Chapter 18

What Should “Biodiversity” Be?

Sahotra Sarkar

Abstract This paper argues that biodiversity should be understood as a normative concept constrained by a set of adequacy conditions that reflect scientific explications of diversity. That there is a normative aspect to biodiversity has long been recognized by environmental philosophers though there is no consensus on the question of what, precisely, biodiversity is supposed to be. There is also disagreement amongst these philosophers as well as amongst conservationists about whether the operative norms should view biodiversity as a global heritage or as embodying local values. After critically analyzing and rejecting the first alternative, this paper gives precedence to local values in defining biodiversity but then notes many problems associated with this move. The adequacy conditions to constrain all natural features from being dubbed as biodiversity include a restriction to biotic elements, attention to variability, and to taxonomic spread, as well as measurability. The biotic elements could be taxa, community types, or even non-standard land cover units such as sacred groves. This approach to biodiversity is intended to explicate its use within the conservation sciences which is the context in which the concept (and term) was first introduced in the late 1980s. It differs from approaches that also attempt to capture the co-option of the term in other fields such as systematics.

18.1 Introduction

Many commentators have noted that the term “biodiversity” is of very recent vintage even though biodiversity conservation has become one of the best-known components of both popular and technical discussions of environmental goals today (Takacs 1996; Sarkar 2005, 2017a). The term and associated concept(s) were only

S. Sarkar (✉)
University of Texas, Austin, TX, USA
Presidency University, Kolkata, India
e-mail: sarkar@austin.utexas.edu

introduced in the context of the institutional establishment of conservation biology as an academic discipline in the late 1980s. The introduction of the term is usually attributed to Walter G. Rosen at some point during the organization of a 1986 National Forum on BioDiversity held under the auspices of the United States National Academy of Sciences and the Smithsonian Institution (Takacs 1996; Sarkar 2002).

Originally “biodiversity” was only intended as a shorthand for “biological diversity”; by the time the proceedings of the forum were published as an edited book (Wilson 1988), the new term had been promoted to become its title. The BioDiversity forum was held shortly after the founding of the U.S. Society for Conservation Biology in 1985 (Sarkar 2002). Soulé’s (1985) manifesto for the new discipline of conservation biology and Janzen’s (1986) exhortation to tropical ecologists to undertake the political activism necessary for conservation had appeared in the previous two years. A sociologically synergistic interaction between the use of “biodiversity” and the growth of conservation biology as a discipline then occurred and it led to a reconfiguration of environmental studies so that the conservation of biodiversity became a central concern. Conservation biology, starting in the 1990s, was conceptualized as the goal-oriented discipline devoted to the protection of biodiversity. Soulé (1985) drew a powerful analogy between conservation biology and medicine; biodiversity was the analog of health.

The existence of a goal engenders a corresponding norm for evaluating whether an action contributes to that goal and, in many contexts, of assaying the extent to which it does so. All the major programs for biodiversity conservation, viz., conservation biology (Soulé 1985), conservation science (Kareiva and Marvier 2012), and systematic conservation planning (Margules and Sarkar 2007), acknowledge the normative component of biodiversity conservation. Not surprisingly, many environmental philosophers have followed suit in treating biodiversity as at least partly a normative concept (Callicott et al. 1999; Norton 2008; Sarkar 2008, 2012b).

But not all. Some philosophers (e.g., Maclaurin and Sterelny 2008), following the lead of many biologists (see Gaston 1996b and Takacs 1996), have treated biodiversity as if it were a purely scientific concept bereft of normative content. That perspective has led to a wide variety of scientific (more accurately, *scientistic*) definitions of biodiversity, each disputed, and with no prospect of resolution of these disputes. The persistence of these disputes has led to many deflationary accounts of “biodiversity” (e.g., Sarkar 2002) as well as proposals to eliminate the term completely (e.g., Morar et al. 2015; Santana 2017). These varied approaches have recently been reviewed by Sarkar (2017a) and that discussion is very briefly summarized in Sect. 18.2 of this paper.

Section 18.3 turns to the core purpose of this paper: a defense of normativism in defining biodiversity. Any such defense must address the question: whose norms? Global norms invoked by Northern conservationists must be pitted against the local norms of communities whose livelihoods are often threatened by biodiversity conservationists’ interventions. Section 18.3 traces the ideological underpinnings of global normativism, then rejects it, and critically endorses the use of local values to

define biodiversity. But endorsing local values is hardly unproblematic. Section 18.4 examines the problems that beset local normativism.

Accepting normativism does not mean rejecting the use of science any more in biodiversity conservation than it does in healthcare practices. For biodiversity, a partial synthesis is possible. Section 18.5 argues that a rich tradition of discussions within biology of what constitutes biodiversity can be used to lay down adequacy conditions that constrain the latitude available to normative definitions of biodiversity. It also lays out how this synthetic proposal, integrating values and (ostensibly value-free) technical science, can be used in the practice of conservation. Section 18.6 consists of some final remarks.

18.2 Approaches

Sarkar (2017a) has recently distinguished four approaches to defining biodiversity:

1. *Scientism*: Definitions falling under this rubric claim to use non-normative criteria to define and quantify biodiversity. Three such criteria have most often been deployed: richness, difference, and rarity. Each criterion has been used not only singly but also in conjunction with the others. Richness, measured by the number of units, is probably what most users of “biodiversity” have in mind when the term is not explicitly defined. It has also been partly or wholly explicitly defended by Gaston (1996a) and Maclaurin and Sterelny (2008). Difference, interpreted as complementarity, or how many new biodiversity units are introduced to those already present in an entity (such as an area or a community), has been contrasted to richness and promoted by proponents of systematic conservation planning (Sarkar 2002; Sarkar and Margules 2002; Margules and Sarkar 2007). Rarity, interpreted as endemism, along with richness has formed the basis for identifying biodiversity hotspots (Myers 1988; Myers et al. 2000).

The main problem with these attempts, pointed out by critics such as Santana (2017), is that there seems to be no possible potential resolution of the disagreements between proponents of the different scientific definitions of biodiversity. Difficulties abound: for instance, even within ecology it has long been recognized that richness alone cannot be an adequate characterization of diversity because it does not take equitability into account (Sarkar 2007).¹

Efforts to decide between scientific definitions of biodiversity inevitably end up requiring the use of extra-scientific criteria. For instance, proponents of complementarity argue that its use is preferable to richness as a characterization of biodiversity because of the following argument: Consider three potential conservation areas, *A*, *B*, and *C* of which only two can be prioritized. Let *A* have

¹ Consider two ecological communities, *A* and *B*. Let *A* consist of 90 % species μ and 10 % species ν . Let *B* consist of 50 % species μ and 50 % species ν . Both *A* and *B* have richness 2 (assuming species are the relevant unit). Yet, there is a clear sense in which *B* is more diverse than *A*. Richness does not capture that sense.

richness 100, *B* have richness 90, and *C* have richness 50. If diversity is to be characterized as richness, the diversity ranking of the three areas would be $A > B > C$ and choosing the best two would mean choosing *A* and *B*. However, suppose that *A* and *B* have 80 units in common. Then *A* and *B* together would contain 110 units. Now suppose that *A* and *C* have 30 units in common and *B* and *C* have 5 units in common. Then *A* and *C* would contain 120 units and *B* and *C* would contain 135 units. Thus the richness-based choice of *A* and *B* is the worst choice for biodiversity representation even if we use total richness as the relevant criterion for the biodiversity content of the prioritized set of conservation areas! This leads to the principle of complementarity (Vane-Wright et al. 1991; Sarkar 2012b): a new conservation area should be prioritized from the available ones on the basis of how many new units it adds to what is already present in those that have been chosen earlier.² The relevant point here is that the argument assumes that only two of the three potential conservation areas can be prioritized. Science does not supply this assumption. Its provenance is the existence of some resource constraint that must be respected.

Consider another choice: should richness or endemism or both be a component of biodiversity? Richness appears natural but, as seen earlier, its use is fraught with problems. How about endemism? We may opt for it out of concern for the rare and unusual. The point, though, is that these are no longer scientific claims. We have moved on to talk about values, what aspect of natural variety we deem most worthy of conservation, that is, there has been a transition to an analysis of norms. These cases are typical: extra-scientific considerations are necessary to adjudicate between conflicting scientific definitions of biodiversity.

2. *Eliminativism*: The failure of scientism in the definitional enterprise has led to one extreme response: proposals to eliminate the use of the term “biodiversity” altogether. Such a position has been forcefully argued by Morar et al. (2015) and Santana (2017). However, such a response would only become plausible if there is no other alternative to scientism. The rest of this paper argues that there is a plenitude of other available options. Suffice it here to note that banning “biodiversity” in current environmental discourse would be a daunting task and require efforts that, presumably, even eliminativists would accept as being better used to ensuring conservation in practice.³
3. *Deflationism*: Eliminativism as a response to the failure of scientism was preceded by a weaker strategy of deflationism. A strong form of deflationism was an assumption that, not only was there no fact of the matter about what biodiversity is, but that how it should be defined depends on local contexts, and can be gleaned by studying the practices of conservation biologists, for instance, what

²Note that this choice does not guarantee that the total richness (that is, the number of unique species) would be maximized. In the example earlier, it would lead to the choice of *A* and *C* rather than *B* and *C*.

³For more details of these arguments, see Sarkar (2017a). Meinard, Coq, and Schmid, (Chap. 17, in this volume) give a different perspective on why eliminating “biodiversity” or even allowing it to remain irreducibly vague would lead to problems for the practice of conservation.

is being optimized when areas are prioritized for conservation (Sarkar 2002; Sarkar and Margules 2002).

Strong deflationism was problematic for a variety of reasons, most notably perhaps because it seemed to leave no role for explicit discussion of how biodiversity should be defined, even in a given context. It was replaced by a weaker form in which normative discussion of what merits conservation determines what constitutes biodiversity (Sarkar 2008). But this takes us to normativism.

4. *Normativism*: Normativism will be developed in some detail in Sect. 18.3. What motivates this set of definitions is the recognition that the preservation of natural variety is a desirable social goal. For more than a generation, environmental ethicists have argued about the proper warrant for the admissibility of such a goal without reaching consensus (Norton 1987; Sarkar 2005) but, as environmental pragmatists have argued (e.g., Minter and Manning 1999), these intractable foundational disputes are almost always beside the point in the practical contexts that determine how a conservation policy is formulated and whether it succeeds or fails. For environmental pragmatists, what is of paramount importance is achieving agreement on practical courses of action, shelving foundational disputes in favor of policy achievement. What matters in such contexts is to map, evaluate, and critically engage the values of legitimate decision-makers. These values are not determined by scientific inferences drawn from biological data though those data may—and should—inform the values of the decision-makers. What is critical is a community’s vision of the future it desires including but not limited to its perception of its proper role in the natural world. Natural variety is one of those values and the one that is reflected in *biodiversity*; but biodiversity need not be the only natural value. Given this motivation, it remains to develop normativism more systematically. That discussion begins by moving beyond these assertions to arguments designed to establish that biodiversity must be a normative concept. In line with environmental pragmatism, there will be no further attention to foundationalist concerns in this paper.

18.3 Normativism

There are three loosely related arguments that aim to show why biodiversity must be a normative concept. To motivate these arguments consider what is perhaps the most general scientific definition: biodiversity is the variety of life at all levels of structural, taxonomic, and functional organization. As Gaston (1996b) has documented, many biologists have defended similar definitions (e.g., McNeely et al. 1990; Wilson 1992; Johnson 1993). Is this what *biodiversity* means? If so, it does not seem plausible that biodiversity is the goal of conservation for at least two reasons: (1) There is the venerable ethical principle, *ought implies can*. Can all of biodiversity as defined above be conserved? Ecological communities left undisturbed

lose species diversity through competitive exclusion. Evolving populations lose genetic (that is, allelic) diversity through natural selection. Conserving all such diversity is in practice impossible; (2) Is all such diversity in principle a desirable target of conservation? The human skin hosts thousands of microbial species though interpersonal variability is not as high as in the gut which hosts millions (Grice et al. 2009). Should we feel an imperative to conserve all the microbial diversity on the human skin or gut? Bacterial pathogens are rapidly evolving diversity to generate resistance in response to innovation in antibiotics designed to contain them. Other pathogens have shown similar, if less spectacular, responses to drugs. Should such diversity also merit active conservation?

The first argument for normativism begins with the assumption that concepts should be understood against the historical context of their introduction and use.⁴ For biodiversity that context is the establishment and institutionalization of conservation biology as an academic discipline. As noted earlier, programs for conservation have always accepted the goal-orientation of the project, and the existence of that goal endows biodiversity with an irreducibly normative aspect. Proponents of conservation biology from the 1980s fundamentally disagree about goals with proponents of systematic conservation planning from the 2000s and, especially, the new conservation science from the 2010s (Kloor 2015) but they all agree with the goal-orientation of conservation. In most cultural contexts, pathogen variability is seen as removed from “biodiversity” with its attendant positive connotation.

The second argument builds on the first. As a result of the goal-orientation of conservation, biodiversity has always been used with a positive connotation. It consists of those aspects of biotic variety that should be conserved. That does not necessarily include all of natural variety. Though the rhetoric of contemporary political discourse often suggests otherwise, not all diversity is positive (Sarkar 2010). A society with extreme economic disparities is more diverse than one that is more egalitarian; but it certainly is not better. A population with both healthy and sick individuals is more diverse but less desirable than one that has only healthy individuals.

The third argument notes that, by the time “biodiversity” was introduced in the 1980s, there had been a generation-long tradition of defining and studying diversity within ecology (Sarkar 2007). Much of this work was spurred by a central theoretical hypothesis of ecology dating back to the 1950s, that diversity begets stability of ecosystems. While both the empirical and theoretical status of this claim continues to be debated today, by the mid-1980s its exploration had led to the formulation of a large variety of diversity (as well as stability) measures. These measures and the

⁴This claim is open to philosophical dispute: for instance, adherents of a hard distinction between the context of discovery and context of justification, etc., may deny this assumption (perhaps most famously developed by Mach in his study of physical concepts in the late nineteenth century). Those who view science through the lens of analytic metaphysics and study concepts through intuition and abstraction may also deny it. These issues will be left for another occasion. Suffice it here to note that core analytic methodologies of concept formation (for example, Carnapian explication) accept the relevance of the pragmatic context of conceptual innovation (Carnap 1950).

associated concepts they were supposed to quantify, in contrast to biodiversity, did not display normativity in their use. It is telling that this body of work was entirely ignored by conservation biologists attempting to define biodiversity in the 1990s and since. The most plausible interpretation of this lack of interest in the existing work on ecological diversity is that they viewed their own normative enterprise of designating aspects of natural variety for protection as distinct from these earlier ecological efforts. Thus, scientism was irrelevant to that enterprise. But, then, what requires explanation is why the explicit statements of definitions of biodiversity from biologists, as recorded by Gaston (1996b) and Takacs (1996), are almost always scientific. Perhaps the explanation lies in the discomfort scientists often feel about explicit normative discussions—but this suggestive explanation is no more than sociological speculation at this point (but see Wolpe 2017).

18.3.1 *Global Heritage*

For biodiversity, who should set the relevant norms? In the present context this questions amount to asking who determines what aspects of natural variety should be protected. Here conservation efforts have been marred by serious ethical problems reflecting the structural inequities between the global North and the South. Conservation biology was first academically institutionalized in the United States and its agenda reflected the agenda of what has forcefully been criticized from the South as “radical American environmentalism” (Guha 1989). Soulé and his immediate followers had no hesitation in importing their values to the South, at one point arguing that the U.S. federal legal restrictions be circumvented to allow purchase of land for conservation in the South (Soulé and Kohm 1989): “Land acquisition is a very specific need ... The National Science Foundation should view land purchase and maintenance in exactly the same way that it views the purchase of a piece of fancy machinery ... *If there are legal barriers to direct acquisition of land in other countries by U.S. government agencies, then alternatives such as grants to such countries for the establishment and management of research reserves should be explored*” (p. 89; emphasis added). Available aid money would be better spent satisfying the desires of conservation biologists than, for instance, improving livelihoods of local people: “A potential funding source would be Public Law 480 programs which are currently operating in many developing countries” (p. 89).

If Soulé’s strictures were imperialist proclamations, Janzen (1986) endorsed the missionary position when he urged: “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 (depending on where you are), whatever tropical nature has not become embedded in the cultural consciousness of local and distant societies will be obliterated.... We are the generation [that must] devote [its] life to activities that will bring the world to understand that tropical nature is an integral part of human life” (p. 306). Wilson (1992 and elsewhere) joined many others in declaring biodiversity to be a global heritage. The efforts of Northern

conservationists were codified in various documents emerging from global agencies, most notably, the 1992 Rio Convention on Biodiversity.

But claims of global heritage require careful analysis and, when required, systematic deconstruction. Beyond bland assertion, what makes some natural feature or cultural artefact a *world* heritage? As we shall see there is no pat comfortable answer. Global heritage claims typically promote intervention by politically powerful external agents on decisions affecting the habitats of local residents who may have no interest in these global concerns. Moreover, these claims may not even be backed by any legitimate tangible material interest of these external agents—think of protecting a historical ruin just because of its age or a tropical rainforest because of its species richness.

The salience of these issues is borne out by looking at some particular cases: Was it wrong for the Taliban to destroy the Buddhas of Bamiyan? If so, why? And who decides? What gives the so-called international community—which is hardly a community of equals—a legitimate basis for questioning what a community in Afghanistan decides to do with some cultural artefacts present in its domain through no choice on its part? There is no reason to doubt that the strong feelings generated by the destruction of these statues probably reflects some defensible trans-cultural values. But what are they? How can they be spelt out and legitimized? How do these values serve the interest of the international community? Why do these interests override those of the local community? These questions have not received the attention they deserve. To return to the concern of this paper, turn to a biodiversity-related analog (Bevis 1995): Was it wrong for the Malaysians to log the lowland rainforests of Borneo? Why? And who decides? And so on. In this case there is an additional level of complexity. By and large, the local communities in Borneo were resistant to logging (Bevis 1995). The Malaysians opting for development were mainly economic and political elites from the mainland with the required power. The so-called international conservation community, largely activists from Europe and Australia, adopted and possibly manipulated the communities' concerns. But no one bothered to spell out whose heritage the great forests of Borneo were. And why. No matter how strongly we feel about these cases, the answers are not obvious.

Scholars have argued that concepts of heritage emerged in Europe in synchrony with the emergence of nation-states. Meskell (2014) puts it: "Intimately connected with the Enlightenment project, the formation of national identity relied on a coherent national heritage that might be marshaled to fend off the counter claims of other groups and nations" (p. 218). By the nineteenth century, in the late colonial context, the concept of heritage had begun to be applied across national boundaries, especially into the colonies. However, a concept of supranational cultural heritage only began to be formulated after World War I with tentative attempts at its legal codification originally under the auspices of the League of Nations (Boes 2013; Gfeller and Eisenberg 2016).

Full-fledged self-conscious efforts for global heritage designation and protection began with the post- World War II onset of the decolonization era and the formation of the United Nations Educational, Scientific and Cultural Organization (UNESCO)

in 1945 (Gfeller and Eisenberg 2016). Claims and designations of global heritage emerged in tandem for both cultural artefacts and natural features. Arguably, especially through the Northern domination of UNESCO and other global agencies, they served to maintain Northern control of these entities in the post-colonial South even after decolonization had brought direct control to an end, for instance, when UNESCO’s director Julian Huxley proposed setting aside large areas of central and east Africa as reserves (Huxley 1961; see Adams and McShane 1992 for a critique). (There will be other African examples below.) What is striking is that, beyond implicit appeals to claims of importance for some supranational group of individuals, no argument was advanced to codify why some feature is a global rather than, say, a national heritage; this is a problem that has only recently begun to receive attention (Di Giovine 2015). Instead of argument, attributions of global heritage status have systematically relied on bold assertions by proponents and demands for acquiescence on the part of those who may otherwise have resisted the globalization of their resources.

The first campaign to draw transnational attention to an ostensibly global heritage feature focused on Egypt, starting in the late 1950s, after President Nasser’s modernization plan for the country included construction of the Aswan Dam. The project envisioned the submersion of a large number of historic sites and monuments of the Nile Valley, perhaps most notably the Great Temple of Ramses II at Abu Simbel. The plan generated vocal opposition from archaeologists and historians, mainly from Europe; their rhetoric suggested that Egyptians were not legitimate stakeholders in decisions about the fate of these sites (Boes 2013). Though the nationalization of the Suez Canal and his neutrality in the Cold War hardly made Nasser a popular figure in the West, conservationists were able to co-opt him to their campaign in the late 1950s. In 1960 UNESCO undertook an ambitious rescue project of relocating the monuments at risk to higher elevations. Nasser was applauded for recognizing a “right to heritage.”

Parallel to the developments around Aswan, two German environmentalists, the father and son team of Bernhardt and Michael Grzimek initiated a global campaign for designating the Serengeti Plain of Tanganyika as a global heritage and “saving” it through formal protection and exclusion of local human use. The core component of their campaign was the creation of the documentary, *Serengeti Shall Not Die*, in which they explicitly and controversially drew an analogy between African wildlife and European historical monuments.⁵ Immensely successful, the documentary transformed discussions of the global status of the natural heritage of the South. To continue with the Aswan parallel, shortly afterwards, and this time in India, conservationists from the North, supported by a local elite consisting largely of hunters, co-opted Prime Minister Indira Gandhi to launch Project Tiger in 1973 (Mountfort 1983) in spite of local problems due to tiger-human conflicts. There will be more on Project Tiger below.

⁵The German Filmbewertungsstelle Wiesbaden (FMW) dubbed this an “impermissible equation” (*unerlaubte Gleichsetzung*) and its request for the caption’s removal captivated op-ed pages in the Federal Republic of Germany with discussions of censorship—see Boes (2013) for more detail.

The normative claims of conservation biology fall into this tradition and are based on the assumption that biodiversity is a global heritage. That is what makes it possible for Soulé to demand the acquisition of land in the South for the benefit of Northern conservationists. Janzen is gentler: he only wants to proselytize and convert the perceived heathens in the name of the global deity that is biodiversity. Indeed, it is commonplace for Northern conservationists to propose policies for distant lands in the South and to demand action (Dowie 2009).

For instance, 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 the radical environmental organization, 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).

To drive home the point being made, consider a hypothetical example originally constructed by Sarkar and Montoya (2011). Central Texas 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 U. S. Endangered Species Act (ESA), and then to delineate critical habitat and develop habitat conservation plans (as required by law) have long been controversial and have often led to ugly confrontations between landowners and conservationists (Mann and Plummer 1995). Now, imagine that an environmentalist from Mongolia decides to come to Texas, claim expertise on desert landscapes and cave ecology (perhaps justifiably), and demand that prime real estate around the capital city of 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 denying the legitimacy of local sovereignty over resources or we are not. We are either accepting the legitimacy 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, Northern conservationism, as argued earlier, are continuing colonial attitudes and policies in the South (see, also, Guha 1997).

The critical normative issue here is that of parity. What one community—whether it be Northern conservationists or Mongolian desert experts—values should not be transferred without consent to the habitats of other communities. When we couple this normative claim with the realization that a definition of biodiversity is context-dependent in the sense that the valuation of biological resources varies over space (Escobar 1996), then we must turn to local values.

18.3.2 *Local Values*

Recall that normativism views biodiversity as consisting of entities that merit protection. What is most relevant to the present discussion is that, in practice, different groups have made different choices (Margules and Sarkar 2007). Let us begin with governmental agencies and the big non-governmental organizations (derisively dubbed “BINGOs” by Dowie (2009)) that dominate large-scale biodiversity conservation efforts.⁶ In the United States, most governmental agencies adopt endangered and threatened species as biodiversity units but that is because much of conservation policy is set in the context of the legal requirements of the Endangered Species Act (ESA) of 1973. The ESA envisions protection of both animals and plants, includes subspecies under its purview, but excludes “pest” insects. In contrast, The Nature Conservancy (TNC), one of the best-known BINGOs, uses habitat types defined by characteristic ecological communities. Conservation International (CI), another BINGO, 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 implicitly presume that the extinction of every species that is admissible (excluding insect pest species) is equally (normatively) undesirable. TNC implicitly presumes that ecological communities are the bearers of value. The point is that these definitions embody cultural norms even though they are often presented as if they are universal and purely scientific definitions (Sarkar 2008).

Moreover, there are many other equally defensible choices. Sacred groves are widespread in South Asia, especially in the Western Ghats with evergreen wet forests and northeastern India, in the Eastern Himalayas. Forest communities of the Eastern Himalayas have maintained intact patches of cloud forest amidst an almost completely denuded landscape and have done so in spite of loss of most cultural associations with their sacred groves due to massive conversion of local populations to various Christian denominations starting in the mid-nineteenth century. In the state of Meghalaya, in many of these sacred groves not even deadwood can be removed.⁷ The extant 29 sacred groves occupy over 25,000 ha. These are evergreen forests on a landscape dominated by limestone. Much of the ecology of the region continues to be devastated through coal mining and quarrying for limestone besides swidden farming that has an increasingly shorter cycle (five years now compared to 30 years in 1900). Traditionally each village had at least one sacred grove but local traditions were largely destroyed by the Christian missionaries. Not one of the sacred groves has been systematically inventoried except for major tree species; but they are known to be particularly rich in amphibian species that have a high degree of microendemism. At least 18 IUCN Red List amphibian species occur in this

⁶For more detail and documentation of these examples, see Sarkar (2012b).

⁷Details are from Malhotra et al. (2007) and personal fieldwork.

region. Cave invertebrates in the many caves and fissures under the ground have not been inventoried at all.

Some of the best-known sacred groves of Meghalaya are in the Khasi Hills near the town of Sohra (formerly known as Cherrapunji) which, with an average annual rainfall of 11,430 mm, is one of the wettest places on Earth. (The honor of being the wettest place in the world now belongs to nearby Mawsynram.) Most groves are small and occur on the top of hills but the larger ones also include valleys and the streams that run through them. The most impressive grove here is at Mawphlang which is protected because it is supposed to be inhabited by the spirit “U Basa.” Its 80 ha contains at least 400 tree species; the fauna have never been inventoried. The protection regime (known as “Kw’Law Lyngdoh”) is severe: Mawphlang is one of the sacred groves from which even deadwood removal is not permitted. The land around the grove is severely degraded.

The complete protection of entire ecological communities may be uncommon even though sacred groves occur throughout the South, especially in sub-Saharan African countries, most notably Ghana and Kenya. In most African countries, sacred groves target a single species or small set of species. Many cultures around the world value individual species in other ways (e.g., as totemic species) that may be of symbolic value or associated with religious practices. Some communities value entire forests. Vermuelen and Koziell (2002) report the case of the Irula hunter-gatherers, a semi-nomadic tribe from Tamil Nadu state of southern India. The tribe is well-known for its association with snakes, both in catching them and in treating snakebites. What this community values is reflected in how they choose a site for settlement. First, they assay a forest for medicinal plants, then snakes, then animals hunted for food or money (rats, rabbits, mongoose, wild cats, etc.). The assessment is complex. The size of animal populations matters and is assessed using the density of footprints. Ecological associations between vegetation type and animals are taken into account (for example, rabbits with *arugampul* (*Cynodon dactylon*), that is, Bermuda grass which, despite its name, originated in West Asia). Typically, in a twist opposite to conventional ecology, animals are taken as indicators for plants. The persistence of forests is critical to the survival of the Irula way of life.

However, this divergence of values 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 crosscultural dialectics of engagement, disagreement, and change. Moreover, as we shall see in Sect. 18.5, we may adopt 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 2012b). These adequacy conditions will allow a partial synthesis of

scientific insight and local values. But science will play a subsidiary role: even these adequacy conditions have to be culturally debated.

18.4 Problems with Local Normativism

Since a form of local normativism is being endorsed here, it behooves us to recognize and pay particular attention to potential problems. There are at least four of these.

18.4.1 Problems of Scale

The last section contrasted local values with global heritage claims about biodiversity. The designation “global” is clear enough in most contexts, referring to Earth as a whole. But “local” is far from clear: it could vary from a community defined by a municipality (or perhaps an even smaller spatial unit) to a nation-state. (Nation-states, in turn, can vary in size from the Vatican with a population of a few hundred or Lichtenstein with a few ten thousand, to China or India each with over a billion.) A few nation-states are ethnically almost homogeneous; while some cities alone embrace scores of culturally distinctive ethnic groups. Is there a natural scale at which biodiversity should be defined or at which conservation measures enacted? The former seems implausible and the latter, as we shall see below, is problematic.

As if to mimic this problem, biotic features that are typically held to merit protection also vary in spatial scale (or extent). In central Texas, microendemic salamanders sometimes have their range restricted to a single neighborhood of a city. The Barton Springs Salamander (*Eurycea sosorum*) and the Austin Blind Salamander (*Eurycea waterlooensis*) both have habitat limited to Barton Springs in the middle of the city of Austin. The Devil’s Hole Pupfish (*Cyprinodon diabolis*) is endemic to a single cavern-like habitat in Nevada, United States, and has the smallest known habitat of a vertebrate species, just 0.008 ha (or 80 sq. m.) at the surface (Reed and Stockwell 2014). At the other spatial extreme, the endangered tiger (*Panthera tigris*) ranges from South Asia through Southeast Asia to Siberia (with a large gap at present, though not historically, in China) even after it has lost more than 90% of its habitat during the twentieth century. Earlier it was also present in parts of West Asia.

Different cultural concerns and values may be dominant at different spatial scales. In the case of the two salamander species just mentioned, the International Union for the Conservation of Nature and Natural Resources (IUCN) Red List, the global standard for risk designation, identifies them as “Vulnerable” but this designation is largely irrelevant to their future since the IUCN has negligible influence on conservation efforts in the United States. More pertinently, the United

States Fish and Wildlife Service (USFWS) designates them as “Endangered” which affords them protection under the ESA. So does the state of Texas in its own assessment of risk for its native species. Most importantly, the protection of both salamander species has strong support within the city government of Austin and this support gets translated into actions by city agencies to maintain their habitat. The Barton Springs Salamander, in particular, is woven into the fabric of the city’s cultural life. For those who view such endangered species as important components of biodiversity, this is a happy situation.

In contrast, the situation with the tiger is much more complicated. Globally, few species have dominated the consciousness of individuals for centuries as the tigers. About 70% of the world’s tigers live in India (Gibbens 2017). At the national level, since the 1970s, tiger conservation has been a priority as exemplified by the 1973 launch of Project Tiger. Since 1972, the tiger has been India’s National Animal (replacing the Asiatic Lion, *Panthera leo persica*, a subspecies of which the only extant population is also found in one state, Gujarat, in India). At the local level, conservation is not so simple. Tigers, as predators, often target cattle and other economically important domestic animals. They sometimes prey on humans, especially when habitat degradation and conversion, accompanied by a decrease in their non-human diet options, brings them into close contact with humans. In some tiger habitats, such as the mangrove swamps of the Sunderbans in eastern India and Bangladesh, tigers have long been positively embedded into local culture (Montgomery 1995). In many other tiger habitats in South Asia, human-tiger conflicts have led to local hostility (Gadgil and Guha 1995; Gibbens 2017).

For instance, between 2007 and 2014, in an area near the Chitwan National Park in south-central Nepal, local inhabitants intentionally killed four tigers (Dhungana et al. 2016). In India, local attitudes have been further confounded by the forced dislocation of tens of thousands of resident humans (though accurate numbers are hard to come by) during the process of the creation of Tiger Reserves under the auspices of Project Tiger (Sarkar 1999, 2005). It would come as no surprise that tiger conservation may not be welcome for communities living adjacent to tiger habitats. In fact, local resentment in India sometimes allows tiger poachers to hire local villagers to help them successfully evade anti-poaching efforts using local knowledge; there have even been acts of arson against parks and reserves by villagers adversely affected by their establishment under the aegis of Project Tiger (Gadgil and Guha 1995). Local values in many of these villages will likely not enshrine the protection of as hallowed a conservationist icon as the tiger in India. Returning to our definitional project, tigers would not necessarily be enshrined as a component of biodiversity. What is required are negotiations and tradeoffs between conservationists and victims of tiger depredation.

18.4.2 *Conflicts Between Hierarchical Levels*

The ambiguity of “local” shows the potential for conflicts between entities at different levels of the political (or cultural) hierarchy from communities through cities, districts, provinces, and the nation-state. These conflicts bear upon choices of a place embedded in different levels of this hierarchy. So, a locality is not only accountable to its community or city values, but also to those of the various regions of which it is a part including the nation-state that may well centralize the most relevant power for nature protection. Returning to the problem of tiger conservation in India, local communities suspicious of tiger conservation are typically pitted against conservationists at every other level of government.

The tiger case is hardly unique. In the late 1980s and early 1990s conservation efforts in central Texas were dominated by programs to protect multiple species besides the salamanders mentioned earlier. These included two migratory bird species, the Golden-cheeked Warbler (*Dendroica chrysoparia*) and the Black-capped Vireo (*Vireo atricapilla*) both of which were eventually declared as endangered by the United States Fish and Wildlife Service (Mann and Plummer 1995). Typically, such a declaration must be accompanied by the designation of “Critical Habitat” for the persistence of the species which imposes some limits on habitat use and transformation. Especially in the case of the Warbler, potential designation of Critical Habitat would have affected a wide swath of central and southern Texas. Opposition from ranchers was such that it is believed to have played a role in the defeat of incumbent Democrat Ann Richards to Republican George W. Bush in the gubernatorial election of 1994 (in spite of a promise by USFWS not to designate any Critical Habitat in a forlorn attempt to save the election for Richards). At the height of the conflict, ranchers explicitly promoted the decimation of endangered species. For these ranchers and much of rural Texas from where they came, these species would not form part of natural values that they would have chosen to protect. Yet, many of the same areas have a long history of private conservation of land and wild areas for a variety of reasons including game management for hunting.

18.4.3 *Conflicts Between Localities*

Conflicts occur not only across levels of a hierarchy in which a place may be embedded but between places across space. Returning to our well-worne case of tiger conservation, efforts at the national level throughout South and Southeast Asia were for a long time in conflict with China (where, perhaps, a few wild tigers persist) because of a demand for tiger body parts in a set of practices dubbed traditional Chinese medicine. In Southeast Asia many local communities (for instance, in

Borneo) value their forests which are viewed as cheap sources of timber in neighboring societies such as Japan (Bevis 1995). There are several species that are protected in their home range because they are perceived to be at risk but categorized as undesirable aliens elsewhere (Marchetti and Engstrom 2016).

It is not being suggested here that these conflicts—across geographical scales, within a hierarchy, or across localities—cannot be resolved. Resolution requires tradeoffs between different groups. Because the use of formal techniques for group decision leads to serious paradoxes (such as the Arrow's theorem—see Sarkar (2012b)), the preferred method for resolution requires deliberation, a process that has many other virtues in the resolution of environmental disputes (Norton 1994). However, there remains another problem, very similar to the conflict between places, but not quite identical; it requires cooperation, rather than tradeoffs, between communities across large geographical scales.

18.4.4 *Conservation of Processes*

When conservation efforts are directed towards landscapes and seascapes, their focus is typically on individual places (conservation areas), that is, culturally embedded areas with significant biodiversity content, though (as noted earlier) care must be taken to accommodate interactions between such localities. However, protecting places in isolation is rarely enough to ensure persistence of biodiversity. Persistence requires the maintenance of biophysical processes and these occur at multiple scales, from local wind-borne pollination and seed dispersal to ocean currents.

Processes themselves that can become the goal of conservation efforts include long-distance animal migrations. The spectacular 10,000-km migration of loggerhead turtles (*Caretta caretta*) between Baja California (Mexico) and Japan is well known (Shanker 2015). The annual migration of Monarch butterflies (*Danaus plexippus*) in North America is perhaps even more impressive. It is the longest insect migration known to science and the problems faced for its conservation exemplify the difficulties of conserving processes.

There are two North American migratory populations, one with habitat largely restricted to the west of the Rockies, mainly in California and adjoining states, and the other migrating from central Mexico to the north of the United States and southern Canada east of the Rockies. There are also several non-migratory populations in Florida, the Caribbean, Latin America, and elsewhere. (This means that an end to the migration phenomenon does not constitute the extinction of the species.)

The western population mainly winters in California but some insects do move further south through Arizona to Sonora in Mexico. During the Spring most individuals move to the north and east of California. The eastern population, once over a billion individuals, overwinters in a dozen or so high altitude oyamel fir forests in the Transvolcanic Belt of central Mexico, covering the trees like carpets.

All these winter roosts occur within a 100×100 km square (Brower and Aridjis 2013). In the spring, after a frenzy of mating, the insects fly north to Texas. Most females lay their eggs on Texas milkweeds, typically attached to the underside of a leaf with only one egg per plant. Most of the wintering generation dies.

The eggs hatch into caterpillars that feed exclusively on milkweeds, pupate, and emerge as adult butterflies. (In contrast, adults feed on the nectar of flowers of a wide range of plant species; milkweeds are no longer particularly important.) The new generation hatched in Texas continues the northward journey. The population fans out, covering much of the United States north of Texas and east of the Rockies. Some butterflies probably change course to turn south to Florida to add numbers to a local non-migratory monarch population found in that state. Most continue going north over a third and, sometimes, even a fourth generation. The northern limit of the migration spans the upper Midwest of the United States onwards to Ontario and the southern edge of Canada. Over these three or four generations the butterflies may travel up to almost 5000 km.

The return journey is even more impressive. The last generation produced in the north travels back to the tiny overwintering area in Mexico. The insects sip nectar for fuel along the way, and flying only by day while typically roosting in small groups for the night. How the insects manage to find their oyamel islands is still poorly understood. Each insect must have both a “map” and a “compass” (Agrawal 2017). Here a “map” means that the insect must know where it is: how the monarch does this remains an unsolved problem. Direction is set by a “time-compensated sun compass” by which each insect uses its internal circadian clock to sense the time of day and the position of the sun to orient itself in the correct direction. When the fall migration starts, the preferred direction is south. The compass is reset during the winter; in spring, the preferred direction becomes north.

For the last few decades, biologists have been warning that this process is endangered. (The species itself is not at risk because of the existence of many non-migratory populations.) Because the overwintering population in Mexico is the entire source of the entire northward migratory population in the spring (and, therefore, of the migratory phenomenon itself), trends in its size are directly relevant to the question whether the migration will persist in the future. These overwintering populations numbered 400 million individuals in the early 1990s but only 100 million yearly since 2010, with a historical low of about 35 million in 2013–2014 (Sarkar 2017b). What has caused this decline remains a matter of controversy.

There is some consensus the degradation and disappearance of the wintering habitat in Mexico has contributed to the migratory population decline. For the wintering habitat, Mexican authorities began systematic conservation efforts in 2000, and these now appear promising in spite of past problems (V́ctor Śnchez-Cordero, personal communication). Beyond that, two conflicting hypotheses have been suggested though both could be operative. The *milkweed limitation hypothesis* predicts that spring monarch breeding populations before migration are in decline in the midwestern and northeastern United States and southern Canada. The alternative *migration survival hypothesis* proposes that the southward migratory population is suffering excessive mortality on its way south in Texas and northern Mexico (Sarkar 2017b).

If the milkweed limitation hypothesis is correct, conservation measures should be directed to milkweed restoration at the northern end of the migratory range, and many such efforts have been under way for more than a decade though, arguably, little to show in way of results. If the migration survival hypothesis is correct, efforts should be directed to providing food and shelter to the migrating population towards the southern end of the migration. If both are correct, both measures become important.

The salient point here is that maintaining the monarch migration will require collaboration across a continent-sized landscape. It is dissimilar to the case of conflicts between localities discussed earlier only because there is no potential for a solution through tradeoffs. Those who value monarch migration conservation as an important goal have a difficult task: what they are demanding is the value be attached to a process, not an entity, because the monarch as a species is not at risk of pending extinction.

18.5 A Synthetic Proposal

Where does all this leave us? Recall that, at the end of Sect. 18.3, it was noted that adequacy conditions can be adopted to constrain potential definitions of biodiversity based on local norms. It will be taken for granted that what is being targeted for protection is some aspect of nature (operationally distinguished from what are considered cultural features though, this distinction is not always trivial to maintain). The proposed constraints are intended to prevent all such natural targets of protection to be characterized as components of biodiversity, that is, what, elsewhere, I have called biodiversity constituents (Sarkar 2008, 2012b).⁸ These adequacy conditions are necessary to distinguish biodiversity as a value from cases such as: what is valued is some magnificent geological formation, the desire to preserve pristine wildernesses⁹, the protection of totemic species alone, the targeted protection of charismatic species such as large mammals in eastern and southern Africa, and so on. This is not to suggest that these are not important and culturally salient goals of conservation; biodiversity is not the only feature that deserves protection.¹⁰

More importantly, these adequacy conditions can be used to incorporate many, though not all, of the intuitions behind the many scientific attempts to define biodiversity mentioned in Sect. 18.2. This claim will be elaborated below as the four conditions proposed here are discussed in detail. Suffice it here to know that such a

⁸Earlier in the literature, these were called “true surrogates” for biodiversity—see Sarkar (2002), Margules and Sarkar (2002), and Margules and Sarkar (2007).

⁹There are serious ethical problems with wilderness preservation (Woods 2001) but what is most important here is that the

goal of wilderness preservation is not only distinct but also divergent from biodiversity conservation (Sarkar 1999).

¹⁰See Santana’s contribution to this volume to find a similar claim in different terminology.

strategy allows a partial synthesis between the scientific and normativist approaches, though only partial because only the intuitions behind the scientific definitions rather than their specifics get incorporated into this strategy.

What requirements should we impose on potential biodiversity constituents? Here, four adequacy conditions will be proposed¹¹:

1. *Constituent entities be biotic*: We are proposing a definition of *biodiversity*. This conditions dates back to Sarkar and Margules (2002). It allows biodiversity constituents to be habitat types, taxa, communities, genes, traits, and so on; but it excludes, for instance, physical environmental features such as rock formations or sand dunes. It also excludes human cultural diversity whether or not cultural diversity contributes to the presence or persistence of biodiversity in a given context.

Nonbiotic features may well be good surrogates for the constituents in conservation planning. For instance, Sarkar et al. (2005) showed that sets of abiotic environmental classes are often adequate surrogates for varied classes of biota (the putative biodiversity constituents), while many authors have argued that sets of taxa are very rarely good surrogates for each other (Margules and Sarkar 2007) even though they continue to be used (Caro 2010). The success of environmental surrogates does not provide any argument that such abiotic features should be considered as components of biodiversity; rather, it shows that they are good *surrogates for biodiversity*.

2. *Emphasis must be on variability of the constituent set*: That is why it is *biodiversity*. The motivation for this criterion is best explained using an example. Neotropical rain forests have played an iconic role in conservationist campaigns since the mid-1980s, their public appeal perhaps best exemplified by Caufield’s (1984) haunting account of their disappearance around the world. Yet, neotropical dry forests are far rarer and more threatened (due to ongoing land cover conversion) than rain forests. When neotropical rain forests, which are arguably over-protected in some regions such as Ecuador, are taken to be emblematic of biodiversity at the expense of neotropical dry forests to the extent of being the basis for a characterization of biodiversity, this condition is not met.

For habitat types this means that attention should not be restricted to some subset and exclude all others entirely when biodiversity is defined. When dealing with taxonomic groups, this condition also suggests that differences at higher taxonomic levels than that of species are more salient than inter-specific differences. To put it another way, a species that is the sole member of a phylum (e.g., the aquatic species, *Trichoplax adhaerens*, the sole member of Placozoa¹²)

¹¹ In my own work, these adequacy conditions have evolved over the years due to continued discussion in many forums—see, for example, Sarkar (2008, 2012b). Condition 4 is being proposed here for the first time.

¹² Note that there is some controversy over this uniqueness claim because some taxonomists feel that there is sufficient genetic diversity within this putative species to distinguish it into several morphogenetically very similar species (Voigt et al. 2004).

is more important for conservation than a species which belongs to a genus with thousands of species (e.g., any jewel beetle species of the genus *Agrilus*).

3. *Embrace taxonomic spread*: It is particularly important that the definition does not by fiat place arbitrary limitations on the taxa permitted to fall under the scope of “biodiversity.” This requirement is probably not controversial. Part of the rhetoric of early conservation biology was that there was a need to move beyond charismatic species that had been the traditional foci of conservation campaigns and embrace the full spectrum of life as worthy of preservation. This rhetoric was often matched by the more concrete proposals that emerged from the field. Its sincerity is being accepted here.

An important function of “biodiversity” was to codify this broadening of conservationist intent. It is arguable that not imposing some requirement that is functionally equivalent to the one being proposed here would miss the entire point of why the new term was enthusiastically adopted in the historical context of its introduction.

4. *Biodiversity constituents must be precise enough for their presence and abundance to be measured*: Within conservation biology in the 1990s, one of the motivations for defining biodiversity was to enable its measurement and quantification. For instance, Williams and Humphreys (1994) begin their discussion with two problems that have to be solved: (1) a relatively theoretical one—what is to be measured? and (2) a practical one—can the data “realistically” be collected? So, it seems reasonable to impose a measurability adequacy condition.

Margules and Sarkar (2002) modified Williams and Humphreys’ distinction to distinguish between a quantification problem and an estimation problem which together form what they called a biodiversity assessment problem. Solving the former requires the ability to measure biodiversity constituents *in principle*. That is what this condition requires. Solving the latter problem requires the operationalization of biodiversity for various purposes. For instance, in conservation planning, the detailed spatial distributions of thousands of biodiversity units are required as data. For many biodiversity constituents, obtaining such data, even though in principle possible, is not *in practice* reasonable given time and other resource constraints. What must then be found are adequate surrogates (such as the environmental classes discussed earlier) but these are not part of the definition of biodiversity.

It is instructive to analyze which sets of features survive this adequacy test and which ones do not. One standard approach, that biodiversity is all diversity at the level of genes, species, and ecosystems does not—it calls afoul of Condition 4. Sets of all at-risk species survive community; as do sets of habitat types (so long as they are defined, at least in part, using the ecological communities in them) though it is arguable that the first of these satisfies Conditions 2 and 3 only accidentally rather than as a matter of emphasis. (There is no deep reason why at-risk species—or other taxa—should be varied in their content or span much of the taxonomic hierarchy.)

These cases will probably come as no surprise to conservation biologists who embrace a scientific attitude to biodiversity. In fact, they show how these adequacy criteria help bridge the gap between normativism and scientism. However, the adequacy conditions also admit non-standard collectives as potential constituent sets for biodiversity, for instance, the sacred groves of Meghalaya (India) discussed earlier. Condition 1 is obviously satisfied. Condition 2 is satisfied because different kinds of forests present in the region can constitute sacred groves and, internally, they exhibit the variability of tropical cloud forests. Condition 3 is satisfied because each sacred grove is viewed as consisting of all biotic features within them. Condition 4 is satisfied because the number and type of sacred groves in any given collection is relatively easily assayed. If the earlier cases show that the adequacy conditions enable the relevance of scientific intuitions, this one shows how local normativism does not lose out. These conditions permit wide cultural divergences about what type of natural variety merits protection.

18.6 Concluding Remarks

The discussion of biodiversity in this paper has presumed the categoricity of its use in conservation biology and, more generally, biodiversity conservation. However, other areas of biology, in particular taxonomy, have also laid claim to the term over the years. How would the definitional strategy proposed here fare in these areas? Not very well, at least in the case of taxonomy. Taxonomy, by its own explicit goals, is fundamentally a descriptive enterprise; though its theoretical structure does embrace some normative issues, these are epistemological rather than axiological as seen, for instance, in the debates over cladistics (Platnick 1978). Normativism, as outlined here, is simply irrelevant to taxonomy though most taxonomists no doubt embrace many of the normative goals of biodiversity conservation.

How should we address the potential dissonance between the strategy for defining biodiversity presented here and the concerns of taxonomy? The answer given here will be cynical and based on sociological speculation that must be tested against data before the answer is deemed plausible. The speculations: Classical taxonomy had been underfunded since the dominance of molecular biology over the life sciences was established in the 1960s. By the time that conservation biology and “biodiversity” came along in the late 1980s, classical taxonomy based on macroscopic organismic rather than molecular traits, was a dying discipline. Taxonomists jumped on the biodiversity bandwagon when it became apparent that conservation was becoming a powerful current within and beyond the environmental movement. There was money for biodiversity inventory and conservation and, by endorsing that locution, taxonomists could lay claim to some of those resources.

To continue with the cynicism: Conservation biology was supposed to be a “crisis discipline” (Soulé 1985) because species were becoming extinct before biologists could even describe, let alone study, them. With respect to description, the problem was presented as a shortage of trained taxonomists available for

that task.¹³ The solution? More money for taxonomy. In Costa Rica, there were even moves to generate an army of sparsely-trained “parataxonomists,” akin to China’s barefoot doctors of the Cultural Revolution, with the task of inventory, producing lists of species at individual locations.

Taxonomy obviously does not place any taxonomic limit on what should be described: the more obscure or difficult a group of organisms, the more technical acuity could be deployed in their classification. From this perspective, the operative measure of biodiversity is species richness (or, possibly, richness at some higher taxonomic level). Success in taxonomy is determined in part by the sheer number of taxa that are successfully described. It is perhaps because they take the claims of taxonomists to be as pertinent as those of conservationists that philosophers such as Maclaurin and Sterelny (2008) embrace richness in their account of biodiversity. A major advantage of this approach is simplicity: richness is conceptually easy to grasp and relatively easy to measure in the field. But the earlier discussions in this paper should also underline the problems.

Why reject the salience of taxonomy? It is time to move beyond cynicism and speculation. The point is that the concept of richness was available to taxonomists long before the advent of “biodiversity.” Not only did taxonomy not need the new concept, the neologism made no difference to the practice of taxonomy as a discipline. For taxonomists, “biodiversity” was a slogan, a source of resources for their field.

In contrast, conservation biologists required an operationalized concept of biodiversity to assess the extent to which any measure succeeds or fails because the conservation of biodiversity was the explicit goal of the field (Sarkar and Margules 2002). This is the *argument from necessity*. If we also accept that concepts are best understood in the context of their introduction and use, biodiversity must be understood in the context of conservation biology. But even if we do not endorse this argument from genesis, the argument from necessity makes conservation central to the meaning of biodiversity which, given this context, in turn requires a focus on norms and values for its definition.

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¹³ Even as late as 2000, Wilson (2000) was making this claim.

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