Short communication

From a stationary to a non-stationary ecological state equation: Adding a tool for ecological monitoring

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ABSTRACT

How complex systems are able to self-organize away from equilibrium and maintain their internal functional gradients over time, by adapting themselves and changing their own environment? This is one of the most interesting questions for contemporary ecology because of its potential usefulness to assess the ecological health of our natural environment by means of ecological monitoring. This article shows how a replacement and complementation of variables, that is very simple from the mathematical point of view, can be useful to transform the state equation previously developed to describe stationary ecological conditions into a state equation for non-stationary ecological conditions. The method applied was (a) empirically tested starting from field data collected from five surveys belonging to four different kinds of taxocenosis and (b) explained in a very brief and didactic way that can be easily understandable to everybody with a standard undergraduate training in ecological studies. The main result of this article is a simple mathematical equation that can be useful to perform an instantaneous assessment of the state and trend of ecosystem development in the short run starting from a single survey, that is to say, without the availability of long time series of data that allows the conventional studies of comparative ecology in order to assess the course of ecological succession. This proposal adds an innovative diagnostic tool empirically useful for ecological monitoring.

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1. Introduction

Getting reliable information about the ecosystem *state* and its *trends* is crucial to keep a healthy environment, as well as to handle the economic exploitation of natural resources in a sustainable way. Ecological monitoring (EM) generates a large part of the information required to respond to these challenges. EM has repercussions in three main ambits, at least: (a) at the applied scale, because EM allows a better assessment and management of our socio-economic impact on nature; (b) at the pure informative scale, because scientists are able to get new basic knowledge about nature by gathering the appropriate data to assess the ecosystem status (i.e., surveillance: the systematic measurement of variables and processes at

regular intervals over time in order to establish a series of comparable data, see Spellerberg, 2005); and (c) at the theoretical adjustment scale, because EM in itself is a fruitful scenario to obtain an essential degree of adjustment between theoretical and empirical ecology which is suitable to maintain a healthy state in every science (*"a theory which cannot be confronted with the facts or which has not been verified quantitatively by observed data, is, in fact, devoid of any scientific value"*; Allais, 1990, p. 5).

Although these three scales are connected with each other in an indivisible whole, the latter of them is a sort of "epistemological glue" that confers, or should confer, an all-encompassing coherence in ecology as a science. For example, according to Lindenmayer and Likens (2010, p. 1322), an appropriate conceptual model for EM becomes a focal point for discussions among partners about how an ecosystem might be managed and monitored. If the conceptual model fails, the respective program of EM fails as a whole also, either by excess (*"snowed by a blizzard of ecological details"*, see

Lindenmayer and Likens, 2010, Section 5.3) or by defect of information, or because there is a lack of coherence between information gathered and EM's goals. Correspondingly, given the limited conservation funding and the almost unlimited conservation needs, an additional key point in connection with item (c) is that every program of EM should be useful to perform a selective comparison aimed to discriminate among competing conceptual models in order to enhance the general efficiency level of ecological research (see Nichols and Williams, 2006).

Unfortunately, the general state of conventional ecology does not seem to have been in favor of the above-mentioned points in regard to item (c) given that, during the last four decades ecology (i) has shown a significant degree of theoretical turmoil and contingency (e.g., Watt, 1971, p. 569; Lewin, 1983; Lawton, 1999, p. 178; Woodward, 2001; Simberloff, 2004; Roughgarden, 2009; Palma, 2010); (ii) it has been affected by a chronic lack of appreciation of past literature and theoretical fickleness (see Belovsky et al., 2004); (iii) and the linkage between theoretical and empirical ecology seems to have been systematically, although unwittingly, weakened (see, e.g., Belovsky et al., 2004; Loreau, 2010; Angilletta and Sears, 2011; Joseph et al., 2013; Scheiner, 2013). The main negative effect of this situation in regard to the usefulness of EM has been implicitly summarized in the following way: *"today, the needs of conservation biology for metrics of 'ecological health' that can be applied across types of communities prompts a renewed interest in the possibility of general theory for community ecology"* (Roughgarden, 2009, p. 521). This statement exactly reflects the main problem that gives rise to this paper.

Given the above-mentioned problem (i.e., to set up the elementary principles for a basic metric of ecological health valid for a wide spectrum of ecosystems), the main goal of this article is aimed to answer the following set of interrelated questions in order to add a diagnostic tool to EM: (1) is it possible to know in what a measure an ecosystem is stable, or not, by means of a single survey? Assuming that a given ecosystem is under non-stationary ecological conditions (non-SEC); (2) is it possible to assess the direction of ecological change (either in favor or against the spontaneous pro-diversity trend of ecological succession) in the short run by means of a single survey? Assuming positive responses to these previous questions; (3) is it possible to express such a metric of "ecological health" under non-SEC by means of an obviously simple mathematical formulation in agreement with the ecological state equation (ESE, see Eq. (1) in Table 1, cell A3) previously developed by Rodríguez et al. (2012) under stationary ecological conditions (SEC)? In general, these questions are related to a more general question: How complex systems are able to self-organize away from equilibrium and maintain their internal functional gradients over time, by adapting themselves and changing their own environment?

As commented above, the first section of this manuscript describes a scientific problem of significant importance for conservation and sustainability, and establishes our goals in order to contribute to an operative solution in this regard. The second section is aimed to assess the state of the ecosystem (SEC vs. non-SEC), in order to respond to question number (1). This response was implicitly proposed before by Rodríguez et al. (2012, 2013), but it needs to be retaken in this context with the goal of supporting a chain of reasoning connected with later publications that expand its meaning. The third section is aimed to answer to questions (2) and (3), being the core of this article. Finally, the fourth section of the article performs a straightforward exploration of the meaning of these results for ecology, in general, as well as for EM, in particular.

Table 1
Summary of the main indicators connected with the alternative states of ecosystems.[a]

Raw	Criterion of comparison	Column (alternative states of the ecosystem)	
		A SEC	B Non-SEC
1	B–D_{TO-H}: biomass (m_{ep} or m_{eTp}) ↔ dispersal (I_e) trade-off in function of H_p values	*Yes*; significant *negative correlation* H_p, I_e; and significant *positive correlation* H_p, m_{ep} or H_p, m_{eTp} (Rodríguez et al., 2013). So $m_{ep\ min.}$ ↔ $H_{p\ min.}$; $I_e^2{}_{min.}$ ↔ $H_{p\ max.}$; $(m_e \cdot I_e^2)_{max.}$ ↔ intermediate values of H_p, close to the peak of H_p distribution (see Fig. 2a)	*No*; statistically *non-significant negative correlation* H_p, I_e; and statistically significant *positive correlation* H_p, m_{ep}
2	Ecological equivalent (k_e) of Boltzmann constant (k_B)	*Yes*; *non–significant* difference between the mantissa of observed value ($k_{e(o)}$) and the mantissa of k_B. k_e emerges from B–D_{TO-H} given that k_e is the average value of the product $I_e^2 \times H_p \times m_{ep}$	*No*; *Significant* difference between the mantissa of observed value ($k_{e(o)}$) and the mantissa of expected value ($k_{e(e)} \approx 1.3806504E\varphi\ J_e \cdot$nat/individual). $k_{e(o)} \neq k_{e(e)}$ precisely because of B–D_{TO-H} is broken
3	ESE: ecological state equation	$2E_{eTp} = N_p k_{e(e)}/H_p$; (Eq. (1); see explanations in the footnote, as well as in Rodríguez et al., 2012)	$2E_{eTp} \neq N_p k_{e(e)}/H_p$ (see examples starting from four surveys under non-SEC in Fig. 1a)
4	Adjustment of H_p values to gamma distribution	Statistically *non-significant* difference either with standard gamma distribution or with Eq. (2). See, e.g., Fig. 2a	Statistically *non-significant* difference with standard gamma distribution, but statistically *significant* difference with Eq. (2). See, e.g., Fig. 2b
5	Ecological inertia and elasticity	*Undetectable* because the system is not moving through the species diversity gradient because of the influence of B–D_{TO-H}	*Detectable* at the intra-survey scale. See Fig. 2b,c in comparison with each other and with Fig. 2a. Additional comments below (Section 3)
6	h_e^{ev}: eco-evolutionary equivalent of Planck's constant	$h_e^{ev} = 6.62606957E - 1$ Je per unit of ∇k_e). *Undetectable* at the intra-survey scale	*Detectable* at the inter-survey scale as the regression coefficient of E_e vs. $k_{e(e)}$; intercept = 0. So the regression equation: $E_e = k_{e(e)} \times h_e^{ev}$. i.e., h_e^{ev} indicates the non-continuous variation of E_e per each unit of $\nabla k_e = 1$ Je·nat/individual (Rodríguez et al., 2015a)

[a] SEC: stationary ecological conditions. m_{ep}: mean value of biomass per individual per plot. m_{eTp}: total biomass per plot. I_e: indicator of dispersal activity (see Eq. (1A), Appendix A). H_p: species diversity per plot $= -\Sigma[(n_i/N)\cdot\ln(n_i/N)]$, where n_i: number of individuals of species i, $N = \Sigma n_i$, and ln: natural logarithm (see Shannon, 1948; Magurran, 2004). $k_{e(o)} = I_e^2 \times H_p \times m_{ep}$. $k_{e(o)}$ becomes $k_{e(e)}$ (i.e., the typical expected value for a given kind of taxocenosis) if the mantissa of its mean value in regard to the set of plots of the survey as a whole (excluding plots in which $H_p = 0$, these cases indicate that the plot size should be larger) is not significantly different from the mantissa of physical Boltzmann constant (k_B). $k_B = 1.3806504E\text{-}23$ J/K per molecule. ESE: ecological state equation (Eq. (1)). E_{eTp}: total eco-kinetic energy per plot $= N_p \times E_e$. N_p: total number of individuals per plot. E_e: mean eco-kinetic energy per individual per plot $_{(p)}$ or survey $_{(s)}$ expressed in ecoJoules (J_e; see Appendix A, Section 2) as a proxy for trophic energy $= \frac{1}{2}m_{ep}I_e^2$; in a similar way in which the kinetic energy in physics is $E = \frac{1}{2}m \cdot v^2$; where m: mass and v: velocity. h_e^{ev}: equivalent of Planck's constant at the evolutionary (ev) scale. $\varphi = -x_i, \ldots -3, -2, -1, 0, +1, +2, +3, \ldots, +x_i$, with a typical recurrent value depending on the type of taxocenosis studied.