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## A roadmap for a quantitative ecosystem-based environmental impact assessment

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► **To cite this version:**

J. Coston-Guarini, J.-M. Guarini, Shawn Hinz, Jeff Wilson, Laurent Chauvaud. A roadmap for a quantitative ecosystem-based environmental impact assessment. *ICES Journal of Marine Science*, 2017, 74 (7), pp.2012-2023. 10.1093/icesjms/fsx015 . hal-02572317

**HAL Id: hal-02572317**

**<https://hal.science/hal-02572317>**

Submitted on 13 May 2020

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1 TITLE

2 A roadmap for a quantitative ecosystem-based environmental impact assessment

3

4 SUBMITTED TO:

5 ICES Journal of Marine Science “Special MSEAS-2016 issue”

6 as an ORIGINAL ARTICLE

7

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47 VERSION

48 1 Sept 2016

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50 ABSTRACT

51 A new roadmap for quantitative methodologies of Environmental Impact Assessment  
52 (EIA) is proposed, using an ecosystem-based approach. EIA recommendations are  
53 currently based on case-by-case rankings, distant from statistical methodologies, and  
54 based on ecological ideas that lack proof of generality or predictive capacities. These  
55 qualitative approaches ignore process dynamics, scales of variations and  
56 interdependencies and are unable to address societal demands to link socio-economic  
57 and ecological processes (*e.g.* population dynamics). We propose to re-focus EIA  
58 around the systemic formulation of interactions between organisms (organized in  
59 populations and communities) and their environments but inserted within a strict  
60 statistical framework. A systemic formulation allows scenarios to be built that simulate  
61 impacts on chosen receptors. To illustrate the approach, we design a minimum  
62 ecosystem model that demonstrates non-trivial effects and complex responses to  
63 environmental changes. We suggest further that an Ecosystem-Based EIA - in which the  
64 socio-economic system is an evolving driver of the ecological one - is more promising  
65 than a socio-economic-ecological system where all variables are treated as equal. This  
66 refocuses the debate on cause-and-effect, processes, identification of essential portable  
67 variables, and a potential for quantitative comparisons between projects, which is  
68 important in cumulative effects determinations.

69

70 KEYWORDS

71 Environmental Impact Assessment, ecosystem, drivers of change, modelling, socio-  
72 ecological system

73

## 74 INTRODUCTION

75 When the USGS hydrologist and geomorphologist Luna Leopold (1915-2006) and his  
76 two co-authors published a system for environmental assessment in 1971 (Leopold *et*  
77 *al.*, 1971), they could not have foreseen that 50 years later, their report would be at the  
78 origin of a global industry (Morgan, 2012; Pope *et al.*, 2013). Leopold *et al.* produced  
79 their brief document at the request of the US Department of the Interior after the  
80 National Environmental Policy Act (NEPA) created a legal obligation for federally  
81 funded projects to assess impact. In the year following the passage into law, the  
82 scientific community was quick to point out the absence of any accepted protocol for  
83 either the content of the document or its evaluation (see characterisation in Gillette,  
84 1971). In response, Leopold *et al.* describes a preliminary approach, with a simple  
85 decision-tree like diagram (Figure 1A) relying on structured information tables. These  
86 tables of variables and qualities, or ‘interaction matrices’, are intended to enforce  
87 production of uniform, comparable descriptions, while requiring only a minimum of  
88 technical knowledge from the user.

89 Impact inference rests on a statistical comparison of variables between impacted and  
90 non-impacted sites, but assessing an impact is understood to include value-based  
91 judgements about quality and importance (Leopold *et al.*, 1971) linked with attitudes  
92 held about the environment (Buttel and Flinn, 1976; Lawrence, 1997; Toro *et al.*, 2013).  
93 These judgements, often made *a priori* (Toro *et al.*, 2013), can conflict with the  
94 necessity to reach a legal standard of proof (Goodstein, 2011) when projects are  
95 contested. EIAs therefore embody a compromise between technical descriptions of the  
96 expected magnitude of an impact on a receptor and managerial recommendations about  
97 how to avoid that receptors exceed acceptable values, or mitigate, identified impacts  
98 (Lawrence, 1997; Cashmore *et al.*, 2010; Barker and Jones, 2013). By 1971, under

99 pressure to move development projects forward (Gillette, 1971), the EIA process  
100 became institutionalised as a qualitative exercise focussed on collecting documentation  
101 about a project site supported by individuals' professional expertise, without requiring  
102 quantitative evaluations to back up statements (Lawrence, 1997; Cashmore *et al.*, 2010;  
103 Morgan, 2012; Toro *et al.*, 2013). Hence EIAs today still strongly resemble the  
104 preliminary instructions given by Leopold *et al.* (Figure 1B). Consequently, review  
105 articles, such as that of Barker and Jones (2013) on offshore EIAs in the UK, often  
106 report strong criticisms of the quality of environmental impact documents as being  
107 “driven by compliance rather than best practice”.

108 Over the past decade, technologically sophisticated monitoring tools and baseline  
109 surveys have been integrated (*e.g.* Figure 1B, “Modelling”; Payraudeau and van der  
110 Werf, 2005; Nouri *et al.*, 2009) on a discretionary basis because they contribute to risk  
111 management of sensitive receptors as well as to new dynamic features like the “Life  
112 Cycle Assessment” of a project (Židonienė and Kruopienė, 2015). These changes  
113 suggest that EIA is poised to incorporate quantitative frameworks.

114 Inspired by the application of ecosystem-based management frameworks in fisheries  
115 (Smith *et al.*, 2007; Jacobsen *et al.*, 2016), and by the generalisation of modelling and  
116 statistical tools in ecological and environmental sciences, we describe in this article how  
117 the objective of a quantitative, ecosystem-based EIA could be achieved. We first  
118 examine briefly the awareness of impact and analytical approaches that exist to quantify  
119 this within ecological sciences. We then propose a quantitative reference framework  
120 linking statistical impact assessment to ecosystem functioning and discuss how the  
121 modelling approach may be used to provide reasonable predictions of different  
122 categories of impact. Finally, we explore how our ecological system will behave when  
123 socio-economic “drivers of change” (UNEP, 2005) are implemented. By imposing

124 socio-economic factors as drivers (instead of as variables of a large integrated system),  
125 we show that different types of consequences can occur, which are not represented by  
126 classical feedbacks. For example, this permits the life cycle of the project to be  
127 described as a driver of the dynamic of the impacted system, or the explicit  
128 implementation of cumulative effects scenarios.

129 **Awareness of environmental impact in the past.** There is a long written record of the  
130 awareness that human activities affect the environment. Texts of 19<sup>th</sup> century naturalists  
131 commonly contain remarks about the disappearance of animals and plants attributed to  
132 human activities; some are quite detailed, like George P. Marsh's quasi-catalogue of the  
133 ways "physical geography" (natural environments) has been altered by development  
134 (Marsh, 1865). Most are ancillary comments to make rhetorical points, rather than  
135 scientific observations, like this quote from the marine zoologist Henri de Lacaze-  
136 Duthiers (1821-1901) (de Lacaze-Duthiers, 1881: 576-577):

137 *"Ainsi, lorsque sera crée la nouvelle darse, qui n'a d'autre but que d'augmenter le*  
138 *mouvement du port, que deviendront les localités tranquilles où la faune était si riche ?*  
139 *Resteront-elles les mêmes ? l'eau ne se renouvelant pas, n'aura-t-elle pas le triste sort*  
140 *de celle des ports de Marseille, si le commerce et les arrivages prennent de grandes*  
141 *proportions ?*

142 *"Le mouvement du port augmente tous les jours. Les constructions des darses projetées*  
143 *ne modifieront-elles pas les conditions favorables actuelles ? On doit se demander*  
144 *encore si l'eau conservera son admirable pureté quand le nombre des bâtiments aura*  
145 *augmenté dans les proportions considérables que tout fait prévoir.*

146 “*Port-Vendres ne peut évidemment que se modifier profondément dans l’avenir, et cela*  
147 *tout à l’avantage du commerce, c’est-à-dire au détriment de la pureté, de la tranquillité*  
148 *de l’eau et du développement des animaux.*”

149 *A Banyuls, il n’y a aucune crainte à avoir de ce côté.”*

150 When he wrote this, Lacaze-Duthiers had been lobbying for more than a decade for the  
151 creation of a network of marine stations in France. His text justifies why he chose a  
152 village without a port, instead of one with a thriving port. His reasoning is that  
153 economic development causes increases in buildings, docks, boat traffic, that damages  
154 the “tranquillity”, “water purity”, and the “favourable conditions for development of  
155 fauna”. While he acknowledges this is a gain for local commercial interests, it is also at  
156 the expense of faunal richness, and he predicts this will lead to the “sad situation of the  
157 port of Marseille”. Lacaze-Duthiers feels this degradation should be a legal issue or a  
158 civil responsibility (as “*au détriment de*” indicates a legal context). The attitude and  
159 awareness of Lacaze-Duthiers are symptomatic of ambiguities about the environment  
160 (Nature) and the place of humans in it, that are also at the core of EIA (Cashmore, 2004;  
161 Wood, 2008; Morgan, 2012; Toro *et al.*, 2012). These political conflicts between a  
162 desire to preserve the natural world and its own functioning, and the desire to use,  
163 exploit, order and control parts of it are the main issues of impact assessment  
164 (Cashmore *et al.*, 2010).

165 **Path to reconciliation.** What changed in the latter half of the 20<sup>th</sup> century is that  
166 managers, regulators and stakeholders need to document and quantify impacts as well as  
167 their associated costs. However, important, historical contingencies complicated the  
168 development of quantitative tools for environmental impact. Ecosystem science, which  
169 pre-dates EIA by several decades, describes ecosystem functioning in terms of energy



170 and mass flows (*e.g.* Odum, 1957) and the distribution of species is understood with  
171 respect to how well the ‘conditions of existence’ of a population are met and maintained  
172 (*e.g.* Gause, 1934; Ryabov and Blasius, 2011; Adler *et al.*, 2013). These approaches use  
173 paradigms from biology, physics and chemistry to describe functions and quantify  
174 fluxes. Consequently, ecosystem science was not concerned with characterising  
175 environmental quality, but determining when conditions of existence were met within  
176 dynamic, interacting systems. By the 1970s when EIA practice emerged, ecological  
177 research was busy with adaptation and community succession (Odum, 1969; McIntosh,  
178 1985), while the concepts of environmental quality and impact were being defined  
179 under a “political imperative, not a scientific background” (Cashmore, 2004: 404) using  
180 static components like receptors and indices.

181 Today, several very different, co-existing strategies exist with regards to environmental  
182 management and conservation: ecosystem functioning (*e.g.* Moreno-Mateos *et al.*,  
183 2012; Peterson *et al.*, 2009), ecosystem services and markets analysis (*e.g.* Beaumont *et*  
184 *al.*, 2008; Gómez-Baggethun *et al.*, 2010), and environmental impact. In this context,  
185 knitting together sociological and ecological frameworks has emerged as a very active  
186 area of interdisciplinary research (Binder *et al.*, 2013). An important theme has been to  
187 re-conceptualise environmental dynamics from an anthropogenic perspective to counter  
188 a perception that human activities have been excluded from ecological studies (Berkes  
189 and Folke, 1998; Tzanopoulos *et al.*, 2013). While this is clearly an unfair  
190 characterization (the classic introductory American text on ecology is entitled “Ecology:  
191 The link between the natural the social sciences”; Odum, 1975), we do recognize that,  
192 historically, ecological sciences have often ignored human behaviours and attitudes in  
193 ecosystem studies, despite numerous appeals (Odum, 1977; McIntosh, 1985; Berkes  
194 and Folke, 1998). Inspired by the criticisms of Lawrence (1997) about EIA and the

195 challenge of working between both sociological and ecological systems (Rissman and  
196 Gillon, 2016), we propose a quantitative basis for systems-based impact assessment.  
197 Our goal is to renew the understanding of impact in terms of the interactions and  
198 functions attributable to ecosystem processes, integrating the full dynamics of physical  
199 and biological processes, while allowing for effective evaluation of socio-economic  
200 dynamic alternatives within the modelling framework.

201

## 202 METHODOLOGY

203 **Receptors.** Assuming that the screening process has already demonstrated the  
204 requirement to perform an EIA for a given project, scoping identifies the receptors and  
205 the spatio-temporal scale of the study. Receptors are represented by variables being  
206 impacted by the project implementation. Receptors are determined by the experts in  
207 charge of the EIA. Their qualifications as receptors imply that they will be impacted and  
208 this cannot be questionable. In other words, what is called "testing" impact is  
209 statistically limited to a process of deciding if the observation data corresponding to  
210 samples of the receptor variables permits an impact to be detected. In no case should the  
211 selection of a receptor be made with the objective to decide *if there is* an impact or not.  
212 By definition, receptors are selected because they are sensitive to the impact. However,  
213 all declared receptor variables also represent objects of ecology and can be inserted into  
214 an ecosystem framework. These two points will now be reviewed in more detail,  
215 establishing an explicit link between them.

216 **Statistical rationale for impact assessment detection.** Impact assessment relies on  
217 statistical comparisons of receptor variables in impacted and non-impacted situations.  
218 Assuming that the expertise determined the nature of the impact (*i.e.* decreasing or

219 increasing the variable), the impact assessment consists of testing if the absolute  
220 difference,  $\Delta$ , between the non-impacted ( $\mu_0$ ) and the impacted variable means ( $\mu_1$ ) is  
221 greater than zero ( $H_1: \Delta = |\mu_0 - \mu_1| > 0$ ). Classical testing procedure leads not to accepting  
222  $H_1$ , but to rejecting  $H_0$  ( $H_0: \Delta = |\mu_0 - \mu_1| = 0$ ). However, the power of the test increases  
223 when  $\Delta$  increases, which means that the more the variable is sensitive, the greater the  
224 impact has a chance to be detected.

225 Ideally, as EIAs start before the project implementation, samples of receptor variables  
226 are collected before and, then after the project. We focused on this case even if  
227 sampling may also be carried out concurrently for comparing non-impacted and  
228 impacted zones. For a receptor variable  $x$ , considering two samples of sizes  $n_0$  (before  
229 implementation) and  $n_1$  (after implementation), the empirical averages are  $\bar{x}_0$  and  $\bar{x}_1$ ,  
230 respectively, and their standard deviations are  $s_0$  and  $s_1$ . The statistics of the test is then:

$$231 \quad y = \frac{|\bar{x}_0 - \bar{x}_1|}{s_0 \sqrt{n_0^{-1} + n_1^{-1}}} \quad [\text{Eq. 1}]$$

232 emphasizing the importance of the sample (before implementation), which is used to  
233 estimate the ‘baseline’. The dispersion around the average  $s_0$  has a crucial role in the  
234 calculation of  $y$  ( $y$  decreases when  $s_0$  increases). Besides the size  $n_0$  will be fixed when  
235 the project is implemented (*i.e.* it is impossible to come back to the non-impacted  
236 situation when the project is implemented), while  $n_1$  can be determined and even  
237 modified *a posteriori*.

238 Under  $H_1$  (hence when  $H_0$  is rejected),  $y$  is normally distributed,  $y \sim N(\Delta, 1)$ , and then it  
239 can be centred using:

$$240 \quad \psi = \frac{|\bar{x}_0 - \bar{x}_I| - \Delta}{s_0 \sqrt{n_0^{-1} + n_I^{-1}}} \quad [\text{Eq. 2}]$$

241 This allows us to state that  $\psi$  follows a Student law. Therefore the test leads to rejection  
 242 of  $H_0$  if  $\psi$  is greater than a threshold  $t_{v,\alpha}$ , where  $v$  is the number of degrees of freedom  
 243 ( $v = n_0 - 1$ ) and  $\alpha$ , the type 1 error (rejecting  $H_0$  when  $H_0$  is true), is  $\alpha = \text{proba}\{\psi > t_{v,\alpha} \mid$   
 244  $\Delta = 0\}$ . The type 2 error (failing to reject  $H_0$  when  $H_0$  is false) is then  $\beta = \text{proba}\{\psi > t_{v,\alpha} \mid$   
 245  $\Delta > 0\}$  and the power of the test is  $\pi = 1 - \beta$ .

246 As  $\psi$  follows a Student law:

$$247 \quad t_{v,1-\beta} = \frac{t_{v,\alpha} - \Delta}{s_0 \sqrt{n_0^{-1} + n_I^{-1}}} = -t_{v,\beta} \quad [\text{Eq. 3}]$$

248 Considering that the baseline is estimated by a sampling performed before  
 249 implementation, with  $n_0$  becoming a fixed parameter, the question of detecting  
 250 significantly the impact then consists of determining two unknown variables  $\Delta$  and  $n_I$  by  
 251 solving two functions:

$$252 \quad \begin{cases} \Delta = f(n_I, \alpha, \beta) \\ n_I = g(\Delta, \alpha, \beta) \end{cases} \quad [\text{Eq. 4}]$$

253 By introducing  $\delta = \Delta / \mu_0$ , the variation  $\Delta$  relative to the baseline, and  $C_0 = s_0 / \bar{x}_0$ , the  
 254 variation coefficient of the baseline sample, the system to solve is then:

$$255 \quad \begin{cases} \delta = (t_{v,\alpha} + t_{v,\beta}) C_0 \sqrt{n_0^{-1} + n_I^{-1}} \\ n_I = \frac{n_0 C_0^2 (t_{v,\alpha} + t_{v,\beta})^2}{n_0 \delta^2 - C_0^2 (t_{v,\alpha} + t_{v,\beta})^2} \end{cases} \quad [\text{Eq. 5}]$$

256 At this point in our development, we can make several remarks about how EIA

257 practices shape the calculation of the impact:

258 1. The change relative to the baseline ( $\delta$ ) is positive if  $\delta > C_0(t_{v,\alpha} + t_{v,\beta})/\sqrt{n_0}$ , and

259 hence  $\delta^* = C_0(t_{v,\alpha} + t_{v,\beta})/\sqrt{n_0}$  is the detection limit of the receptor variable which

260 can be calculated *a priori* (before impact).  $\delta^*$  is the smallest absolute relative

261 difference that can be characterized, and it depends only on  $s_0$  and  $n_0$  and the

262 choice of Type 1 and 2 errors. Therefore, the quality of the expertise, which

263 determines the receptors and the baseline, is a fundamental component of impact

264 assessment.

265 2. The parametric framework has many constraints (*i.e.* homogeneity and stability

266 of the variance, stability of the baseline ...), which have to be ensured, but is very

267 useful for establishing a link with modelling. In particular,  $\mu_0$  and  $\mu_I$ , hence  $\Delta$  and

268  $\delta$ , are descriptors of the states of the impacted ecosystem which can be simulated

269 by calculation from a deterministic model.

270 3. *A fortiori*, the change relative to the baseline,  $\delta$ , which depends on the nature of

271 the impact and the temporal scale of the observations, can be determined *a priori*

272 (or plausibly predicted) by the deterministic model. However it implies assuming

273 that the variations which create the dispersion around the trend of the variable are

274 white noises,  $e_t$  (defined by  $\{E(e_t) = 0, E(e_t^2) = s_0, E(e_{ti}, e_{tj}) = 0\}$ ). In this case, the

275 design of the ecosystem becomes particularly important, not only for diagnosing

276 the amplitude of the impact, but also the exact condition of the survey (*i.e.*

277 calculation of  $n_I$ ).

278 **Building an ecosystem model with receptors.** Our means to reconcile impact  
279 assessment with the theory of ecology is to replace the notion of receptors into a  
280 dynamic ecological model (Figure 2A). Receptors are placed in a network of  
281 interactions which represent an ecosystem. The “ecosystem” is a system in which the  
282 living components will find all conditions for their co-existence in the biotope (abiotic  
283 components and interactions that living organisms develop between themselves and  
284 with their environment). This classical definition (Tansley, 1935) encounters problems  
285 when translated into systemic frameworks. In particular, if the notion of co-existence is  
286 often linked to stable equilibrium, there is not one single definition of the notion of  
287 stability (Justus, 2008) and the precise nature of the complexity-stability relationship in  
288 ecosystems remains unsettled (Jacquet *et al.*, 2016).

289 Even with these caveats, the formulation is useful to explore a system-based EIA. First,  
290 stable equilibrium, for a given time scale (from the scale of the project implementation  
291 to the of the project life cycle scale) ensures that the baseline would not be subject to  
292 drift. Thus, variations will be due to the impact of the project and not by other sources.  
293 Secondly, spatial boundaries have to be determined such that the ecosystem has its own  
294 dynamics, even if it exchanges matter and energy with other systems. The stable  
295 equilibrium is then conditioned by the ecosystem states and not by external forcing  
296 factors. This last criterion ensures that the impact can be observable, and not masked by  
297 external conditions to the project. At the same time, boundaries are defined by the  
298 actual system under investigation and not by the presumed extended area influenced by  
299 the project.

300 For sake of simplicity, we proposed to consider a minimum ecosystem model (Figure  
301 2B). A minimum ecosystem has to ensure the co-existence of two populations: one  
302 population accomplishes primary production from inorganic nutrients, and a second

303 degrades detrital matter generated by the first population to recycle nutrients. Hence,  
304 there must be four different compartments (pool of nutrients (R), population of primary  
305 producers (P), population of decomposers (D) and a pool of detrital organic matter (M)),  
306 plus the corresponding four processes linking them, namely, primary production,  
307 mortality of primary producers, degradation of detrital organic matter, and  
308 remineralization (Figure 2B). Remineralization is linked to the negative regulation of  
309 the population of decomposers. Our ecosystem is considered as contained within a well-  
310 defined geographic zone (*e.g.* it has a fixed volume), receiving and dissipating energy,  
311 but not exchanging matter with the ‘exterior’. The energy source is considered  
312 unlimited and not limiting for any of the four biological processes. Finally, a generic  
313 process of distribution of matter and energy ensures homogeneity within the ecosystem.  
314 The formalism of signed digraphs (Levins, 1974) is employed in Figure 2B,  
315 emphasizing classical feedbacks as positive (the arrow) or negative (the solid dot)  
316 between compartments.

317 The minimum ecosystem defined as such, requires four variables: R, which represents  
318 the state of the nutrient pool, P, the state of the primary production population, M, the  
319 state of the pool of detrital organic matter, and D, the state of the decomposer  
320 population, and assumes that the units are all the same. The model is formulated by a  
321 system of four ordinary differential equations as:

$$322 \left\{ \begin{array}{l} \frac{dR}{dt} = -p \frac{R}{k_R + R} P + rD \\ \frac{dP}{dt} = +p \frac{R}{k_R + R} P - mP \\ \frac{dM}{dt} = +mP - dMD \\ \frac{dD}{dt} = +dMD - rD \end{array} \right. \quad [\text{Eq. 6}]$$

323 where  $p$  is a production rate ( $\text{time}^{-1}$ ),  $r$ , a remineralization rate ( $\text{time}^{-1}$ ),  $m$ , a primary  
 324 producers mortality rate ( $\text{time}^{-1}$ ), and  $d$ , a decomposition rate (unit of  $\text{state}^{-1} \cdot \text{time}^{-1}$ ). The  
 325 constant,  $k_R$  (units of  $R$ ) is a half-saturation constant of the Holling type II function  
 326 (Holling, 1959) that regulates intake of nutrients by primary producers. The ecosystem  
 327 is conservative in terms of matter; the sum of derivatives are equal to zero, hence  
 328  $R+P+M+D = I_0$ .

329 We then fix a set of initial conditions  $\{R_0, P_0, M_0, D_0\} \in \mathbb{R}^+$  which are the supposedly  
 330 known conditions at time  $t_0$ . Equilibriums were calculated when time derivatives are all  
 331 equal to zero [Eq. 7], and their stability properties are determined by studying the sign  
 332 of the derivative around the calculated solutions:

$$\begin{aligned}
 E_1 &: \{R^* = R_0, P^* = 0, M^* = M_0, D^* = 0\} \\
 E_2 &: \{R^* = R_0 + M_0 + D_0, P^* = 0, M^* = 0, D^* = 0\} \\
 333 \quad E_3 &: \{R^* = 0, P^* = 0, M^* = R_0 + P_0 + M_0, D^* = 0\} & \quad [\text{Eq. 7}] \\
 E_{4a} &: \left\{ \begin{aligned} R^* &= \frac{km}{p-m}, P^* = \left( I_0 - \frac{dkm+r(m-p)}{d(p-m)} \right) \left( \frac{r}{r+m} \right) \\ M^* &= \frac{r}{d}, D^* = \left( I_0 - \frac{dkm+r(m-p)}{d(p-m)} \right) \left( \frac{m}{r+m} \right) \end{aligned} \right\} \\
 E_{4b} &: \left\{ R^* = I - \frac{r}{d}, P^* = 0, M^* = \frac{r}{d}, D^* = 0 \right\}
 \end{aligned}$$

334 where  $R_0 > 0$ ,  $P_0 > 0$ ,  $M_0 \geq 0$  and  $D_0 > 0$ , and *a fortiori*  $I_0 = R_0 + P_0 + M_0 + D_0 > 0$ . All  
 335 five equilibriums listed above are stable and coexisting with the unstable trivial  
 336 equilibrium  $\{R^*=0, P^*=0, M^*=0, D^*=0\}$ .  $E_{4a}$  is reached if  $p > m$  and  $E_{4b}$  is reached  
 337 otherwise (assuming that the decomposers are acting fast with respect to the dynamics  
 338 of the entire system).  $E_1$ ,  $E_2$  and  $E_3$  equilibriums do not respect our definition of an  
 339 ecosystem:



- 340 •  $E_1$  is the case of no living organisms at the beginning (spontaneous generation is  
341 not allowed), and
- 342 •  $E_2$  and  $E_3$  are equilibriums with the initial absence of the primary producer or  
343 decomposer populations respectively, leading to the extinction of the other  
344 population (hence the condition of the co-existence of P and D is not fulfilled).

345 **Calculating changes in receptors and modelling the influence of drivers of change.**

346 In the model presented above, many receptor variables X can be identified. They can be  
347 the state variables (mainly representing the living populations, *i.e.* P or D) or the  
348 processes (like the ecosystem functions: primary production, decomposition and  
349 nutrient recycling). For all these variables, we calculated an impact as  $\delta = \Delta/X^*$ , the  
350 relative variation from the baseline  $X^*$ , consecutive to a virtual project implementation.  
351  $\Delta$  is the difference between two equilibrium values  $X^*$  to  $X^{**}$ , after a change in states  
352 (such as nutrient or detrital organic matter inputs) or parameters (mostly decreases in  
353 primary production rate, increases in primary producers' mortality rate, decrease in  
354 decomposition and recycling rates) consecutive to project implementation.

355 For the Environmental Impact Assessment, it is only required to know the amplitude of  
356 the changes consecutive to modifications of states or parameters to predict an impact on  
357 receptors. However, since we wish to include socio-economic aspects, we linked in a  
358 second step the change in ecosystem state and function to the possible influence of  
359 stakeholders on the project development (or the project 'Life Cycle'). The project  
360 development is controlled by groups of stakeholders, and the related "activity" depends  
361 on many factors that do not depend directly on ecosystem feedbacks (Binder *et al.*,  
362 2013).

363 Treating a ‘socio-economic-ecological system’ using systemic principles generates  
364 outcomes with little interest due to possible socio-economic feedbacks that are not  
365 connected as reactions to a physical system (*i.e.* "A" has an action on "B", and in return,  
366 "B" modifies "A", as in Figure 2B). We thus revise the notion of feedbacks by "A" has  
367 an action on "B" until "A" *realizes that* the action on "B" can be unfavourable to its own  
368 development. This formulation partly overlaps with the notion of “vulnerability”  
369 presented in Toro *et al.*, 2012 and “risk” (Gray and Wiedemann, 1999). The socio-  
370 economic system is introduced as a driver of change for the minimum ecosystem,  
371 instead of as a state variable like in other SES frameworks (Binder *et al.*, 2013).  
372 Consequences for the impacts on receptors are described in terms of the relative  
373 "activity" A ( $A \in [0, 1]$ ) of the project, related to the change in states or parameters by  
374 minimal linear functions (*i.e.* if  $x$  represents any potential change rates - in parameters  
375 or states - the effective change rates,  $y$ , are expressed by  $y = Ax$ ). The project activity is  
376 calculated as the complement of the relative socio-economic cost,  $C$ , of project  
377 development, expressed as:

$$378 \begin{cases} \frac{dC}{dt} = (\rho C + \sigma)(1 - C) \\ A = 1 - C \end{cases} \quad [\text{Eq. 8}]$$

379 where  $\sigma$  is a relative social awareness rate (increase, in  $\text{time}^{-1}$ , of the number of  
380 stakeholders aware of the negative consequences of the project within the total number  
381 of stakeholders), and  $\rho$  is the reactivity rate (the standardized speed, in  $\text{time}^{-1}$ , at which  
382 the socio-economic cost corresponding to mitigation or remediation measures  
383 increases).

384 All simulations and related calculations were performed using open source software  
385 (Scilab Enterprises, 2012).

386

## 387 RESULTS

388 **Examples of the impact predictions estimated by the model.** Three different  
389 scenarios were set-up for specific receptors (Table 1). Examining the steady-states of  
390 the system and their stability stresses the position of the set of parameters  $\theta = \{p, m, d,$   
391  $r, k_R\}$  and their relative importance in the definition of the system equilibrium. For  
392 building scenarios, it is assumed that the parameters' orders of magnitude are:

$$393 \quad p \gg m \gg r, \text{ and } r \approx d$$

394 Nonetheless,  $d$  is controlled by the quantity of substrate available.  $k$  is considered as  
395 small and the primary producers being assumed to have a good affinity for the available  
396 nutrients. When changes of parameters were simulated (as in Scenarios 2 and 3) they  
397 were varied in the same proportions. Inputs were simulated separately and then  
398 cumulated (CE), and their impacts on the 4 state variables at equilibrium ( $R^*$ ,  $P^*$ ,  $M^*$   
399 and  $D^*$ ) were examined.

400 The first scenario simulated direct inputs of nutrients and detrital organic matter.

401 Results show that in all cases,  $R^*$  and  $M^*$  did not vary (despite their initial increase). On  
402 the contrary, the variables representing living compartments,  $P^*$  and  $D^*$ , increased.

403 Results also show that the relative variation to the baseline,  $\delta$ , is identical for  $P^*$  and  $D^*$   
404 (both positive deviations, Table 1). Concerning processes at equilibrium, the primary  
405 production and the primary producer mortality both increased, as well as the processes  
406 of decomposition and recycling, since none of these parameters were affected by the  
407 project implementation.

408 The second scenario simulates an impact which consists of the decrease in primary  
409 producer performance. This could be due to the physiological capacities of the  
410 organisms being affected by the project or because the environmental conditions limit  
411 their expression (*e.g.* a strong increase in water column turbidity). In this situation, the  
412 parameters affected are  $k$  and  $m$  (which increased), and  $p$  (which decreased). It should  
413 be recalled that  $p$  was kept greater than  $m$  ( $p - m > 0$ ), as per our parameter hierarchy. A  
414 decrease of  $p$  and an increase of  $k$  (global decrease of primary productivity) always has  
415 a negative effect on  $P^*$  (hence on primary production), a positive effect on  $R^*$ , and a  
416 negative effect on  $D^*$ . In both cases, the relative variations to the baseline,  $\delta$ , are  
417 identical for  $P^*$  and  $D^*$ . An increase of  $m$  has a similar effect on  $P^*$  and  $R^*$ , but has a  
418 negative effect on  $D^*$ . The cumulative effect ( $p + m + k$ ) is almost equal in magnitude to  
419 the effect of a decrease in  $m$ , which is much higher (by several orders of magnitude)  
420 than the effects of  $p$  and  $k$ . Effects of  $p$  and  $k$  are quite negligible, each having a typical  
421 order of magnitude of the parameters in  $\theta$ .

422 The third and final scenario simulated a change in the decomposer activity. This could  
423 be triggered by a change in taxonomic composition, and also by the action of chemical  
424 substances released during the project. Decreases and increases in  $d$  and  $r$  were  
425 simulated, first separately and then together. Changes in  $d$  and  $r$  have no effect on  $R^*$ . A  
426 decrease of  $d$  as a negative effect on  $P^*$  (hence decreasing primary production) and  $D^*$ ,  
427 and logically, an increase of  $d$  has a positive effect on  $P^*$  (thus the increasing primary  
428 production) as well as  $D^*$ . In both cases, the relative variations to the baseline,  $\delta$ , are  
429 identical for  $P^*$  and  $D^*$ . Effects of a decrease or an increase in  $r$  on  $P^*$  and  $D^*$  are  
430 opposed.  $P^*$  increases and  $D^*$  decreases when  $r$  increases, and  $P^*$  decreases and  $D^*$   
431 increases when  $r$  decreases. Cumulative effects reinforce slightly the effect of a change  
432 in  $r$  which is largely predominant in the dynamics of  $P$  and  $D$ . The changes of  $d$  and  $r$

433 affect the primary production *via* a change in the availability of R. When the recycling  
434 is enhanced (mainly by the increase of  $r$  but also by an increase of  $d$ ), R production  
435 increases but an excess of R is used to increase the state of the primary producer P. It is  
436 because the production rate  $p$  is high compared to  $r$ , that  $R^*$  is not affected by changes  
437 in  $r$  or  $d$ . Changes in  $r$  and  $d$  have opposite effects on  $M^*$ . A decrease (respectively,  
438 increase) of  $d$  has a positive (respectively, negative) effect on  $M^*$ , and a single decrease  
439 (respectively, increase) of  $r$  has a negative (respectively, positive) effect on  $M^*$ . When  
440 changes are cumulated (in equal proportions), the effect of changes in  $r$  and  $d$  on  $M^*$  is  
441 null, showing that they have the same amplitude on  $M^*$ .

442 **Behaviour of system when drivers of change were included.** In the impact  
443 assessment *per se*, the effects of changes in ecosystems components (states and  
444 functions) were considered as a deviation of stable equilibrium values regardless of the  
445 time scales of the transitory phase. The consequences of introducing socio-economic  
446 drivers were considered by numerical simulations. To take into account the potential  
447 influence of socio-economic drivers, simulations were performed introducing explicitly  
448 a changing rate that depends on the relative project activity within Equation 6, affecting  
449 either states or parameters. Figure 3 shows results of simulations for just two different  
450 examples of impact taken from Table 1. The first scenario illustrated (Figure 3b, c) is  
451 for a project development that induces a change in state (a nutrient input triggering an  
452 initial increase of R, scenario 1), and the second illustration (Figure 3d, e) suggests what  
453 can occur when a project induces a change in parameters (in this case an increase in the  
454 mortality rate of primary producers and hence a decrease of their survival, scenario 2).  
455 The reactivity rate  $\rho$  was set to  $0.02$  ( $\text{time}^{-1}$ ) and the awareness rate  $\sigma$  was set to  $10^{-4}$   
456 ( $\text{time}^{-1}$ ). For both scenarios, the project activity starts at  $t = 200$  (time), the dynamics  
457 being considered at steady state before. Figure 3a shows the activity of the project

458 reaches instantaneously 1 at ‘time’ 200 when the project is implemented and then  
459 decreases smoothly as global awareness of negative impacts among stakeholders’  
460 increases [Eq. 8]. The project activity thus decreases to 0 by ‘time’ 800. This is a  
461 consequence of the relative socio-economic cost of the project reaching 1, which in our  
462 model, defines the limit of the exploitability of the project (*i.e.* when all possible time  
463 and resources are being invested in side issues).

464 In the first scenario, when R increased sharply, both P and D increased as well, but  
465 more slowly (Figure 3b). When the project activity stopped (outside the grey area, after  
466 ‘time’ 800), all states have reached an equilibrium, which is, for M, the equilibrium  
467 prior to the implementation of the project, but for P and D, a different higher  
468 equilibrium. In that sense, the outcome is similar to the outcome of the previous  
469 scenario 1. Figure 3c shows that the  $\delta$  for P and D varies differently showing the  
470 modulation by the project activity tends to alter the final amplitude of the impacts on  
471 each of the receptors.

472 In the second scenario, the configurations for the relative socio-economic cost and  
473 activity of the project are identical, but the outcomes were very different from those in  
474 scenario 2. In this case, when project activity stopped, causes for changes in the  
475 mortality rates disappeared and equilibrium states came back to the values prior to the  
476 project implementation (Figures 3 d, e). Therefore, around ‘time’ 400, the impact of the  
477 project on all receptors reaches a maximum, but all impacts relative to the baseline,  $\delta$ ,  
478 decreased and returned to zero afterward (Figure 3e).

479

480 DISCUSSION

481 The practice of EIA arose from a societal imperative to have documented expertise  
482 about potential impacts on the environment from development projects. This was the  
483 result of a legal framework created to defend environmental quality of communities and  
484 regions in the US (Cashmore, 2004; Morgan, 2012), and coinciding with a rise in  
485 visibility of ecological sciences (Supplementary Information, Figure A). Subsequently,  
486 similar requirements for environmental impact assessment were adopted by a majority  
487 of countries (Morgan, 2012). This has engendered repeated calls to develop a theory of  
488 impact assessment (Lawrence, 1997) as the practice dispersed. The need for an EIA  
489 process created a profession with a vital role in the safeguard of environmental quality,  
490 but that relies heavily on disputable methods and has an uneven record (*e.g.* Wood,  
491 2008; Wårnbäck and Hilding-Rydevik, 2009; Barker and Jones, 2013). Public pressure  
492 from stakeholders may provide some measure of accountability, however, *post hoc*  
493 analyses are rare (Lawrence, 1997) and systems can differ significantly between  
494 countries (Lyhne *et al.*, 2015). Critical review may only happen in the aftermath of a  
495 dramatic accident, such as the Macondo well blow-out in 2010 (US Chemical Safety  
496 and Hazard Investigation Board, 2016) or after management failures (Rotherham *et al.*,  
497 2011).

498 **The value of quantification.** Our study reflects on the two main scientific components  
499 of EIAs: expertise and prediction. The first is the role of the expertise. We have stressed  
500 the needs for the experts to identify receptors and to provide proper estimates of  
501 baselines. The second one is the ability of ecological theory to prediction ecosystem  
502 dynamics. We have emphasized the critical importance of the formulation of the  
503 ecosystem model to calculate correctly baselines and predict impacts. The intention of  
504 Leopold *et al.* (1971) was however far from this approach. Their approach consisted in  
505 providing a sort of template for EIA and EIS documents and to ensure a common logic

506 for how the “magnitude and importance” of the impacts identified would be presented  
507 to federal evaluators. They did not provide any details about how exactly impacts would  
508 be assessed beyond a comparison between conditions before and after the project. We  
509 therefore replaced this generic matrix approach by a quantification of system dynamics,  
510 which allows scenarios to be designed and tested.

511 **Receptor selection.** Scenarios are selection of the possible combinations that could be  
512 examined, and which are usually specific to the type of project that would be  
513 implemented. The ecosystem model is then used as a tool to help experts identifying  
514 specific receptors. Receptors can only be identified if their  $\delta$  is different from zero  
515 (either strictly positive or strictly negative). It can be identified easily in Table 1, but  
516 this is not the only condition: to be a receptor the  $\delta$  must indeed be greater (in absolute  
517 value) than the  $\delta^*$  corresponding to the limit of detection of the impact [Eq 5]; this is a  
518 statistical concept required to estimate the dispersion of the values of the receptor  
519 variables around their average. These two conditions then define what receptors are.  
520 Receptors are indeed subject to change and must be sensitive enough to be detectable  
521 with the statistical tests applied. Hence, an EIA, in contrast with a risk assessment,  
522 implies automatically a change in the receptors and aims to quantify them with a  
523 defined level of certainty and accuracy. A consequence of this is that if two receptor  
524 variables were identified as having the same dispersion, the impact will be better  
525 assessed if the averages have higher values. For example, in a marine system, the  
526 biomass of decomposers  $D$ , can be much greater than the biomass of the primary  
527 producers,  $P$  (Simon *et al.*, 1992), which means that it could be better to assess impact  
528 on  $D$ , than on  $P$ . This can be completely different for terrestrial ecosystems (Cebrian  
529 and Lartigue, 2004).



530 **Baselines and reference conditions.** In our model, the description of changes is based  
531 on the calculation of equilibrium (the baseline) and their stability, and then follows the  
532 displacement of the equilibrium values under changes in state variables, forcing  
533 variables, or parameters (Figure 3b-e). This description is a basis for clarifying our  
534 understanding of the problem. A dynamic model constrains our investigation to  
535 plausible causal relationships between the variables (receptors) and permits us to  
536 explore their contribution to the entire system. The dynamic behaviour provides a point  
537 of reference for comparisons between scenarios (as shown in Table 1 and Figure 3), or  
538 as they correspond to a specific project development. Formulating a minimum  
539 ecosystem as an example, has shown that complex behaviours can emerge with only  
540 four state variables. These results illustrate for the first time the dynamics of impact  
541 responses by receptors, revealing how complicated the evaluation of recommendations  
542 to mitigate impact may be. Furthermore, this underscores the importance of monitoring  
543 to ensure accountability over the project life cycle, including cumulative effects.

544 **Minimum ecosystems and complexity.** Models are simulation tools which aid  
545 exploration of possible outcomes and the evaluation of the simulated baseline, as well  
546 as the relevance to simulated scenarios (Tett *et al.*, 2011). Our minimum ecosystem  
547 model is essentially a representation of a perfect and autonomous bioreactor, which  
548 does not exist, nor can one be created as presented. Nutrients and detrital organic matter  
549 are 100% recyclable by one functional group of decomposers. Populations are stable  
550 indefinitely if conditions on the parameters (essentially  $p > m$ ) are respected. These  
551 conditions are not realistic, but serve the development and presentation of our approach.  
552 The proposed procedures can be applied to more complex systems, encompassing large  
553 quantity of variables (or compartments) as well as non-linear processes and hybrid  
554 dynamics, like what would be expected in more realistic representations of ecosystems.

555 However, the condition that a certain form of stability can exist in the system must be  
556 respected. It should be noted that the question of stability in ecology is part of an on-  
557 going scientific discussion recently summarized by Jacquet *et al.* (2016). This is critical  
558 to environmental impact theory because it is the presumption of stability which ensures  
559 the baseline is maintained (does not drift) during the project life cycle (Thorne and  
560 Thomas, 2008; Pearson *et al.*, 2012). In other words, an EIA is supposed to certify that  
561 what is measured as change only corresponds to an impact from the project, not external  
562 variations. Hence, monitoring takes on a new importance. For example, monitoring a  
563 non-impacted site as a reference to detect possible ecosystem drift, may be one way to  
564 assure that this condition of baseline stability is valid. This solution is conditioned itself  
565 by the necessity to have a reference site which can be characterized by exactly the same  
566 ecosystem.

567 The second basic assumption of our minimum ecosystem implies that the distribution of  
568 elements is homogeneous inside the project area. This is not always (and even rarely)  
569 the case and in aquatic systems, hydrodynamics leads to partial mixing that cannot be  
570 assimilated to complete homogeneity. Therefore, accounting for the spatial distribution  
571 structure of the elements would require the model structure be modified. For example,  
572 we can use partial differential equations or any other formulation that can treat spatial  
573 covariance. When spatial covariance is proven to exist for relevant receptors, the  
574 corresponding statistics for the test of impact must account for the spatial covariance  
575 using geostatistical methods (*e.g.* Agbayani *et al.*, 2015; Wanderer and Herle, 2015).

576 **Socio-ecological systems.** The idea that all components (*i.e.* Environmental, Social,  
577 Health ... impacts) can be inserted into a single system framework remains quite  
578 challenging. While a considerable number of propositions for conceptual frameworks  
579 and planning charts exist (Haberl *et al.*, 2009; Binder *et al.*, 2013; Bowd *et al.*, 2015;

580 Ford *et al.*, 2015) offering some insights into the complex social interactions and policy  
581 constraints involved, there is little in the way of theoretical development for impact  
582 theory. We only studied here the project activity controlled by its socio-economic cost  
583 (side costs being related to remediation and mitigation measures) as a driver of  
584 ecosystem changes. We have not, for example, considered that changes in some  
585 receptors can trigger an increase in cost and a decrease in activity. In other words, we  
586 have not considered feedbacks between the receptors and cost, because it did not appear  
587 clearly how awareness of stakeholders and reactivity of managers could be directly  
588 linked to changes in receptors (Binder *et al.*, 2013; Bowd *et al.*, 2015) for which  
589 “acceptable” remediation or mitigation measures should have already been considered  
590 during the process (Figure 2B; Drayson and Thompson, 2013). Indeed, stakeholders’  
591 awareness depends on many factors, like information or education (Zobrist *et al.*, 2009),  
592 and reactivity of managers can be constraints by many other economic and political  
593 factors (Ford *et al.*, 2015). However, the minimal model that we proposed for  
594 expressing the dynamics of the drivers of change [Equation 6] can (and should) become  
595 more rich to take into account more complete descriptions of the mechanisms that  
596 modulate awareness, activity and reactivity rates within sociological networks. We  
597 suggest that our approach could be particularly useful in the scoping step as a means to  
598 explore possible scenarios outcomes.

599

## 600 CONCLUSIONS

601 This study has linked statistical tests and mathematical modelling to assess an impact  
602 and consider some of the socio-economic drivers that mitigate it. This constitutes a first  
603 step toward an ecosystem-based approach for EIA, which needs to be proven and

604 improved. If technically, there are possibilities for EIA to rest on objective quantitative  
605 approaches, these can only be valid if the predictive capacity of the model is assured.  
606 This was, and still is, a major limitation. Furthermore, all forms of environmental  
607 impact assessment are complicated by the absence of fundamental laws in ecology  
608 (Lange, 2002) which has limited the understanding of complex objects in ecosystems.  
609 Most of the time, ecosystem models simulate dynamics with properties that are not  
610 found in realistic systems (May, 1977). We believe that to progress toward quantitative  
611 EIA it is necessary to build much closer, interdisciplinary collaborations between  
612 applied and fundamental research on ecosystems, to overcome the historical  
613 divergences. This exchange could be encouraged through concrete measures such as  
614 including funding for fundamental development within EIA as well as requiring that  
615 data collected for IA be made available in open source repositories, accessible for  
616 fundamental research.

617

#### 618 SUPPLEMENTARY MATERIAL

619 **This material is not included in this version.**

620

#### 621 ACKNOWLEDGEMENTS

622 This work was presented in part at the ICES meeting “MSEAS2016” in Brest, France  
623 and is part of the dissertation of J Coston-Guarini, inspired by the original insights of  
624 Jody Edmunds.

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800

801

802 TABLE

803 **Table 1. Summary of model outcomes for three scenarios.** Relative changes in  
804 impact are calculated in terms of mass or energy content and compared for the scenarios  
805 described in the results.

806

807 FIGURES

808 **Figure 1.** Environmental impact assessment, then and now.

809 (a) The original flow chart as it appeared in Leopold *et al.* 1971. This chart responds to  
810 a specific request by the US Department of the Interior to propose a system that would  
811 structure information in EI documents. The original figure is captioned: “Evaluating the  
812 environmental impact of an action program or proposal is a late step in a series of  
813 events which can be outlined in the following manner.”

814 (b) Example of a flow chart used by consultants today in offshore projects. Important  
815 changes include: the addition monitoring and the possibility of using modeling. Steps  
816 external to the core EIA steps are in grey. Redrawn after Edmunds *et al.* 2016.

817 **Figure 2.** The minimum ecosystem model.

818 (a) The simplest representation of a model in ecology requires two state variables at  
819 least one parameter and a ‘forcing’ variable to describe the external forcing by dynamic  
820 environmental conditions such as light, temperature, tides. State variables  
821 (compartments) are written as a function of the parameters, forcing variables, or other  
822 state variables, for a given time interval. Because these vary dynamically they are  
823 written as differential equations. Forcing variables are fixed externally, and are not

824 affected by the model calculation of the interaction represented between the two state  
825 variables (dashed line).

826 (b) The minimum ecosystem model used in this article is closed in matter but not  
827 energy, the energy source is unlimited (forcing variable) and the environment is well-  
828 mixed. Feedback interactions between the receptors (state variables) are shown using  
829 Levin's notation, where positive feedback is indicated by arrows and the negative  
830 feedback direction is shown by filled circles. Parameter values may be taken from the  
831 literature, experiments or field observations.

832 **Figure 3.** Impact as influenced by stakeholder awareness and project cost-effectiveness.

833 (a) Inverse relationship between the Project Activity and Project social cost (awareness  
834 of a negative impact among stakeholders) for the simulated scenarios. The grey shaded  
835 area is the project activity duration (between time step 200 and 800 here).

836 Behaviour of the four state variables (b, d) and the relative changes in impact (c, e)  
837 during scenario 1 and 2, respectively. These scenarios are also listed in Table 1. Filled  
838 triangles indicate in which direction the relative impacts are changing for each of the  
839 four compartments as the state variables evolve (b, d), and the unfilled triangles are  
840 placed at or near the end of the curves. All curves start at "0" in these simulations.

841

842

**Table 1. Model Outcomes.**

Relative changes in impact calculated in terms of mass or energy content for each scenario

	Nutrients	Primary Producers	Detritic Organic Matter	Decomposers
	R*	P*	M*	D*
<b>Scenario 1: project leads to R and/or M inputs to system</b>				
$R_{inp}^1$	0	+	0	+
$M_{inp}$	0	+	0	+
$R_{inp} + M_{inp}$ (CE)	0	+	0	+
<b>Scenario 2: project leads to decrease of primary producer performance</b>				
$p$ (decrease)	+	-	0	-
$m$ (increase) <sup>2</sup>	+	-	0	+
$k$ (increase)	+	-	0	-
$p+m+k$ (CE)	+	-	0	+
<b>Scenario 3: project leads to change in decomposers performance</b>				
$d$ (decrease)	0	-	+	-
$r$ (decrease)	0	-	-	+
$d+r$ (CE)	0	-	0	+
$d$ (increase)	0	+	-	+
$r$ (increase)	0	+	+	-
$d+r$ (CE)	0	+	0	-

<sup>1</sup> Simulation results shown in Figure 3b,c

<sup>2</sup> Simulation results shown in Figure 3d,e

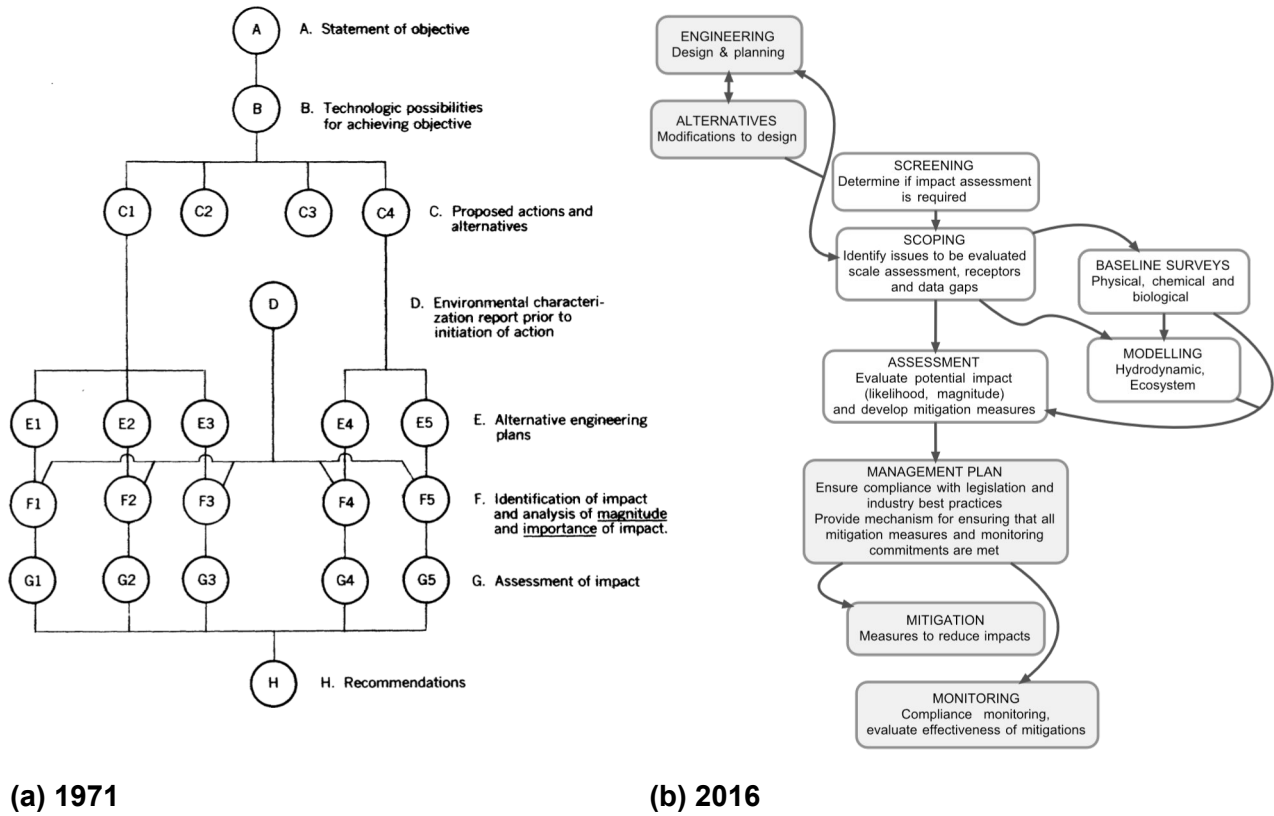
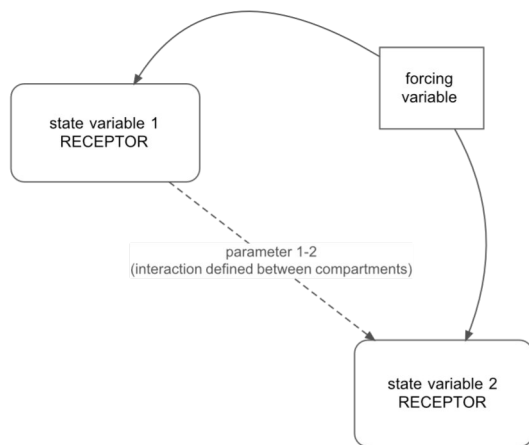


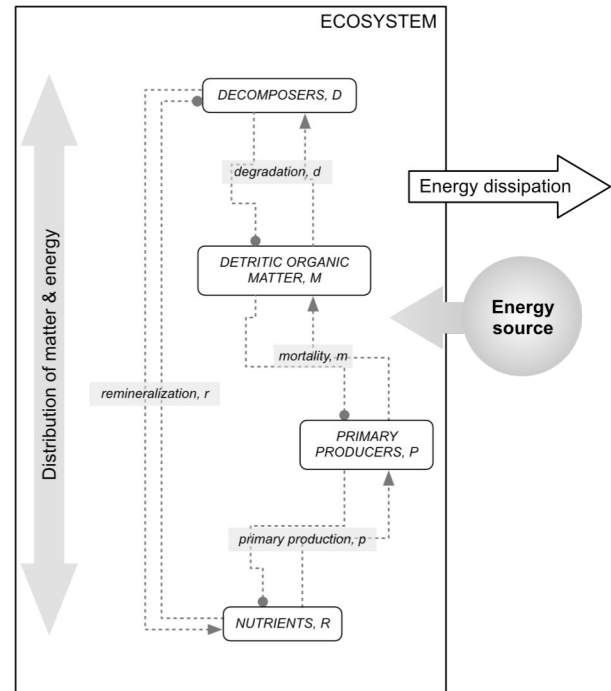
FIGURE 1.

Coston-Guarini, J. et al. "A Roadmap for a quantitative ecosystem-based environmental impact assessment"





(a)



(b)

FIGURE 2.

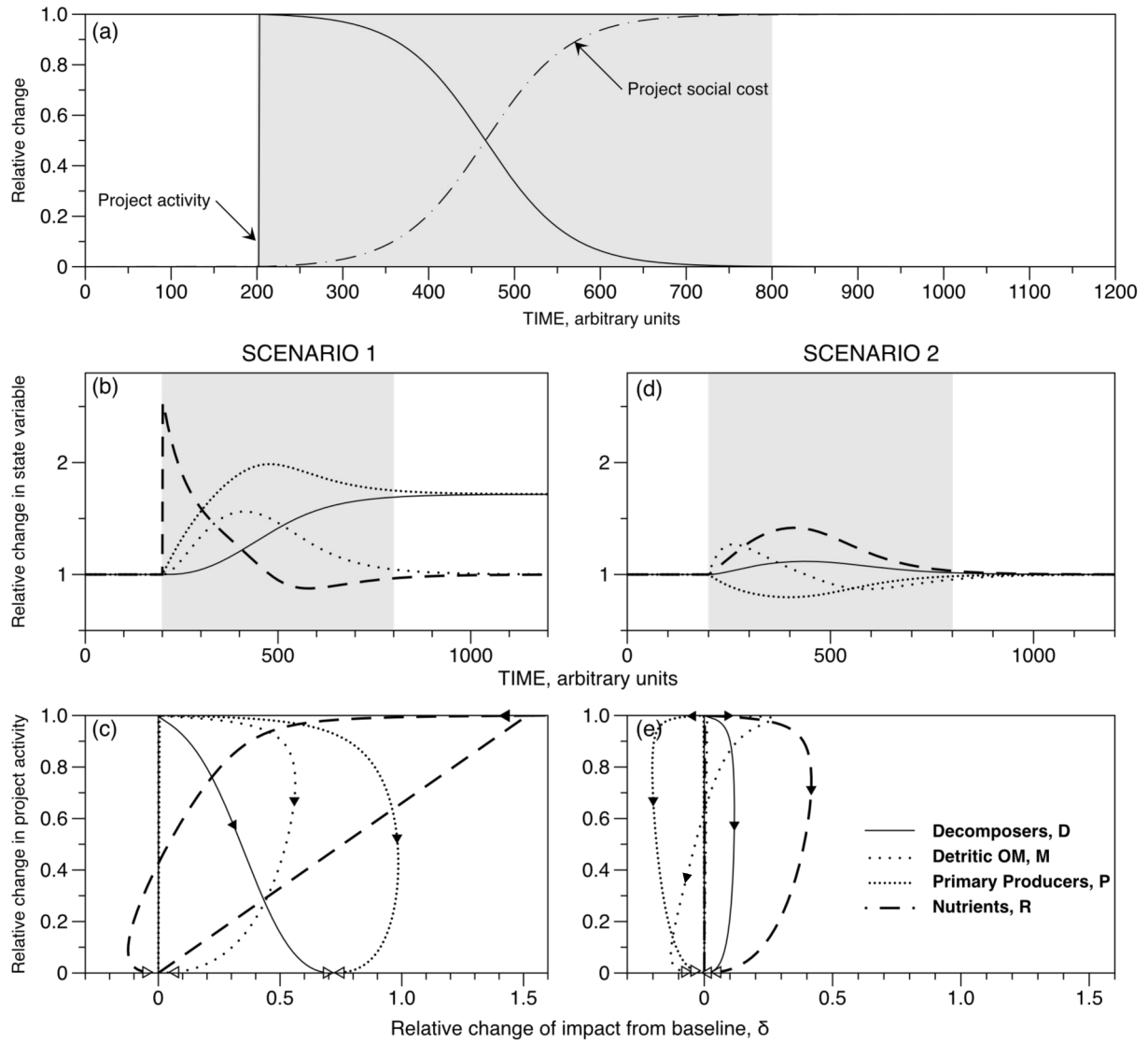


FIGURE 3.