

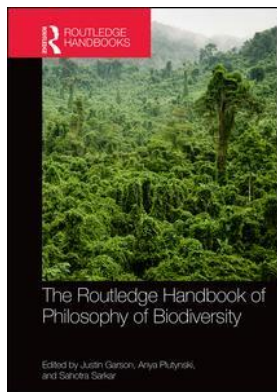
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ESTIMATING BIODIVERSITY LOSS

Yrjö Haila

Introduction

What is known today as the biodiversity crisis is a relative newcomer among environmental concerns. The concept originated in the 1980s, and has since then succeeded in claiming a prominent place on the agenda of global environmental problems. How and why the concept achieved a successful reception is told and retold in a vast literature; I think Lawler *et al.* (2002: 295) offer a good concise explanation by writing: “Biodiversity loss has become a rallying point for conservation activists because it provides a scorecard for how greatly humans are making an impact on nature.”

“Loss” is a keyword in this statement. Moreover, loss interpreted as a “scorecard” implies quantitative assessment, and this requires criteria that can be quantified at least on an ordinal scale. The need for criteria and quantification is supported by ambitious international goals concerning the discontinuation of loss of biodiversity, as these have been formulated since the enactment of the Convention on Biological Diversity (CBD). A precondition of reaching such goals is that somebody be able to assess whether loss of biodiversity still goes on or has, actually, been curbed.

My aim in this essay is to chart the approaches adopted to estimate the magnitude of biodiversity loss. Coming up with reliable criteria has turned out to be a hard nut to crack. The all-encompassing character of the concept of “biodiversity” is a source of confusion. Reliable indicators that would join together generality (global significance) and specificity (what is observed locally) are difficult to come by. An even more confusing issue is that the assessment has to be fixed within a temporal horizon from a significant past over the present to a foreseeable future. Mere recording of change in nature is inconsequential; criteria are needed on what kind of change represents loss.

For estimating biodiversity loss, the first necessary step is to specify under what conditions and for which particular purposes this is possible at all. I adopt a pragmatic perspective: it is the need to formulate effective policies and rules of management that require that the characteristics and driving forces of biodiversity loss ought to be understood as precisely as possible. In the next section I explore conceptual difficulties in clarifying the goal by specifying ambiguities inherent in efforts to operationalize biodiversity. In the following sections I review approaches adopted thus far. Finally, I get back to specifying the preconditions of reasonable assessment.

Ambiguities in “biodiversity”

The understanding of the meaning and significance of biodiversity has changed during the short time it has been in focus. To begin with, the way biodiversity loss was originally framed as a crisis deserves attention. The talks given at the foundational event, the National Forum on BioDiversity (Washington, DC, 1986), were published in an edited volume two years later (Wilson and Peters 1988). In his “Editor’s foreword” to the volume, Edward O. Wilson named two “more or less independent developments” that supported the crisis perception: “The first was the accumulation of enough data on deforestation, species extinction, and tropical biology to bring global problems into sharper focus . . . The second development was the growing awareness of the close linkage between the conservation of biodiversity and economic development.”

There is an internal tension between Wilson’s two points. His first point refers directly to “accumulation of data” and is, in principle, amenable to an assessment by observations. A precondition is, of course, an agreement on what kind of observations are relevant. The second point, by contrast, is primarily about a novel understanding of the value of biodiversity for human sustenance. Wilson’s reference to “economic development” is loose, to say the least. It is far from obvious how a link between biodiversity and economic development could be made specific enough to be quantifiable.

The duality inherent in Wilson’s characterization shows that analytic and normative appraisal of the problem cannot be torn apart. Specifically, the analytic question “What is lost?” and the normative question “What does the loss mean?” have to be evaluated together. This unavoidable tension implies that the critical question for estimating biodiversity loss is what to estimate, and for what specific purpose. In other words, assessment has to be problem-driven. A critical step is problem definition such that we can define tasks that can be meaningfully taken.

A useful step toward realistic problem definitions is to acknowledge a set of ambiguities that follow from the dual nature of the issue. Such ambiguities concerning problem framing should be understood as concretely as we ever dare, in analogy with famous perceptual ambiguities such as the duck–rabbit duality. When staring at the duck–rabbit image, we can detect either a duck or a rabbit, but not both at the same time. By taking turns in looking at one or the other of the images, we can gain understanding on how the duality works. Similarly, conceptual ambiguities can create clarity of the conceptual field in question if we approach the alternatives in turns.

I take up three ambiguities, partially leaning on Maclaurin and Sterelny (2008). One of the ambiguities they identify (p. 2) is the tension between biodiversity regarded as the object of protection in itself, and biodiversity as providing instrumental benefits for humans. This formulation reproduces E. O. Wilson’s two factors in a different phrasing.

Another ambiguity Maclaurin and Sterelny take up, the “units-and-differences problem” (p. 21), is directly relevant for assessment. To succeed in assessment, we have to identify as book-keeping elements such units of biodiversity that may be lost, but in addition, we have to assess the significance of differences recorded between the units. Taxonomic units such as species provide good examples. Are two taxonomically closely related species “equally different” from each other as two taxonomically distant species? If not – as many authors have argued with good grounds – the significance of the difference has to be included in the assessment.

But there is also a third type of ambiguity, not as commonly recognized: what, precisely, is the “past” that offers a standard of comparison with present and future? A prevalent tendency is to use the “natural state” of nature as a standard. This, however, gives rise to a further question: how to reckon with human environmental modification? Has past human influence always and unexceptionally been bad for biodiversity? Change and loss have to

be distinguished from one another. Throughout history, humans have learned to modify biodiversity to better answer to their needs – for instance, by clearing fields and establishing gardens. To assume that “untouched nature” is a gold standard for biodiversity reflects an ideological bias stemming from a dualistic view of humanity versus nature (see Haila 1999a, 1999b).

I refer to these three ambiguities below to bring into focus points of confusion in defining criteria for assessing biodiversity loss. Clearly articulated ambiguities can play a positive role both in scientific thinking and in political discussion and thought (Majone 1989, Rein 2006), but if not taken into account, ambiguities give rise to muddled thinking. Ambiguities can also be used in a purely rhetorical fashion to play down arguments of assumed opponents. In this specific sense the brief history of the biodiversity concern has given rise to interpretative problems. The concern was originally painted as a crisis using the strongest and darkest expressions the proponents were able to come up with. The crisis framing elevated the problem to an enormous social and political scale (Haila 2004). I believe this has been an important and misleading part of the legacy of the concern.

Any assessment of biodiversity loss must be in a close relationship with a clearly articulated perspective on what biodiversity is and what it means. Hence, in what follows I have adopted a dual strategy to structure my argument. I use as my material on the one hand results of investigative practice – the real stuff of scientific work (Dyke 1988) – and on the other hand shifting views on what is the most fertile perspective of framing the biodiversity issue as a problem. Contrast space is a useful conceptual device for getting hold of the framing of arguments, especially in situations involving ambiguity (Garfinkel 1981, Dyke 1988). A contrast space articulates the basic alternatives between which an explanation ought to provide a resolution. Then, if the context is ambiguous, different alternatives can be clarified by constructing several contrast spaces and comparing the conclusions they give rise to. For instance, the units-and-differences ambiguity mentioned above can be clarified by constructing alternative contrast spaces: recording differences in species composition versus assessing the significance of the differences on taxonomic grounds require different contrast spaces.

Quantifying the rate of extinctions

Nature conservation has its roots in broad cultural consciousness concerning proper human behavior toward non-human nature, but extinction was already a major concern at the end of the nineteenth century (Adams 2004). The early focus was on large emblematic animals decimated by hunting, but in the course of the twentieth-century conservation goals became more comprehensive. New investigative practices were developed in particular to survey the geographic range and abundance variation of both single species and specific ecological communities. These efforts gave rise to a rich tradition of practical sampling methods, accompanied with procedures developed for correcting methodological biases to obtain “true” values from samples collected in the field (Haila 1992, Kohler 2002). Articles in Magurran and McGill (2011) provide a rich overview of sampling methods focused on biodiversity. By and large, it seems that the investigative practice of field surveys fed upon itself – more confidence was felt about figures of species richness than is warranted on realistic ecological grounds. Gaston (1996) and Gotelli and Colwell (2011) present overviews of both methodological and conceptual problems of measuring species richness.

Biodiversity brought about a new dimension into conservation concerns: the threat of an imminent biotic impoverishment, driven by an extinction wave. Statistics played a prominent role in efforts to assess the seriousness of the threat: first, as statistics on extinctions, and later

on, as statistics on a range of indirect indicators such as the area proportion of protected areas of various political and biogeographic units.

It is very difficult to construct reliable extinction statistics for several interconnected reasons. First of all, direct observations do not offer a place to start. Unambiguous documentation of extinction is well-nigh impossible except for large and visible creatures such as the famous cases from earlier times such as the dodo, the great auk, the passenger pigeon, and so on. A list of such species does not add up to a reasonable estimate of extinction rate. More comprehensive population-level conclusions can only be drawn when a specific valuable and unique habitat has been destroyed; Gentry (1986) gives examples from the Amazon. However, in such cases representativeness remains a problem – that is, it is uncertain whether such observation can be extrapolated to other areas and larger scales.

Extinction statistics are faced with the problem of specificity versus generality: all well-known cases of extinction are unique (just take a look at a classic such as Halliday 1978), but meaningful statistics require generality. Habitat area was adopted as a surrogate for extinction probability in the 1970s. The inspiration came from the species–area relationship, brought into the domain of nature conservation by a short note in the *Theory of Island Biogeography* by MacArthur and Wilson (1967), with a reference to woodlots in Cadiz Township, Wisconsin, leaning on an original data set published by plant ecologist John T. Curtis (1956). Frank Preston (1962) had earlier made a similar argument more extensively. For a while, habitat destruction assessed by the reduction in habitat area was adopted as a primary surrogate for estimates of species extinctions. Wilson (1988) was really explicit on this: “The area–species curves of island systems, that is, quantitative relationship between the area of islands and the number of species that can persist on the islands provide minimal estimates of the reduction of species diversity that will eventually occur in the rain forests.”

However, the use of area as a surrogate is problematic on several counts, as has been pointed out almost innumerable times. The analogy with island biogeography assumes that the remaining patches of habitat are surrounded by “biological desert” after the reduction in total area, but this is never the case. Furthermore, if the surrounding areas comprise forests, secondary growth starts on the surrounding areas, and sooner or later some of the forest species find suitable habitats there. The use of area as a surrogate builds upon a flawed contrast space. Ariel Lugo has defended this qualification from the very beginning of the debate, backed by his empirical experience from Puerto Rico (Lugo 1988). I find it very strange that Lugo’s work has not found a foothold in the estimates of extinction rate.

Furthermore, statistics based on habitat area imply a highly problematic step from the present to the future. The assumption is that the quality of the habitats both in the preserves and in the surroundings remain the same, but this is never the case. This problem is inherent in the notion of “extinction debt,” that is, that populations inhabiting an assumedly too small habitat space are “doomed” to extinction, being merely “living dead” at present (Tilman *et al.* 1994). “Extinction debt” elevates an assumed future to a criterion of the present. While the reality of such dynamics is, indeed, quite plausible, the lack of generality hits back. The lower boundary of area that triggers extinction debt is difficult to pin down, and it certainly varies across taxa, and probably also across biogeographical regions.

Overall, the situation presents an inherent paradox. There is no doubt whatever that human activities have changed and continuously change environments negatively, compared with the previous state. However, precisely documenting and quantifying this change is far from easy. In my view, it is necessary to use some background variables as indicators (“proxies”) for estimating change on an ensemble level, no matter that proxies necessarily bring additional interpretative ambiguity into the picture (Haila and Henle 2014).

Lindenmayer and Likens (2011) disagree. They promote, instead, what they call “direct measurement” of the entities of concern. However, the requirements they demand of the “right” entities are very stringent. If the emphasis is on single species or some very well defined habitat types, this approach would certainly work out, but the problem of generality remains. When larger suites of species or environments are of concern, surrogates are necessary, whether we like it or not.

In search of reliable indicators

In any case, the indicator chosen has to be valid, that is, the variation of indicator values must correlate reliably with the variation of what it is supposed to indicate. The theory of island biogeography got a critical role by proposing that area be an adequate surrogate. This interest had a cycle of vigorous life as a concern over habitat fragmentation, that is, the perception that human influence reduces previously continuous habitat areas to sets of “island-fragments” isolated from each other. A merely declarative reference to “habitat fragmentation” leans on the misleading contrast space of “intact habitat” versus “ecological desert on the outside” (see Haila 2002, Lindenmayer and Fischer 2006).

Lists of endangered species offered a remedy to overtly general statistics covering all species and types of environments. The lists were also adopted to indicate extinction risk more generally. Fulfilling this aim became plausible with increasing capacity of computers that could be utilized to develop algorithms for identifying reserve networks such that some specific criteria be fulfilled – for instance, the inclusion of rare and endangered species in the network. Australian ecologists were active in developing this line of inquiry (see Margules and Austin 1991). The total area of preserves and other protected areas has been another indicator. These measures are very coarse, however; as has been emphasized many times, the success or failure of biodiversity preservation takes place outside preserves.

At the opposite end of the range of scales are efforts to compile composite indices that give an overall view of the status of biodiversity in larger and internally varied geographical regions. This approach has good potential when used with care. One example of such an approach is the “biodiversity intactness index” suggested by Scholes and Biggs (2005) for their target region of southern Africa. The index that they propose is compiled in such a way that it would meet the criteria defined by the CBD. The index puts together estimates of the population sizes of vertebrates and major groups of plants, as well as main habitat types. The index requires a reference; Scholes and Biggs (2005) suggest “that which occurred in the landscape before alteration by modern industrial society.” We cannot go into the detail of the algorithm they use, but the result is interesting: “Overall, we estimate that $84 \pm 7\%$ of the pre-colonial number of wild organisms persist in present-day southern Africa.” As they note, 99 percent of the species persist, which indicates that indices based on extinctions are less sensitive.

Another promising example, methodologically a close relative of the intactness index of Scholes and Biggs (2005), is the so-called Norwegian Nature Index. The idea and its practical implementation are introduced in a set of articles published in *Norsk Geografisk Tidsskrift – Norwegian Journal of Geography* 66(5) (2012); for an overview, see Nybø *et al.* (2012). The Norwegian index was compiled by making detailed interviews with a large number of specialists in different fields of biology and management of natural resources. The process was initiated in 2005, inspired by the goal set within the confines of the CBD to halt the loss of biodiversity by 2010 (later, the focal year was extended to 2020). Similar to the southern African experience, the reference state was chosen in a very pragmatic manner, as the natural state of the environmental types, with baseline year at 1950. The point was merely to enable a convergent

perception among the specialists who participated in the exercise, without any deeper meta-physical commitments as to what “primeval” nature of Norway might have looked like. In the case of Norway, such pragmatism makes perfect sense as the whole country was covered by continental ice as recently as 20,000 years ago, save, perhaps, a few glacial refugia in the mountains or along the Atlantic coast.

The southern African and Norwegian experiences are encouraging by suggesting that data aggregated from a large number of heterogeneous sources is usable in assessing the status of biodiversity. Clearly, a basic requirement is that adequate data are available. Norway is certainly a special case, as a country with a strong tradition of field naturalism and relatively small total area. On the other hand, the environmental types found in Norway are very diverse, and hence, the success in compiling the Norwegian Nature Index is very promising indeed.

On an intermediate scale between habitat-specific area calculations and composite indices are comparisons based on the distinction between alpha, beta, and gamma diversities. These notions derive from the interest in patterns of variation in species richness across heterogeneous landscapes that took off in the 1970s. A distinction was drawn between within-site (“alpha”), between-site (“beta”) and regional (“gamma”) diversity. When ecological realism is given due care, meaningful comparisons are possible; several articles in Magurran and McGill (2011) describe the background and assess the present status of this line of research.

Considerable controversy has been created by the question: if (when) species disappear from local and regional species assemblages, is the effect observable as a decline in local (alpha) diversity or as a change in species identities (“turnover”)? Dornelas *et al.* (2014a) conducted an extensive meta-analysis on the level of local communities and concluded “that local assemblages are undergoing biodiversity change but not systematic biodiversity loss” (p. 299). This result is interesting, but it brings up the ambiguity concerning the equality of species: are the species that make up local communities in modified environments “equally valuable” as in the original conditions? An exchange by Cardinale (2014) and Dornelas *et al.* (2014b) focuses on this issue.

Beta diversity, or variation in biodiversity across localities on a larger spatial scale, is another facet of the intermediate scale. Gaston *et al.* (2007) present a thoughtful review of the relevant issues which are firmly embedded in pre-biodiversity investigative practice. Gaston *et al.* (2007) conclude that there is not much by way of general theory, let alone predictive theory in this regard. Hence, it seems variation in beta diversity does not qualify as an indicator of biodiversity loss.

Species turnover in time does not fare much better, although it is difficult to decide as the database is much scarcer than in the case of spatial turnover. The reason is simple: exploring temporal variation requires that the same targets be monitored for extensive periods of time, but field ecologists have only occasionally got that chance. A few remarkable exceptions are well known, though, most prominent among them the long time series of both plants and insects collected at the Rothamstead Agricultural Experimental Station in England. The insect data gave invaluable material for exploring variation in species abundances in the early and mid-twentieth century (Fisher *et al.* 1943, Williams 1964). The contribution of the Rothamstead experiment to ecology was reviewed by Silvertown *et al.* (2006) who point out several insights the experiment allows on factors influencing local variation in plant diversity across the experimental plots through time.

Another important aspect is that in every local assemblage studied over a number of years, species turnover has been observed between the years. This is a natural part of the dynamics of local assemblages – comparable to population “kinetics.” Breeding birds, for instance, shift locations of their nests from year to year even when the population level remains

stable (Haila *et al.* 1996). The pattern was characterized as “equality of space and time” by Frank Preston (1960).

The instrumental dimension: the functional significance of biodiversity

Quite early on some conservation biologists began to feel that the value of biodiversity in itself does not provide sufficiently convincing grounds for preservation. What is additionally needed according to this perspective is a view on what biodiversity does such that the basis for human sustenance and welfare is strengthened. Edward Wilson made this point in the 1986 Forum by referring to economic benefits, as we saw above.

An emphasis on the role of biodiversity for various ecosystem functions took off in the early 1990s; a collection of essays edited by Schulze and Mooney (1993) is often mentioned as the first effort to give the functional role of biodiversity a systematic presentation. In the foreword to the volume, Paul Ehrlich (1993) makes the following statement: “Of special interest to humanity is the relationship of biodiversity to the variety of services provided by ecosystems and, in particular, to the stability of the flow of those services, such as the maintenance of the gaseous composition of the atmosphere, preservation of soils, recycling of nutrients, and provision of food from the sea.” In effect, Ehrlich in this quote gave a precise formulation to Wilson’s loose talk of economic benefits: the benefits are ecosystem services, produced by normal functioning of healthy ecosystems.

Exploration of the functional significance of biodiversity has proceeded along two complementary lines which, in a sense, reproduce the division of arguments into analytic and normative questions: is the functional role real?; and which functions are important?, respectively. On the analytic branch, experimental studies were started on the correlation between species number and specified functional aspects of ecosystems. Some of this work predated the biodiversity concern and was primarily inspired by previous interest in the relationship between diversity and stability in local communities. Several essays in Schulze and Mooney (1993) summarize the results. Grassland plants have dominated this line of research, for understandable logistical reasons. On the normative side, specifying what the critical functional features of ecosystems are, and how their role can be demonstrated, has turned out to pose important conceptual and practical problems. Such problems highlight the ambiguity of biodiversity *per se* versus instrumental value of biodiversity: a precondition for presenting specific instrumental benefits of biodiversity as support for preservation is that those benefits can be named and demonstrated.

Results of the experimental studies have been summarized several times. While the general positive correlation between biodiversity and ecosystem functions is uncontroversial particularly when the numbers of species in the experiments are relatively small, the generalizability of the results remains a tricky problem. Ecosystem ecology and community ecology have partially different pedigrees, and one of the problems in finding a common ground has been the difficulty of mapping structural elements (individuals and populations of different species in communities) to functional elements of ecosystems in which they reside. In the ecosystem tradition, the notion of functional type has been in use for some time (e.g. O’Neill *et al.* 1986), but the ambiguity concerning the “units-and-differences problem” hits back at this point: which types are similar, which are different? A weakness in much of the work on ecosystem functions is that the critical role of microbes is often not recognized at all, but we lack the space to get any deeper into this problem; Meyer (1993) offers an early overview; Øvreås and Curtis (2011) discuss the question albeit mainly from an inventory perspective.

In other words, specifying what the functional types actually are in specific ecosystems is tricky (see also the thoughtful review by Weiher 2011). Conceptual problems concerning functional types feed back to empirical research. It is difficult to get investigative practices on ecosystem function to stabilize. Cardinale *et al.* (2011) draw this conclusion in their comprehensive review. More or less convincing experiments on the statistical relationship between species number and one or another functional feature measurable in a particular ecological community are piling up, but the mechanistic bases of what is observed are mainly unknown. Even if functional groups could be identified with some degree of confidence, assessing their relative quantitative importance poses additional problems, as pointed out by Hooper *et al.* (2005).

Some empirically grounded distinctions have been tried, which are in themselves interesting. Duffy *et al.* (2007) draw a distinction between diversity within trophic levels (horizontal diversity) and across trophic levels (vertical diversity). Another interesting distinction studied by Srivastava *et al.* (2009) is between the effect of bottom-up versus top-down diversity on ecosystem function. They made a meta-analysis of studies on decomposition efficiency and concluded that in such “brown” food webs the diversity of detritivorous organisms has a clear effect on decomposition whereas resource diversity (types of detritus) seems not to have any effect on consumption. As the authors note, their conclusion conflicts with the established view that in “green” foodwebs resource diversity has a major influence on consumption efficiency (see Cardinale *et al.* 2011).

A methodologically relevant side effect of drawing refined distinctions is that the dimensionality of the problem increases. As the relationships among the dimensions are most certainly non-linear, the prospect of drawing firm generalizations fades away. For instance, Cardinale *et al.* (2011) conclude that the positive relationship between producer diversity and ecosystem functioning is well established, but as regards assumptions about possible mechanisms, the literature “is a ‘reader beware’ field. Less than half of all claims made in the abstracts or discussions of papers are backed by any direct statistical test” (p. 580). As a final conclusion, they present a set of tasks for the future, calling for ambitious long-term experiments focusing on multifunctional and non-linear effects as well as “studies to embrace and try to explain natural variation rather than experimentally control it” (p. 589). In effect, they emphasize problems of interpretation and extrapolation inherent in the relationship between the constructed specificity of controlled experiments and the broad spectrum of natural conditions.

Ecosystem services is a 1990s addition to the discussion of ecosystem functions that are beneficial to humans. I cited Ehrlich above, but in his programmatic statement the services are identified on a very general level: “the maintenance of the gaseous composition of the atmosphere, preservation of soils, recycling of nutrients, and provision of food from the sea.” Ehrlich’s statement rings true: the global characteristics of the current Earth are, indeed, produced and maintained by life. A critical problem is, however, that the claims are too general to offer sites for drawing distinctions. Everything alive supports the services on the level Ehrlich refers to them – the statement has no diagnostic power.

Another problem that has created quite a lot of controversy is the question of functional redundancy, that is, whether the roles of different species in some ecosystems can be similar enough that they can compensate for the loss of one another. Lawton and Brown (1993) present an early overview of the problem of redundancy. By and large, they support the view that functional redundancy is a real phenomenon. This, of course, makes assessment of biodiversity loss murky: how do we know that extinctions necessarily reduce the functional strength of ecosystems if some species are redundant?

Against assumptions of functional redundancy some authors have argued that high diversity is always needed to maintain ecosystem services (e.g. Isbell *et al.* 2011), claiming support for

the precautionary principle that “all species should be conserved because we cannot be certain which species actually produce ecosystem services.” As a statement about high principles this sounds convincing, but when looking at the existing ecology of the Earth we come across the ambiguity about the nature of human influence: what is the significance of species inhabiting environments intensively modified by human actions? What about novel ecosystems on sites heavily disturbed by humans? Isbell *et al.* (2015) give additional emphasis to the principle by invoking what they call “ecosystem service debt.” From the point of assessment this idea shares with extinction debt the problem that an assumed but basically unknown future is elevated to a criterion to assess the present.

Quite expectedly, the broad use of ecosystem services as an argument for biodiversity protection has also raised criticism. For the perspective of estimating loss, a major criticism draws upon the ambiguity of “units-and-differences.” Ridder (2008), for instance, points out that resilient ecosystem services are unlikely to depend on rare species, which are often the focus of conservationists. A distinction between resilient and sensitive ecosystem services emphasizes the context specificity of instrumental arguments.

A policy perspective: assess causes instead of symptoms

A strong motivation to present precise estimates of biodiversity loss has been to support views on the urgency of the problem. But as documentation of the amount and significance of biodiversity loss has turned out to be unexpectedly difficult, a promising alternative, in my view, would be to put more emphasis on such human practices that are known to be harmful. I base this view on a simple argument: wanton destruction of ecological resources should be tackled irrespective of whether its global significance in terms of loss of biodiversity can be quantified. In a sense, this would be an updated version of the age-old concern that biological resources should be used in a sustainable fashion.

Oceanic fisheries provide good examples: there is no doubt that all over the world's oceans, criminally defective and exploitative fishing methods cause destruction of biodiversity. Examples of collapses of fish stocks have accumulated since the cases of the Peruvian anchovy, the Californian sardine and the cod off the northeastern Canadian coast. Worries concerning industrial-scale overfishing in basically all parts of the world's oceans are certainly founded.

The fisheries example raises some problems that have to be acknowledged. The past hardly offers standards, as reliable quantitative data on fish stocks are quite recent (e.g. Caddy and Gulland 1983). Instead, the attention has to be focused on the perspective from the present to the future – in this, of course, qualitative knowledge of old fishing possibilities can be used for advice. I think this is feasible when the focus is restricted enough – on the fishing grounds off the coast of Newfoundland, say. Also, the sustenance of local human communities, dependent on exploiting local fish stocks, has to be woven in. The challenge is to link social and ecological systems in a harmonious way. As a precondition, strong vested interests should be held at an arm's length from management operations. For instance, the fisheries policy of the European Union is too strongly influenced by strong fisheries interests.

Overall, the idea that different types of disturbance are important in ecology, broadly accepted since at least the 1950s, offers one starting point. Dornelas *et al.* (2011) present a thorough overview of different forms of disturbance and also point out that forms of disturbance due to natural and to human-induced causes grade into one another. In general terms, getting human-induced effects on ecological processes to resemble as closely as possible forms of natural disturbance is a reasonable piece of advice, easily stated, but more complicated to follow (Haila

and Levins 1992). The “resilience” school of ecological research, building upon the work of C. S. Holling and his collaborators (Gunderson and Holling 2001), has elaborated this approach. The idea of resilience is difficult to quantify on large scales, let alone globally, but it has analytic strength in local, specified contexts for efforts to prevent the destruction of biological resources.

A general problem is that the causes of the destruction of biodiversity may be named using conceptual frames that are way too general to be analytically useful. The frame adopted by the European Environment Agency dubbed “Driving forces – Pressures – States – Impacts – Responses” (DPSIR) is an example. A critical problem of the scheme is that it is presented as a causal chain but each one of the elements is too general to be of any specific analytic use. Maxim *et al.* (2009) argue that the scheme is analytically useless but can, nevertheless, be used for communication between scientists and users of environmental information. That may be a valid argument (which I tend to doubt), but only under the condition of explicitly defining the context.

Pragmatics: problem framings, investigative practices, and critical timeframes

Biodiversity is a rich and multifaceted perspective on the conditions of life on Earth, with deep roots in ecological and evolutionary understanding. Maclaurin and Sterelny (2008) offer an excellent guide to this richness. It is due to this richness that straightforward quantification of the status of biodiversity has been and is difficult. My view is that this is in the nature of the idea itself: there is no single and simple way to gain understanding or to plan policy or management.

Butchart *et al.* (2010) looked at thirty-one general indicators on whether the target of halting biodiversity loss by the year 2010 had been reached by the time of their analysis (2009, apparently). They concluded that some local successes had been achieved, but nevertheless, the general decline of biodiversity had not been halted. The result is not unexpected, taking into account the generality of the criteria that were defined for assessing the target. The indicators are so grossly aggregated and all correlate strongly with overall trends in social development and change that any other conclusion is hardly possible.

I suggest the preconditions of adopting a pragmatic perspective include, at a minimum, three elements: (1) framing research problems such that they illuminate the real-life problems at hand; (2) making sure that we have, or are able to develop, investigative practices which allow solving the problems; (3) considering the temporal perspective within which successful addressing of the problem is possible at all, also taking into account how successfully solving one, perhaps apparently minor problem paves the way for successfully addressing other, deeper problems. Such a “chaining” or “coupling together” of problems is a characteristic feature of any successful policy in the incomplete world of human actors (Majone 1989).

Honest assessment of the situation is a major requirement on every step. In practice, moving along the chain proceeds iteratively, not linearly. A source-book of methods such as the volume edited by Magurran and McGill (2011) offers invaluable help. Note, too, that the assessment always includes both analytic and normative dimensions.

Where does this leave us as regards estimating biodiversity loss? To be honest, my view is that estimating biodiversity loss is no general priority. Biodiversity is an enormously important phenomenon, and doing something about the human infringement on biodiversity is an enormously important task, but “estimating biodiversity loss” on a general level does not help this endeavor in any way. McGill *et al.* (2015) specify fifteen forms of biodiversity trends that conservation ecologists ought to pay attention to. In effect, they also step off from the goal of a unified assessment.

As regards critical temporal frames, a basic task is to draw distinctions between different types of threats to biodiversity; McGill *et al.* (2015) give a timely reminder on this need. Some of the factors bring about risks that can be quantified, however coarsely. This is different from genuine uncertainty that cannot be quantified. The latter situation approaches the setting of “post-normal science,” described by Funtowicz and Ravetz (1993); for specifications concerning biodiversity research, see Haila and Henle (2014). A basic question that brings genuine uncertainty into the concern over biodiversity is: what are the critical timeframes for avoiding the possible collapse of key mechanisms that maintain a livable biosphere for a rich biota that includes humans?

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