" as conceptualized by Northern conservationists. We noted in Section 2 that even Northern conservationists do not have a single definition of biodiversity and, indeed, no such veridical definition is possible. But, in the case of the Kandozi, at least at first glance, two of their goals would seem to have the effect of protecting what may qualify as biodiversity under some definition, no doubt partly because of the engagement of WWF and other NGOs in the formulation of the management plans. First, one goal was to protect the entire habitat, and not some specific component of it—this would satisfy at least TNC's definition since it would include all regional habitat types (see Section 2). Second, an over-riding goal was to maintain stocks of a large number of fish species which, presumably, will have the effect of protecting an even larger number of diverse taxa. It is quite likely that, if we try to apply explicit concepts of biodiversity to habitat management by Southern resident communities, we will rarely fare any better. Perhaps the lesson to be learnt is that, when we think of protecting the richness and variety of biota, we should not necessarily think of it as protecting something of as recent vintage as "biodiversity".

Finally, the SE model does not put any premium on setting up protected areas to be distinguished from the surrounding landscape matrix. Here, it differs most markedly with the fortress model with its focus on National Parks. However, one unintended consequence of the fortress model is that land outside the National Parks often degrade much more rapidly than when there is continuity between different management regimes across a landscape (which is a hallmark of the BR model). Costa Rica provides a well-known example from the 1980s in which deforestation levels outside its exquisite National Parks were extreme enough to offset forest protection inside National Parks compared to several other neotropical countries (Sader and Joyce, 1988; Harrison, 1991; Faber, 1993). Rather, the SE model envisages integrative landscape management by resident communities. It does not exclude setting up protected areas but leaves such decisions up to local communities which must determine how to best manage their habitats. Whether this leads to less degraded landscapes than the fortress and BR models remains to be studied.

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