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Procrustean beds and empty boxes: on the magic of creating environmental data

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2

3 **Keywords:** biodiversity databases, knowledge-gaps, conservation policies, data sharing policies

4

5 **Abstract** – This article explores a dark side of the current enthusiasm for compiling large
6 datasets in support of evidence-based conservation, at various scales. We use a series of concrete
7 examples to show how data gathered in biodiversity databases can be poorly informative for the
8 design and implementation of effective conservation strategies and actions. This is due to two
9 mechanisms. The first is the use of ill-advised formats, compelling contributors to fill in data
10 which are at odds with the ecological information that they are truly able to capture. The second
11 corresponds to the fact that, unless knowledge gaps are explicitly and emphatically highlighted,
12 the elaboration of databases bestows visibility on knowledge and invisibility on knowledge gaps.
13 Given these risks, we call for a cultural shift among conservation practitioners, consultants and
14 others, to embrace the idea that documenting and acknowledging knowledge gaps and
15 uncertainties is just as important as compiling data and taking known information into account.
16 We also propose a series of concrete reforms to address these risks. In particular, we point the
17 need for procedural improvements in the process through which conservation databases are
18 elaborated and we suggest introducing differentiated data sharing policies for conservation
19 databases.

20

21 **1. Introduction**

22 Fifteen years ago, Sutherland et al. (2004) published a manifesto calling for a revolution in
23 conservation decision-making. They argued that little evidence on the effects of conservation

24 actions was collected, let alone compiled in databases providing feedback to others, and that
25 most conservation decisions were consequently based on “myths”. The latter could refer either to
26 ideas originating from purely theoretical reasoning rather than from empirical evidence (echoing
27 the classical philosophical opposition between idealism and empiricism), or more prosaically to
28 beliefs enshrined in conservationists’ minds, based on common sense or on anecdotal evidence.
29 Sutherland et al. (2004) advocated that conservationists should set-up databases gathering
30 information on conservation practices and base their decisions on the evidence such databases
31 contain. Ferraro and Pattanayak (2006) followed suit by calling for empirical “evaluations of
32 biodiversity conservation investments.” Such manifestos are part of the broader enthusiasm,
33 emerging in the 2000s, for “evidence-based policy making” (De Marchi et al., 2016), defined as
34 an approach that “helps people make well-informed decisions about policies, programs and
35 projects by putting the best available evidence from research at the heart of policy development
36 and implementation” (Davies, 1999). This approach is broader than concerns by Sutherland et al.
37 (2004) in the sense that, beyond encouraging the collection and analysis of *ex post* evidence of
38 the efficiency or inefficiency of conservation actions, it promotes the generalization of
39 measurements at all stages in environmental policy design, implementation and evaluation.
40 Witnessing the success of this approach in environmental policies over the past 15 years, there
41 has been a proliferation of publications dedicated to evidence-based conservation (e.g. Dicks et
42 al., 2014; Matzek et al., 2014, and even a new journal, *Conservation Evidence*), but also of
43 databases aimed at gathering conservation evidence (www.conservationevidence.com,
44 www.environmentalevidence.org) and, more broadly, environmental data, such as the databases
45 of the IUCN Red List (www.iucnredlist.org), the Living Planet Index (LPI;
46 <http://www.livingplanetindex.org>), Key Biodiversity Areas (KBA;

47 <http://www.keybiodiversityareas.org>), Global Biodiversity Information Facility (GBIF;
48 www.gbif.org), Protected Planet (www.protectedplanet.net), and Delivering Alien Invasive
49 Species Inventories for Europe (DAISIE; www.europe-aliens.org).

50 This evolution is also reflected in policy making, which increasingly includes *ex ante* (impact
51 assessment) and *ex post* (fitness check) evaluation requirements, buttressed by more or less
52 precisely specified data-gathering mechanisms. The European Union's biodiversity conservation
53 policy structured by the two Natura 2000 Directives (the Birds Directive and the Habitat
54 Directive), is a prominent example (European Commission, 2016).

55 Although many such conservation databases provide valuable support to conservation policies
56 (Hoffmann et al., 2008), they have been criticized on several counts, mainly by researchers
57 concerned with improving them (Boitani et al., 2015; Troudet et al., 2017; Trull et al., 2018). An
58 important issue are the numerous biases plaguing these databases (Yang et al., 2013; Beck et al.,
59 2014; Meyer et al., 2015; McRae et al., 2017; Proença et al., 2017). Many of the databases
60 focused on species ecology, distribution and conservation status have gaps in species coverage
61 (global IUCN Red List, LPI; GBIF) or uneven data coverage among species (PanTHERIA,
62 FishBase; GBIF), which makes them highly biased towards developed countries, better studied
63 species groups, charismatic and economically valuable species (Troudet et al., 2017). For
64 example, although it covers most of the species in some groups such as birds, the global IUCN
65 Red List contains information on less than 50% of fish species (16,803 species in IUCN Red List
66 as compared to 34,100 species in FishBase, or 35,604 species in the Eschmeyer's Catalog of
67 Fishes; Fricke et al., 2019), and those included are highly biased in ways described above. Gaps
68 in species coverage and data are even larger in the case of invertebrates and some other species
69 groups. Such biases are to a great extent driven by the uneven scientific and societal taxonomic

70 attentions, which in turn drives the amount of knowledge and available data generated (Jarić et
71 al., 2019).

72 Beside incomplete and biased coverage, information on the reliability of the data is often
73 lacking: i.e., precision of sighting locations or value estimates, specimen identification
74 uncertainty, life history data reliability, sample size that the data was based on, information on
75 specific data sources and identity and experience of assessors, which could be expressed, for
76 example, through confidence intervals, probability values, or reliability scores. Individual data
77 entries can vary widely in their reliability (Heeren et al., 2017).

78 Despite these limitations, databases such as the IUCN Red List or KBA are central to many
79 conservation decisions, including those made by the private sector (Bennun et al., 2018).
80 Together with other databases, they form the basis for biodiversity risk screening by investors,
81 either directly (e.g. through IBAT - <https://www.ibat-alliance.org/>) or through *ad-hoc* software
82 such as LEFT (Long et al., 2018) or TESSA (Peh et al., 2013) which compile and analyze global
83 datasets.

84 In this article, we explore a dark side of this enthusiasm for data gathering, which has been
85 neglected by the above literature. Based on a review of the academic and grey literature, our own
86 experience as practitioners and discussions with other practitioners, we show that, in some cases,
87 important information is lost because of the chosen format and an impression of knowledge ends
88 up hiding knowledge gaps. The following two sections use concrete examples to illustrate the
89 two main mechanisms generating these problems.

90

91 **2. Procrustean beds: creating uninformative data through ill-advised formats**

92 A first mechanism is the use of poorly designed formats, compelling contributors to fill in data
93 which are at odds with the ecological information that they are truly able to capture. We term
94 such cases “Procrustean beds”, echoing the Greek mythological figure of Procrustes, a rogue
95 who used to stretch people or cut their legs off to make them fit a bed. In the policies we analyze,
96 experts’ insights are stretched and cut off to make them fit in a database.

97 A first illustration is given by the GWERN software (<http://www.forum-zones-humides.org>), which is
98 now used to standardize wetland mapping across France. Using the software and database is a
99 precondition imposed by several public agencies to finance conservation actions targeting
100 wetlands. Experts using this software have to assess if the various wetlands perform “hydraulic
101 functions”, such as “natural regulation of flooding events” or “protection against erosion”,
102 “biological functions”, such as “carbon storage”, among numerous other functions, and if these
103 functions are performed to a “low”, “middle”, “high” or “non-significant” degree. Due to the
104 imposed format, uncertainties are silenced and reporting the reason for choosing one answer over
105 another is optional. Rigorously filling out the information would require measurements and
106 investigations that experts in the field are never in a position to perform. The result of the
107 generalization of this software is that a large amount of data on wetlands is now organized as a
108 database with a coherent structure, which is certainly positive, but its relevance for evidence-
109 based conservation action is questionable. Similar (albeit less formalized) approaches are
110 implemented in other countries, such as Portugal (C. Pinto Gomez, personal communication) and
111 Switzerland (R. Delarze, personal communication).

112 The Standard Data Forms (SDF) used to collect information on Natura 2000 sites
113 (http://ec.europa.eu/environment/nature/natura2000/sites/index_en.htm), similarly illustrate the
114 problems associated with databases asking experts to report synthetic assessments rather than

115 concrete data. Experts are required to provide synthetic assessments of the “degree of
116 conservation” of the “structure”, “function” and "restoration possibility” of habitats. Such
117 parameters are not directly observable. They can be obtained by aggregating information
118 observable in the field and through GIS, or through expert judgments. The methodologies used to
119 choose and compile these information are not standardized (Tsiripidis 2018), and SDF do not ask
120 countries to report information on the observable data used to compute them (if any). As a
121 consequence, users do not have access to the concrete ecological parameters measured and there
122 is an incentive for Member States to use cheaper, unsupported expert assessments.

123 The difficulty of capturing meaningful ecological information is similarly trivialized in some
124 usages of the EUNIS habitat classification across Europe. EUNIS is based on different habitat
125 typologies, which were combined and structured through series of expert meetings (Rodwell et
126 al. 2018). Habitat typologies are a prerequisite to prioritizing habitat types for protection, and our
127 point here is not to discuss their scientific credentials. Rather, we want to stress a collateral effect
128 of the existence of such typologies. In the case of terrestrial habitats, an important part of work
129 that led to the EUNIS classification was to buttress this common European-wide classification by
130 a robust phytosociological vegetation classification (Rodwell et al. 2018). However, the
131 denomination of items in EUNIS uses a more easily understandable vocabulary and does not
132 refer to phytosociological categories (although links between the two classificatory frameworks
133 are given). Davies et al. (2004) also elaborated keys clarifying the parameters to distinguish the
134 different habitats. These parameters (e.g. origin of water in bogs, inundation frequencies in
135 grasslands, whether a vegetation community is oligotrophic or not, or thermophilous or not, etc.)
136 are mostly not directly observable in the field. They are rather synthetic parameters that experts
137 can infer on the basis of repeated visits and measurements, or can hypothesize on

138 phytosociological bases. However, such keys make it look as though any expert with a basic
139 ecological knowledge can apply the EUNIS typology, whereas applying phytosociological
140 categories actually requires specialized botanical expertise. As a consequence, managers
141 increasingly use the EUNIS typology when conducting habitat surveys, which is encouraged by
142 some methodological guidelines (e.g. ATEN, 2018). A recent technical report (Gayet et al.,
143 2018b) explicitly endorses this approach by claiming that most habitat identifications in EUNIS
144 terms can be performed by experts with basic ecological knowledge (except for identifications at
145 level four, five, six and seven of this nested typology, which are admitted to require specialized
146 botanical knowledge).

147 The various examples above show that, by bypassing the need for specialized expertise to
148 capture information on habitats and ecological functioning, some supposedly conservation-
149 relevant databases end up compiling data of doubtful ecological significance. A similar
150 trivialization of ecological complexity can be witnessed in the case of species, as unveiled by
151 Devictor and Bensaude-Vincent (2016) in the case of the GBIF.

152

153 **3. Empty boxes: creating the appearance of knowledge**

154 A second mechanism is the construction of large-scale databases which are largely empty. Our
155 point here is not to denounce knowledge-gaps in such databases. Instead, our point is that, given
156 the existence of such knowledge-gaps, if the latter are not emphatically highlighted, gathering
157 information in a database tends to render knowledge visible and knowledge-gaps invisible.

158 The European Union Natura 2000 database ([https://www.eea.europa.eu/data-and-](https://www.eea.europa.eu/data-and-maps/data/natura-9)
159 [maps/data/natura-9](https://www.eea.europa.eu/data-and-maps/data/natura-9)) which gathers the information contained in SDF, illustrates that the
160 existence of the database itself creates an illusion of knowledge. The database contains a total of

161 399 210 assessments for species and 148 444 for habitats. However, the data are reported to be
162 of “good” quality for less than 42% of habitats, and less than 20% of species. In poorly-known
163 groups such as invertebrates, only 12.3% are of “good” quality. Besides, the information given
164 on the quality of data is based on vague categories that can be variously interpreted: ‘Good’ is
165 defined as “based on surveys”; ‘Moderate’ as “based on partial data with some extrapolation”;
166 ‘Poor’ as “rough estimation”. No references or even explanations are provided to support quality
167 assessments. The database elaborated as a part of the periodical European-wide evaluation of the
168 Natura 2000 policy for 2007-2012 ([https://www.eea.europa.eu/data-and-maps/data/article-17-
169 database-habitats-directive-92-43-ee-1/article-17-database-zipped-ms-access-format/article-17-
170 database-zipped-ms-access-format](https://www.eea.europa.eu/data-and-maps/data/article-17-database-habitats-directive-92-43-ee-1/article-17-database-zipped-ms-access-format/article-17-database-zipped-ms-access-format)), illustrates a similar problem. Among the 3117 evaluations of
171 “structure and function” of habitats, only 359 are based on “Complete survey or a statistically
172 robust estimate”, and it is unclear how to distinguish, at a technical level, between a “Complete
173 survey or a statistically robust estimate” and an “Estimate based on partial data with some
174 extrapolation and/or modelling”. The database does contain a field where “published data
175 sources” should be reported, but this is an unstructured field gathering methodological reports
176 concerning the various aspects of the evaluation, multiple data sets and even historical surveys of
177 indirect relevance to the assessment.

178 Another database provides an even more striking illustration: the French database of areas of
179 particular conservation interest (ZNIEFF: *Zones Naturelles d’Intérêt Ecologique Faunistique et*
180 *Floristique*: inpn.mnhn.fr/programme/inventaire-znieff/presentation), analogous to “Key
181 Biodiversity Areas”. ZNIEFF sites are largely seen as a nation-wide reference and are routinely
182 used to elaborate and spatialize conservation strategies such as sub-national “green and blue
183 vein” ecological corridors (Amsallem et al., 2010) or “no net loss” goals applicable to

184 development projects (Quétier et al., 2014). The ZNIEFF database provides species and habitat
185 data for 18 185 sites and an assessment of the state of knowledge about various groups for each
186 site. For the two best known groups (mammals and birds), that knowledge is considered “good”
187 for only 2.5% of the sites (N=462). Unsurprisingly, the figures are even lower for less popular
188 groups: 0.1% for mosses (21 sites) and 0.07% for non-insect invertebrates (12 sites).

189 Outside Europe, the REBIOMA database (<http://data.rebioma.net>), which includes an index of
190 current and likely future habitat suitability for a large number of species in Madagascar, gives the
191 impression of there being a lot of adequate and useful data on species occurrence for the purpose
192 of conservation planning. MaxEnt (a software predicting species niches and distribution through
193 maximum entropy modeling) is used to produce continuous distribution models based on
194 occurrence records and an apparently generic set of relatively coarse-scale environmental data
195 layers. While this may result in useful outputs for many taxa, especially well-surveyed ones, the
196 data for many others appears inadequate for the purpose. For example, the habitat suitability
197 models for *Asteropeia rhopaloides* and *Sanzinia madagascariensis* respectively appear to be
198 based on 4 and 6 records only, yet they are used to map habitat suitability at the national scale.
199 Our point is not to recall the trivial fact that habitat niche models can be misused when input data
200 are insufficient, but to emphasize that gathering the results of such models in a database can give
201 the illusion that a good level of knowledge on current and future distributions is actually
202 available. Although the known records and models are accessible for each species, these are not
203 made obvious to a user who is only looking for the maps, which are also not clearly marked with
204 any uncertainty or reliability factor, based on the underlying data This makes it difficult to
205 distinguish between robust outputs and those that are less reliable.

206 Moreover, databases aimed at supporting conservation policy and management also often contain
207 pooled, aggregated data, where detailed information is lost. This is illustrated by the species
208 distribution data in DAISIE, which is in most cases provided only at the country-level, as
209 presence/absence. It lacks information on data reliability, species coverage, differences in data
210 availability among countries and regions, and often the identity and experience of assessors.

211 Data in databases are in some cases accompanied with references to source publications. This
212 may give the impression that the problem of uncertainty and data-gaps is accounted for, but this
213 is not necessarily true in practice as sources cannot be readily checked for reliability and
214 uncertainty, especially for large databases comprising thousands or millions of data and related
215 sources. In such cases, the data that can be used to account for uncertainty is essentially
216 provided, but the dataset size will render their use impossible. This problem is present in most of
217 the existing databases used for conservation decision-making, including those discussed
218 previously in the text. For such information on data sources to be usable, it would have to be
219 extracted and provided to users in a ready-to-use format, with all the ancillary information,
220 including the estimate of data reliability and uncertainty.

221 In some cases, relevant and detailed data can be included in the database, but not made freely
222 available to users. For example, IUCN Red List assessments are commonly based on input by
223 multiple species experts (“contributors”), ranging from evidence-based estimates to basic rule-
224 of-thumb approaches, which are in turn collated and aggregated by one or several expert
225 coordinators (“assessors”) and, together with data from the available literature, used to estimate
226 species status and trends. However, the detailed data used for assessments is not made available
227 to the users, who are provided only with the final outcome of the analysis and a general

228 overview. Detailed information on the contributors and the data used for estimates is stored in
229 the internal database that is generally off limits to general users.

230

231 **4. The way forward: facing the scarcity of conservation information**

232 The various databases mentioned above play important roles in environmental policies and
233 conservation at various scales. In spite of the problems unveiled above, they are still very
234 valuable in providing current available knowledge to conservation science and practice. Our
235 point is neither to discredit these initiatives, nor to argue that they should be discarded. We rather
236 aim to suggest avenues to address the risks we have highlighted.

237 Valuable progress could be achieved by improving the process through which data formats are
238 produced and integrated into public procurement rules, in particular by including experienced
239 field experts in this process. In the case of wetland assessment, numerous methods based on field
240 measurement are elaborated (e.g. Gayet et al., 2018a) and could compete with databases such as
241 GWERN in the years to come. The UK's National vegetation survey methodology
242 (<http://jncc.defra.gov.uk/page-4258>), which pursues objectives similar to EUNIS cartographies,
243 illustrates how the requirement for the elaboration of vegetation maps that “each surveyor should
244 be a competent botanist” (<http://jncc.defra.gov.uk/page-2468>, p.14) is not unrealistic, even if not
245 always effectively applied or sufficient (e.g. Wawrzyczek et al., 2018). Similarly, although we
246 criticized above the vagueness of data quality assessments in Natura 2000 and Article 17
247 databases, they are a step in the right direction, and fact sheets of European Red Lists of Habitats
248 (https://forum.eionet.europa.eu/european-red-list-habitats/library/index_html) illustrate progress
249 in this respect.

250 Involving experts in ecology and computer sciences in the design of databases is needed to
251 eliminate Procrustean beds. It is, however, insufficient to address empty boxes. Indeed, the
252 ZNIEFF database, for example, was produced through an extensive consultation involving
253 hundreds of experts (Elissalde-Videment et al., 2007).

254 A cultural shift is also needed among conservation practitioners, consultants and others, to
255 embrace the idea that documenting and acknowledging knowledge gaps and uncertainties is just
256 as important as compiling data and taking known information into account. Knowledge gaps
257 should play just as important a role in supporting conservation policies and actions. This echoes
258 Pe'er et al. (2014) call for “a different attitude to uncertainty”. Whereas empty boxes illustrate
259 the risk of misuse of incomplete information, Procrustean beds illustrate how seeking an
260 unachievable certainty leads to the creation of uninformative data. Overcoming these problems
261 involves recognizing the positive aspects of uncertainty and facing hard truths concerning the
262 difficulties it can generate.

263 Such a shift is currently hindered by the fact that the majority of existing databases cannot be
264 fully understood given the lack of direct links between the assessments and the data sources and
265 raw information they are based on. While such a comprehensive database structure would
266 probably be challenging to implement, an alternative would be to provide ancillary information
267 that would reflect quality and characteristics of data sources, such as reliability, value ranges,
268 confidence intervals, uncertainty, etc. While there is a certain unavoidable level of trade-off
269 between the ecological quality and interoperability of data, efforts should be made to ensure that
270 proper integration, harmonization and interoperability does not compromise ecological meaning
271 of the stored information, and that metadata management and standardization also reflect
272 ecological meaning (Kissling et al., 2018). There is a growing number of promising initiatives

273 focused on these issues, such as Darwin Core and Humboldt Core, the two standards for
274 publishing and integrating biodiversity data and inventories (Wieczorek et al., 2012; Guralnick et
275 al., 2018), and Ecological Metadata Language, a widely used metadata standard (Michener,
276 2006).

277 Another important improvement would be to use differentiated data sharing policies, as initiated
278 by Botanical Conservatories in France for other purposes. The Botanical Conservatories have
279 important databases on occurrences of plant species and habitats. This information is freely
280 available in a grossly aggregated form. In this aggregated form, these data are not usable to
281 inform management plans or even regional scale environmental strategies. More detailed
282 information can be obtained by signing a memorandum of understanding (MOU) specifying that
283 the data cannot be used outside the specific scope of the project for which the MOU was
284 established, and various requirements designed to prevent misuses of the data. A similar
285 approach could be implemented for many databases mentioned here, so as to impose
286 requirements to take account of knowledge gaps and uncertainties in data analyses. In the case of
287 the ZNIEFF data, for example, access to detailed spatial data could be conditioned by making
288 explicit analyzes and explanations of data gaps and uncertainties, and their implications. This is a
289 readily enforceable requirement by data providers, and could find its way in legally binding
290 requirements for conservation strategies, management plans or environmental impact
291 assessments.

292

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