

Biodiversity and Systematic Conservation Planning for the Twenty-first Century: A Philosophical Perspective

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Summary

The concept of biodiversity, its introduction in conservation biology, and its evolution in the framework of systematic conservation planning, are analyzed. Attempts to quantify biodiversity and to find surrogate measures for it are described. It is shown that biodiversity originated as and remains a fundamentally normative concept. However, while attempts to reduce biodiversity conservation to the achievement of sustainability are misplaced, natural values other than biodiversity also merit promotion. Multi-criteria analysis can be used to capture necessary trade-offs between these values when they are in conflict. Moreover, given the possibility of trade-off analysis, socio-cultural values can also be integrated into habitat use decisions along with natural values. Thus biodiversity conservation can be integrated into more general framework of habitat use planning.

Keywords biodiversity, conservation biology, decision theory, history, multi-criteria analysis, surrogate, sustainability, systematic conservation planning, trade-offs

Essay

Article history

Received	30 May 2014
Accepted	07 June 2014
Published Online	12 June 2014
Editor	Prakash K. Paudel

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Citation: Sarkar S (2014) Biodiversity and Systematic Conservation Planning for the Twenty-first Century: A Philosophical Perspective. *Conservation Science* 2, 1–11

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Funding: None

1. Introduction

The conservation of facets of nature is age-old (Sarkar 2005). Most cases consist of the conservation of resources, e.g., the protection of forests by the Mauryan emperor, Asoka (299–237 BCE), of India so as to preserve the habitats of elephants for the imperial army (Sukumar 1984). Other cases involved the spiritual needs of a culture, e.g. the sacred groves of ancient Greece (Hughes 1994). After the Industrial Revolution in Europe led to massive transformations of natural habitat, conservation of natural values became part of the political agenda in many European countries and their former and present colonies. In the late twentieth century, these efforts led to the emergence of an organized discipline of conservation biology (Takacs 1996, Sarkar 1998). In the initial phase of conservation biology, in the 1980s and 1990s, efforts at conservation were widely viewed as being in conflict with economic development and, according to many (though not all) of its proponents, of greater ethical significance (Sarkar 2005). During the last decade, these assumptions have been brought into question (Margules and Sarkar 2007, Sarkar and Frank 2012). The purpose of this paper is, first, to review these developments and show how they fit into the framework of systematic conservation planning for the protection of biodiversity and, second, to propose an agenda for the future that negotiates the conflicts that have arisen.

Section 2 notes that the term “biodiversity” is of recent vintage, dating back only to the late 1980s, and introduced in a particular socio-cultural context; it gained popularity because of the synergy between its use and the establishment of conservation biology as an institutionalized discipline in the global North. Section 3 notes the problems with defining “biodiversity”—problems that require resolution in the practice of planning for its conservation. However, Section 4 argues that biodiversity conservation should not be reduced to sustainability attainment but also notes that biodiversity does not exhaust all natural values that deserve protection. Moreover, some of these values may be in conflict with each other. Section 5 shows how multi-criteria analysis, which is part of the standard toolbox of systematic conservation planning, allows the incorporation of trade-offs between these values. Moreover, Section 6 shows that the same strategy can be used also to negotiate between natural and other cultural values. Section 7 consists of some final remarks.

2. The Genesis of “Biodiversity.”

The neologism “biodiversity” was only coined by Walter G. Rosen at some point during the organization of the 21–24 September 1986 National Forum on Biodiversity held in Washington, D.C., under the auspices of the United States National Academy of Sciences and the Smithsonian Institution (Takacs 1996, Sarkar 2002).

The new term was originally intended as nothing more than shorthand for biological diversity for use in internal paperwork during the organization of that forum. However, by the time the proceedings of the forum were published (Wilson 1988), it had emerged as the title of the book. The Washington forum was held only shortly after the founding of the U.S. Society for Conservation Biology in 1985 which can be taken to have institutionalized an emerging new discipline (Sarkar 2002). Soulé’s (1985) manifesto for the new discipline of conservation biology and Janzen’s (1986) influential exhortation to tropical ecologists to undertake the political activism necessary for conservation appeared in 1986. A sociologically synergistic interaction between the use of biodiversity and the growth of conservation biology as a discipline occurred and it led to a reconfiguration of environmental studies with the conservation of biodiversity as a central concern.

The term ‘*biodiversity*’ immediately found wide use following its invention. As Takacs (1996, p. 39; italics as in the original) has pointed out: “In 1988, *biodiversity* did not appear as a keyword in *Biological Abstracts*, and *biological diversity* appeared once. In 1993, *biodiversity* appeared seventy-two times, and *biological diversity* nineteen times.” The first journal with “*biodiversity*” in its title, *Canadian Biodiversity*, appeared in 1991 (it changed its name to *Global Biodiversity* in 1993); a second, *Tropical Biodiversity*, appeared in 1992; *Biodiversity Letters* followed in 1993 (Sarkar 2005). Meanwhile conservation biology as the science with the explicit goal of conserving biodiversity emerged as a highly visible enterprise with considerable political appeal in Europe and neo-Europe. Primack (1993) published the first textbook of conservation biology in 1993, Meffe and Carroll (1994) followed with their comprehensive survey in 1994. As a perusal of any issue of this journal will show, by now, the use of “biodiversity” is pervasive—and has been so since the mid-1990s.

3. What Is Biodiversity?

But, what does “biodiversity” mean? In spite of the wide use of the term since the early 1990s there was initially little concern for precise definition even in textbooks and no consensus among those who claimed to practice conservation. Later, in surveys conducted in the mid-1990s, Gaston (1996b) and Takacs (1996) found little agreement among conservation biologists about what the scope of “biodiversity” was or even whether a precise definition was necessary. There was also no agreement among philosophers concerned with the practice of conservation biology. Sarkar (1998 2005) proposed a deflationary account, arguing that what the term meant should be customized to conservation decisions; Sterelny and Maclaurin (2008) defended a narrow scope with a focus on species richness; recently, Santana (2014) has proposed elimination of the term altogether.

3.1. The Assessment Problem

The trouble with eliminativism is its excessive ambition: it would require a complete restructuring of conservation practice with no recourse to “biodiversity.” While elimination may assuage the philosophical concerns of some of those who are bothered by the ambiguity and imprecision of the term, it is hard to view this as a sufficient benefit for radical reconstruction of the discourse of a discipline. At the other end, it has always been recognized that there is much more to biodiversity than species richness (Primack 1993, Meffe and Carroll 1994, Gaston 1996a); at the very least, there must be some attention to complementarity (or differences—see below) between the biotic entities intended for conservation (Vane-Wright et al. 1991, Sarkar 2002, 2005).

There are other four well-known problems with using richness as a core component of a measure of biodiversity that Maclaurin and Sterelny (2008) do not address:

1. The contribution of ‘complementarity’ mentioned earlier (Kirkpatrick 1983): Suppose sites are ranked for their biodiversity values on the basis of richness, say, of species. Two sites may have very high rank but have virtually the same species in them. A third may be ranked lower in richness than either but have different species. Consequently a combination of the first and third (or second and third) may have higher total richness than the first two together. A measure of biodiversity should reflect such differences, which came to be called complementarity (Vane-Wright et al. 1991), which is what new entities at a site contribute to an existing set of sites.
2. Biodiversity measures should reflect ‘equitability’ (Sarkar 2002, 2005): Consider two sites, both with species A and B present in them. Both have the same richness (of two). Now suppose that the first consists of 90 % A and 10 % B, whereas the second consists of 50 % of both. There is a clear sense in which the second is more diverse than the first that richness does not capture.
3. ‘Disparity’ should matter: the taxonomic distance between the entities is relevant to biodiversity value. An example from central Texas can help. Consider two areas, both with species richness of two. The first contains two insects (both implicated as vectors of Chagas disease), *Triatoma indictiva* and *Triatoma sanguisuga* which were only firmly reclassified as two separate species in the 1970s (Lent and Wygodzinsky 1979). The other (Comal Springs) contains the Comal Springs dryopid beetle, *Stygoparnus comalensis* and the Peck’s cave amphipod, *Syngobromus pecki*, both of which, besides being taxonomically distant, are endangered (Gibson et al. 2008). Richness does not capture the higher biodiversity value of the second site
4. ‘Endemism, rarity, etc.’ should matter: The same example as earlier from Texas underscores this point: the two species at Comal Springs mentioned there are endemic to the region.

This discussion here has used species as the appropriate unit of biodiversity but these arguments do not depend on that choice. They can be recast for any other unit (e.g., other genera or even adequately distinguished [character] traits).

Even deflationary accounts require supplementation (Section 3.4). The critical point is that, if the goal is to conserve biodiversity (and resources must be allocated for this task), there must be publicly recognizable techniques to assess success. This means that there must be “indicators” for biodiversity and these must be: (i) quantifiable; and (ii) estimable in practice (Williams et al. 1994, Sarkar 2002, Sarkar and Margules 2002). These requirements immediately eliminate one popular—and, initially, very attractive—proposal: that biodiversity be defined as diversity at all levels of taxonomic, structural, and functional organization. This definition cannot be operationalized (Sarkar and Margules 2002). There is no plausible way in which such a broadly characterized concept of biodiversity can be measured at even a local, let alone a regional or global level.

3.2 Systematic Conservation Planning

The assessment problem becomes particularly severe in the context of systematic conservation planning (SCP) which emerged as a central sub-discipline within conservation biology in the 1990s and 2000s (Margules and Pressey 2000, Cowling and Pressey 2003, Margules and Sarkar 2007, Pressey and Bottrill 2009, Sarkar and Illoldi-Rangel 2010, Sarkar 2012a). SCP is “a structured step-wise approach to mapping conservation area networks, with feedback, revision and reiteration, where needed, at any stage” (Sarkar and Margules 2007). Conservation areas are sites that are managed primarily for the persistence of biodiversity. Because conservation biology was founded at a time of perceived crisis due to accelerated loss of natural habitats (which, in turn, was believed to foster species’ extinctions), the optimal selection of conservation areas emerged as a critical problem for conservation efforts (Sarkar 2012a). Thus, systematic conservation planning became a core component of conservation biology.

The term “conservation area” replaced the older “reserve” in the 2000s for two reasons (Sarkar 2003): (i) The representation of biodiversity in conservation areas should be conceptually decoupled from the design of optimal policies for its management so as to ensure the persistence of biodiversity. Empirical results should determine what management practice is best; this may but may not involve treating a site as a reserve (Sarkar 1999). (ii) Because of the long history of conflicts

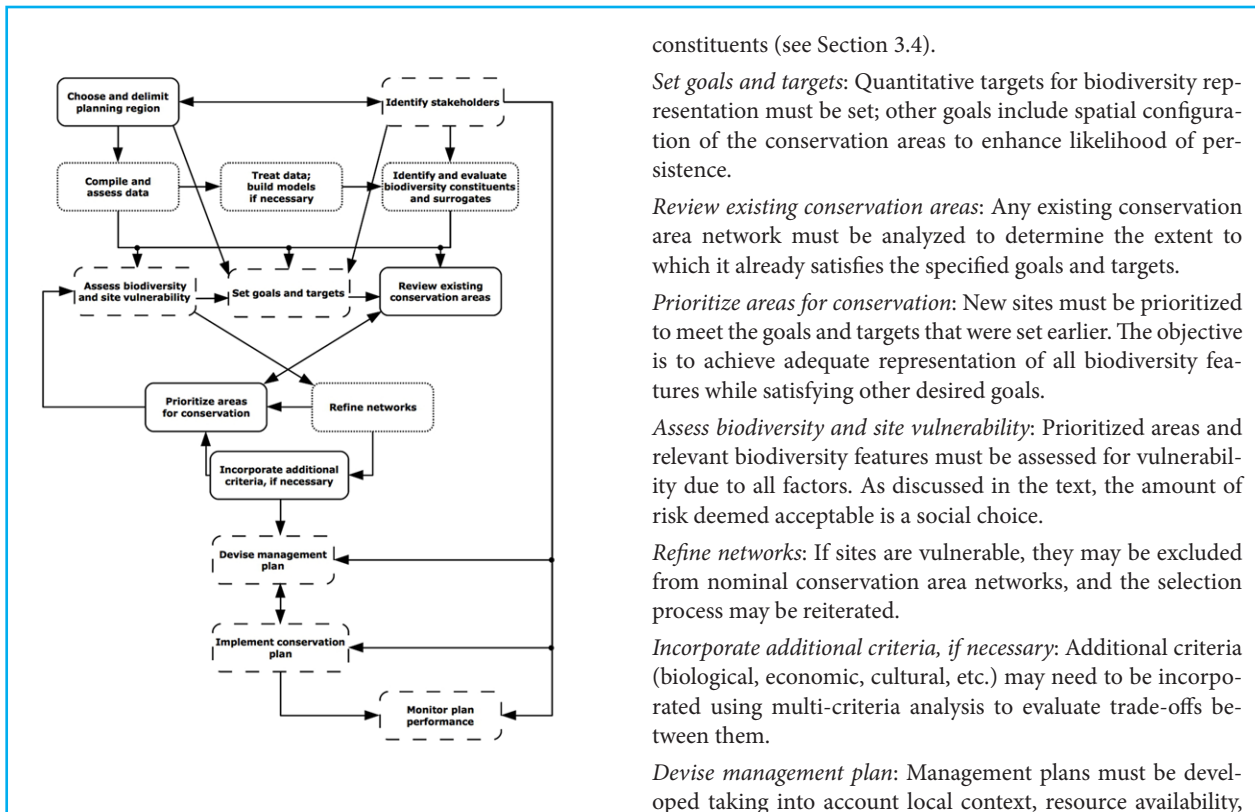


Figure 1: Stages of systematic conservation planning (SCP) (after Sarkar and Frank [2010]). Arrows indicate which stages directly influence others. A bidirectional arrow indicates interaction between stages. Only major influences are shown as there is potential for interaction between almost any two stages. Boxes with solid borders indicate that a stage is relatively well-understood (in terms of scientific capabilities); dotted borders indicate those that are fairly well-understood; and dashed borders indicate the least understood stages.

Choose and delimit planning region: Precise geographical boundaries of the planning region should be explicitly discussed and chosen—how boundaries should be drawn (e.g., whether they are based on political or ecological criteria) may raise ethical issues.

Identify stakeholders: Stakeholders include those who significantly affect or are affected by conservation plans—they have a legitimate stake in what happens. There could be feedback between this stage and almost any other stage.

Compile and assess data: Relevant biological, ecological, and socio-political data must be collected in a cost-effective manner.

Treat data; build models if necessary: Data treatment through statistical analysis is often required. Modeling is needed when treatment is insufficient to produce spatial data on relevant biological and socio-political factors.

Identify and evaluate biodiversity constituents and surrogates: Stakeholders identify biodiversity constituents which requires discussion of normative commitments as discussed in the text. Surrogates consist of quantitative estimators of biodiversity

constituents (see Section 3.4).

Set goals and targets: Quantitative targets for biodiversity representation must be set; other goals include spatial configuration of the conservation areas to enhance likelihood of persistence.

Review existing conservation areas: Any existing conservation area network must be analyzed to determine the extent to which it already satisfies the specified goals and targets.

Prioritize areas for conservation: New sites must be prioritized to meet the goals and targets that were set earlier. The objective is to achieve adequate representation of all biodiversity features while satisfying other desired goals.

Assess biodiversity and site vulnerability: Prioritized areas and relevant biodiversity features must be assessed for vulnerability due to all factors. As discussed in the text, the amount of risk deemed acceptable is a social choice.

Refine networks: If sites are vulnerable, they may be excluded from nominal conservation area networks, and the selection process may be reiterated.

Incorporate additional criteria, if necessary: Additional criteria (biological, economic, cultural, etc.) may need to be incorporated using multi-criteria analysis to evaluate trade-offs between them.

Devise management plan: Management plans must be developed taking into account local context, resource availability, etc.

Implement conservation plan: The management plan must be implemented for conservation to work. Consultation with local stakeholders is imperative for both ethical and practical reasons.

Monitor plan performance: Plan performance must be monitored to devise responses as necessary for adaptive management into the future.

between “top-down” conservation efforts and local residents, particularly in the South (Guha 1997, Jacoby 2003), including the ethically unacceptable creation of conservation refugees (Dowie 2009), it was ethically and politically important to recognize that creating reserves that require the removal of human residents or the cessation of human activity is not the only option for conservation. Thus, using “conservation area” rather than “reserve” is also a matter of ethical sensitivity.

The stages of systematic conservation planning are described in **Figure 1**. In the context of this paper, what is most relevant is that SCP proceeds by selecting surrogates for biodiversity and representation targets (in networks of conservation areas) for them. (The term “surrogate” will be discussed in detail in Section 3.4 below.) It then attempts to solve either of two problems: (i) the minimum area problem: how to satisfy the representation targets for biodiversity surrogates in a minimal set of conservation areas; or (ii) the maximum representation problem: given a maximum total area that can be

set aside for conservation, how to choose a set of conservation areas below this constraint so as to maximize the representation of biodiversity areas up to that target (Sarkar et al. 2004, 2006). For either case, the quantifiability and estimability criteria for biodiversity must be met. Further progress is impossible without disambiguation and more precision about “biodiversity.”

3.3 Normativity

To make such progress, what needs explicit recognition is that biodiversity is a normative concept. This was sporadically recognized by conservation biologists (e.g., Gaston 1996b) since the 1990s but was also routinely emphasized by philosophers (Callicott et al. 1999, Norton 2003). Two types of normativity should be distinguished in this context: (i) weak normativity which indicates that biodiversity is the desired end of the goal-directed discipline of conservation biology; and (ii) strong normativity which sees biodiversity conservation as an ethically salient goal. While some of the discussions among conservation biologists seem to accept only weak normativity (Gaston 1996b, Takacs 1996), much of the discussion in philosophy (especially environmental ethics) has presumed strong normativity.

That philosophical discussion has often focused on the ethical basis for biodiversity conservation (Norton 1986, 1987, Sarkar 2005). There has been a long-standing debate about whether biodiversity has intrinsic value or only anthropocentric value; the philosophical consensus at present seems to be against intrinsic value attributions because of a variety of conceptual problems (Norton 1987, Sarkar 2005, 2012b, McShane 2007). However, the resolution of this philosophical dispute is not of concern in this paper—what will matter in Section 3.4 is the normativity of biodiversity, however it should be philosophically justified.

That normativity also accounts for an oddity in the history of conservation biology. Within ecology there had been extensive discussion of measures of ecological diversity since the 1950s, generally in the context of whether there is a relation between such diversity and the stability of ecosystems (Sarkar 2007, 2010). Yet, during discussions of measures for biodiversity in attempts to satisfy the quantifiability and estimability criteria mentioned earlier, there seems to have been no attempt to co-opt these measures of ecological diversity for conservation biology. Yet, as Magurran (2003) eventually pointed out, one of the standard measures of the biodiversity value of a site, viz., complementarity (what it would add to an existing set of conservation areas [Section 3.1]) is a form of the measure, β -diversity, which has long been part of the repertoire of theoretical ecology. A likely explanation is that, because of the normative nature of their discipline, conservation biologists believed themselves to be embarking on a project

markedly different from traditional ecology (e.g., Soulé 1995); consequently it did not occur to them that the conceptual resources of ecology would be of much use in their efforts.

3.4 Constituents and Surrogates

How should biodiversity be assessed? The idea that surrogate measures will be necessary is fairly old (Austin and Margules 1986). However, invoking surrogates is question-begging in a fundamental way: to decide whether some entity is an adequate surrogate for biodiversity, the question of what biodiversity is must be independently answered prior to assessing the adequacy of the surrogate.

Typically surrogates are chosen with estimability in mind, that is, in such a way that there is a credible possibility of carrying out the necessary quantitative measurements in practice. Sarkar and Margules (2002) distinguished between “true” and “estimator” surrogates. The former are supposed to represent biodiversity in general in an SCP exercise. They could be entities such as all species, all assemblages, or all traits (Margules and Sarkar 2007). However, in many (perhaps most) circumstances measuring all true surrogates is not feasible under typical time constraints on the formulation of conservation plans. Estimator surrogates stand in for true surrogates.

Sarkar (2002) argued that the adequacy of true surrogates is not an empirical question; rather it must be settled by convention. (This came to be called a deflationary account of biodiversity [Santana 2014].) However, the adequacy of estimator surrogates is an empirical question. Techniques of surrogacy analysis were devised to answer this question. The crucial test was whether conservation plans devised using estimator surrogates were concordant with those devised using true surrogates (Sarkar et al. 2005). In a positive development for conservation planning, Sarkar et al. (2005) showed that small sets of (abiotic) environmental variables were adequate surrogates for several true surrogate sets. This was a desirable result because such environmental data sets are available for almost the entire world. It meant that systematic conservation planning would not stall because of a paucity of available data (as The Nature Conservancy (TNC) once used to argue [Redford et al. 1997]).

Nevertheless, viewing the adequacy of true surrogates purely as a matter of convention was unsatisfactory—perhaps too deflationary. Sarkar (2008) argued that the true surrogates were the desired constituents of biodiversity (the term “true surrogate” was to be replaced by “biodiversity constituent”) and that their choice should be based on normative considerations. This move finally establishes consistency between how the concept of biodiversity originated in a normative context and how it

is used in the practice of conservation biology. But it has a debatable implication: what biodiversity should be is relativized to the context, typically to the norms of the relevant culture. But, unlike what has been urged by Santana (2014), the concept of biodiversity is not eliminated; and this normative concept is less deflationary than Sarkar's (2002) original account.

Moreover, Sarkar (2012b) went on to make the concept even less deflationary by adding three necessary adequacy conditions for any biodiversity constituent set: (i) that the entities be biotic; (ii) that variability in these entities should be captured in the chosen set; and (iii) that taxonomic spread be important. These adequacy conditions were supposed to capture much of the use of "biodiversity" in contexts of its conservation. The second one is critical in ensuring that the focus remains on diversity; this is why attempts to eliminate "biodiversity" (as argued for by Santana [2014]) are misplaced. (A fourth adequacy condition, that not all biodiversity constituents be immediately useful resources, was regarded as desirable but not necessary.)

A variety of constituent sets satisfy these conditions. For instance, reflecting a cultural norm in the United States, viz., the Endangered Species Act of 1973, biodiversity is taken to be the set of species at risk of extinction in that country. The influential non-governmental organization, Conservation International, takes it to be the set of at-risk species in the Red List of the International Union for the Conservation of Nature and Natural Resources (IUCN). The Nature Conservancy takes it to be the set of habitat types (species assemblages). The European Union follows a similar strategy. Australia and India accept both some species and some habitat types. Some traditional cultures opt for sacred groves and all entities in them—even this apparently non-standard choice satisfies the three adequacy conditions (Sarkar 2012b).

The important point is that, in each case, the focus of conservation efforts is a cultural choice reflecting the relevant norms. This does not mean that there is no rational basis for these choices; rather, it means that the choice must be deliberated upon like any other normative policy choice. (The policy choice, in this context, is that of what should be conserved.) In a sense, then, the concept of biodiversity carves out those aspects of natural variety that a culture deems important enough to conserve. (However, Frank [2013] has argued that this concept of biodiversity still remains too deflationary; Sarkar [2014b] provides a response.)

Biodiversity constituents can include processes, for instance, the migration patterns of the Monarch Butterfly (*Danaus plexippus*) in north America (Brower and Malcolm 1991), seasonal migrations of wildebeest in Africa (Brower and Malcolm 1991), and the synchronous flowering of bamboo in India (Bahadur 1986), all of which are in danger of disappearance (Sarkar 2002, 2005). The flexibility of this normative framework for

defining biodiversity constituents embraces the different ways in which natural variety may become culturally salient enough to protect.

Biodiversity constituents may also include charismatic species—thus costly attempts to conserve the Bald Eagle (*Haliaeetus leucocephalus*) in the United States can be justified even though the species is common in Canada and was in no risk of global extinction. Perhaps most importantly, from this perspective, biodiversity is in general not a global good to be legislated upon (or otherwise subject to decisions) by external players. Rather, it reflects local contextual values about what part of nature merits protection.

4. Sustainability and Natural Values

The 1992 Earth Summit in Rio de Janeiro (technically the United Nations Conference on Environment and Development) produced a Convention on Biodiversity in which biodiversity was viewed as an aspect of sustainability. In effect, this means that biodiversity conservation was being reduced to the achievement of sustainability. If the discussion of the concept of biodiversity in Section 3.4 has any merit, any such reduction is illegitimate: the conservation of adequate biodiversity constituent sets neither requires nor necessarily achieves sustainability. However, this argument depends critically on the definition of sustainability. The issue has been discussed at length by Sarkar (2012b). Suffice it here to note that if the definition of sustainability allows even limited fungibility (i.e., the replacement of some resources by others) sustainability may not require the retention of every element of every constituent set. Thus, for example, an endangered species may become dispensable if its functional roles, if any, can be performed by some replacement species in the relevant community. (A very strong notion of sustainability, which rejects fungibility [Norton 2005] can avoid this problem.)

A more plausible strategy is to embrace a non-reductive pluralism of natural values, "those that promote the persistence and increase of non-human biota or enhance non-anthropogenic aspects of the physical environment" (Sarkar 2012b, p. 21). Natural values include both biodiversity conservation and the achievement of sustainability. Sarkar (2012b) has pointed out that there has been surprisingly little philosophical discussion of natural values even with all the attention that environmental issues have received over the last half-century. Besides biodiversity, other species' welfare, fidelity to some reference type, ecological service, and wild nature have all been suggested as natural values. These typically have subordinate components. For species and for ecosystem, the objective of biodiversity include sub-objectives related to risk status (vulnerability), rarity, richness, adaptation to habitat (suitability), domicile (including endemism), and cultural role. The welfare of

other species is promoted, for instance, by animal rights activists. Fidelity is a typical goal of ecological restoration (Higgs 2003) (though it is problematic [Sarkar 2011, Garson 2014]). Ecological service includes productivity (including food security), environmental security, and myriad environmental services. Wild nature subsumes both wilderness and wilderness—its value may well be aesthetic.

What emerges from this discussion is the salience of a spectrum of natural values. From this perspective, biodiversity conservation is embedded in a variety of desirable environmental practices not all of which are reducible to the attainment of sustainability. The next question to be addressed is how these are accommodated to the context of systematic conservation planning.

5. Multi-criteria Analysis

Natural values may be in conflict with each other. For instance, food production may be in conflict with both biodiversity conservation and wilderness preservation and the latter two may be in conflict with each other (Callicott and Nelson 1998, 2008, Sarkar 1999). Systematic conservation planning (SCP) can incorporate these conflicts by co-opting techniques from multi-criteria analysis (MCA) developed within decision theory (Moffett et al. 2006, Regan et al. 2007, Sarkar 2012b). Such attempts go back to the late 1980s (Anselin et al. 1989, Mendoza and Sprouse 1989, see Moffett and Sarkar [2006] for a review). Even if biodiversity is the only relevant natural value for a decision, there may be conflicts between sub-objective, e.g., between the conservation of different at-risk species. Central Texas provides a telling example (Sarkar 2011): two endangered bird species, the Black-capped Vireo (*Vireo atricapilla*) and the Golden-cheeked Warbler (*Dendroica chrysoparia*) have intersecting ranges but disjoint habitat requirements (scrubland versus mature Ashe juniper [*Juniperus ashei*] woodlands, respectively). MCA becomes relevant to any regional conservation plan.

MCA begins with (i) the identification of alternatives (policy options or decisions, such as which sites to include in a conservation area network) and, especially, (ii) of all the values or objectives that are relevant to the formulation of a conservation plan and (iii) the relations among these objectives (Sarkar 2012b). Using these relations, these values are structured into what is known as an objectives hierarchy (OH). This hierarchy is constructed through elicitation of preferences (and other relevant information) from stakeholders.

Since OH construction requires some care and has rarely been explicitly attempted in systematic conservation planning, it will be useful to provide some detail (see, also, Keeney and Raiffa 1993). The process involves the iterated use of variants of two questions: “What are the objectives of the plan?” and “Why is this objective

important?” The first question provides the set of relevant values or objectives; the second establishes the structure of the hierarchy. The elicitation process stops at the top (of the hierarchy) when fundamental objectives are reached. These objectives are those that are ends-in-themselves; that is, there is no further answer to the second question (Keeney 1992). Objectives lower in the hierarchy (sub-objectives) are important for what they contribute to the fundamental objectives. Elicitation can stop at the bottom when the lowest-level objectives are such that they can be associated with measurable attributes.

In the context of conservation plans, biodiversity as specified by its constituents is almost always a fundamental objective with the protection of at-risk species, endemic species, etc., as sub-objectives. However, each of the natural values mentioned in Section 5 (e.g., wild nature) can also play the role of fundamental objectives (e.g., wild nature with sub-objectives of wildness and wilderness [Sarkar 2012b])—this is the point of not viewing some values as reducible to others. (The contrast here is with the view rejected there, that these other natural values are sub-objectives with only sustainability as the fundamental objective).

Once an OH has been constructed, a variety of methods can be used to rank (and, whenever possible, put weights) on the attributes, again through the elicitation of preferences from stakeholders. Compounding methods can then be used to rank the alternatives themselves (Moffett and Sarkar 2006). These range from simple use of rankings (e.g., excluding dominated alternatives [Sarkar and Garson 2006]) to the use of techniques of multi-attribute value and utility theories that are a generalization of standard economic analysis (Moffett et al. 2006). In multi-attribute utility (or value) theory, a linear utility (or value) function is then computed to rank the alternatives quantitatively (Moffett et al. 2006). (A variety of more ad hoc methods, including the Analytic Hierarchy Process (Saaty 1980) have also often been used (e.g., Regan et al. 2007) but are not recommended (Moffett et al. 2006, Sarkar et al. 2006). The relevant point here is that MCA provides a systematic methodology for incorporating myriad natural (and other—see Section 6) values into conservation decisions.

5. Towards Integrative Habitat Planning

The techniques of MCA described in Section 5 can also be used to incorporate socio-political values along with natural values. In fact, most uses of MCA in SCP have incorporated economic factors besides biodiversity representation (Moffett and Sarkar 2006). The reason is straightforward: biodiversity conservation is not the only ethically salient use of a site. There are sound ethical prerogatives for other uses for sites including, in particular, human habitation and production, which can be in conflict not only with biodiversity conservation

but also with the enhancement of other natural values (Borgerhoff-Mulder and Coppolillo 2004, Margules and Sarkar 2007).

Since the early 1990s, there have been many suggestions that there are “win-win” solutions for these situations which simultaneously achieve all objectives—these suggestions include debt for nature swaps, extractive reserves, community-based conservation, and integrated conservation and development projects (Ambler 1999, McShane et al. 2011, Sarkar and Montoya 2011). But the last fifteen years has seen the emergence of general skepticism about the success of such initiatives (Oates 1999, Terborgh 1999, Hutton et al. 2005). Much of this criticism is based on extensive monitoring of results and cannot be dismissed as irrelevant—perhaps, what is more surprising is that “win-win” solutions were ever expected even in a large minority of cases. However, most critics (e.g., Oates 1999, Terborgh 1999) have advocated a return to what has been derided as “Fortress Conservation” (Brockington 2002) and is, in fact, ethically indefensible (Sarkar 2012b).

In this context, it is critical to recognize that the purpose of MCA is to induce trade-offs between conflicting objectives. The use of MCA almost always presumes that there are no “win-win” solutions, that compromises must be made—and that this is so not always because of any illegitimate intentions on part of some stakeholders but because the different objectives are fundamentally in conflict. Rather than a return to a Fortress Conservation model, MCA shows that these trade-offs can be incorporated into a conservation plan. The rational course to pursue here is to find the best compromise—thus MCA emerges as central not only for the conservation of biodiversity through SCP, or biodiversity along with other natural values, but of all integrative habitat planning.

The challenge for the future is to develop an integrated protocol in which MCA plays a fundamental role and which takes into account all ethically sound aspirations of all legitimate stakeholders. For a given region, at the very least there must be decisions that optimally allocate sections for biodiversity conservation and the promotion of other natural values, for human habitation, as well as agricultural and industrial production. Given the widespread degradation of natural habitat, the promotion of these natural values (including biodiversity conservation in many contexts) will require habitat reconstruction at least in the form of conventional ecological restoration (Sarkar 2011). Protocols for restoration that are related to SCP have already begun to be devised (e.g., Wilson et al. 2011).

SCP would form part of a protocol for integrative habitat planning, but perhaps only a minor part in many area where challenges to human well-being will require much of the habitat to be designated for consumptive human use. The protocol would even have to

consider problems such as the location of undesirable facilities (landfills, potentially polluting industrial units, etc.) along with the location of conservation areas and wilderness preserves. There has been some work on the problem of locating hazardous facilities (e.g., Erkut and Neuman 1989, Colebrook and Sicilia 2013) but none that integrates these problems with those relevant to biodiversity conservation. Integrative habitat planning is at the stage of development that SCP was in the 1990s before a well-developed framework first emerged (Margules and Pressey 2000).

7. Final Remarks

In his 1986 exhortation to tropical biologists Janzen (1986, p. 306) observed: “If biologists want a tropics in which to biologize, they are going to have to buy it with care, energy, effort, strategy, tactics, time, and cash... . Within the next 10 -30 years[,] . . . whatever tropical nature has not become embedded in the cultural consciousness of local and distant societies will be obliterated to make way for biological machines that produce physical goods for direct human consumption.” The discussion of Section 6 also highlighted how biodiversity conservation should be integrated with other aspects of human life. However, whereas Janzen’s concerns seem to have been primarily pragmatic (with a focus on what must be done to achieve success at biodiversity conservation), the concern of Section 6 was on deeper ethical questions.

The use of MCA obviously puts conservation and integrative habitat planning exercises within the ambit of decision theory (Sarkar 2012b). However, SCP was always, fundamentally, a decision-theoretic protocol; and minimum area and maximum representation problems were decision problems since the purpose of optimization was to find the best alternatives (which sites to include in putative conservation area networks). Decision theory provides a range of formal techniques for rational decisions of which MCA (Section 5) is an example. However, a point that deserves emphasis is that these formal techniques must be deployed critically (Sarkar 2012b): they should not become a substitute for rational deliberation between stakeholders (Sarkar 2014b). An example will illustrate this pitfall. As noted in Section 3.3, there continues to be an ongoing philosophical debate on whether biodiversity has intrinsic value. Some recent papers (see, e.g., Colyvan et al. 2010) have argued against such an attribution because intrinsic values cannot be adequately quantified (e.g., through incorporation into a utility function to be maximized). Thus, according to this argument, intrinsic value attributions can have no proper role in environmental decisions. This is a seriously misplaced invocation of formal decision theory.

As Callicott (2006) has pointed out, it is routine

in a democratic society to reason about options that are not incorporated into utility functions or otherwise quantified. Policies are made about them, for instance, about the compensation due to a family from the death of a member due to negligence. Presuming human life to be intrinsically valuable, this process involves reasoning about intrinsic values without quantifying them. A critical attitude towards decision theory is intended to guard against the sort of dubious reasoning displayed in the argument (in the last paragraph) against intrinsic value attributions. To make the point sharper: decision theory does not define rationality (or “good” policy making); rather, formal decision theory can be fruitfully deployed in circumstances when the assumptions of its framework are met. Decision theory captures part of rational deliberation. (For further discussion, see Sarkar [2012b, 2014a,B]).

Even the choice of biodiversity constituents, because it is a normative choice, can be viewed as a decision problem. The critical difference between this choice and the other ones within SCP (and integrative habitat planning) is that there is no reason ever to expect that the choice of biodiversity constituents can even partly be solved algorithmically. Rather, what has been said here strongly suggests that the choice of biodiversity surrogates is entirely a matter for rational deliberation rather than computation. Biodiversity conservation simply cannot avoid rational ethical deliberation—philosophy thus has a central role in the conservation planning (and integrative habitat planning) of the future.

Acknowledgements

For discussions over the years, thanks are due to Jim Dyer, David Frank, Trevon Fuller, Justin Garson, Chris Margules, Anya Plutynski, and Bob Pressey.

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Biography

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