especially clear for journals published in English-speaking countries (Figure 1b). This means that the science produced by ecologists who are NoNES is adequate enough to raise the average citation rates of journals, or is at least not so deficient as to harm their scientific prestige.

The language bias seems much stronger than any other possible source of inequality among ecologists (eg Primack *et al.* 2009). I have argued elsewhere (Clavero 2010) that the unequal competition for publication between scientists – NES and NoNES – is a clear example of linguistic injustice. In my opinion, native-English-speaking editors and scientists should be aware of their advantageous position and assist non-native English-speaking writers with their English writing, while native-English-speaking publishers should provide English editing services to non-native English-speaking ecologists who submit quality science.

Regardless of how it is accomplished, clearly these issues must be addressed in order to attain greater fairness in scientific publication (Clavero 2010). Moreover, besides ethical concerns, there are other reasons to facilitate the access of non-native English-speaking authors to international scientific publications. For example, an unknown amount of ecological research, undoubtedly including some important findings, may in effect be lost to a wider international audience when it is not published in English, mainly as a result of the language limitations of non-native English-speaking ecologists (eg Waal 2003). This loss of information may hinder the development of meaningful meta-analyses (eg Dunn *et al.* 2007). Finally, by studying Figure 1b, editors wanting to raise the IF of their respective journals may even find more practical reasons to promote works by NoNES.

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Revisiting climate thresholds and ecosystem collapse

Policy dialogues regarding climate-change mitigation require the setting of "safe" targets for atmospheric carbon dioxide (CO_2) concentrations. These targets are influenced by analyses of the response of ecosystems and society to rising levels of greenhouse gases. Much of the science has focused on the existence of threshold levels of CO_2 beyond which systems will gather momentum toward a deleterious state (Lenton *et al.* 2008). Policy makers are familiar with thresholds, but confusion exists over the consequences of exceeding these limits. This is partly because some systems respond to multiple types of threshold, and we lack a nomenclature to distinguish their nature and impacts. Without such structure, there is a risk that policy makers will fail to discriminate between the consequences of exceeding different types of threshold. Doing so could lead to grossly misplaced policies.

The problem can be illustrated by looking at coral reefs. "Threshold" literally means the level beyond which a new event occurs and although reefs face several types of threshold, not all imply ecosystem collapse (Figure 1). First, there is a *degradation threshold* beyond which the average state of an ecosystem begins to degrade. A recent report called for a reduction of CO_2 levels from their present level of 387 parts per million (ppm) to < 350 ppm in order to return the environment to the conditions to which corals are adapted (Veron *et al.* 2009). The rationale is simple, in that once atmospheric CO_2 exceeded ~340 ppm in the early 1980s we began to witness large-scale coral bleaching, caused by rising sea temperatures and greatly exacerbated by climatic oscillations such as the El Niño–Southern Oscillation. Three hundred and fifty ppm is a threshold beyond which the average state of a reef ecosystem becomes degraded, but this does not imply ecosystem collapse.

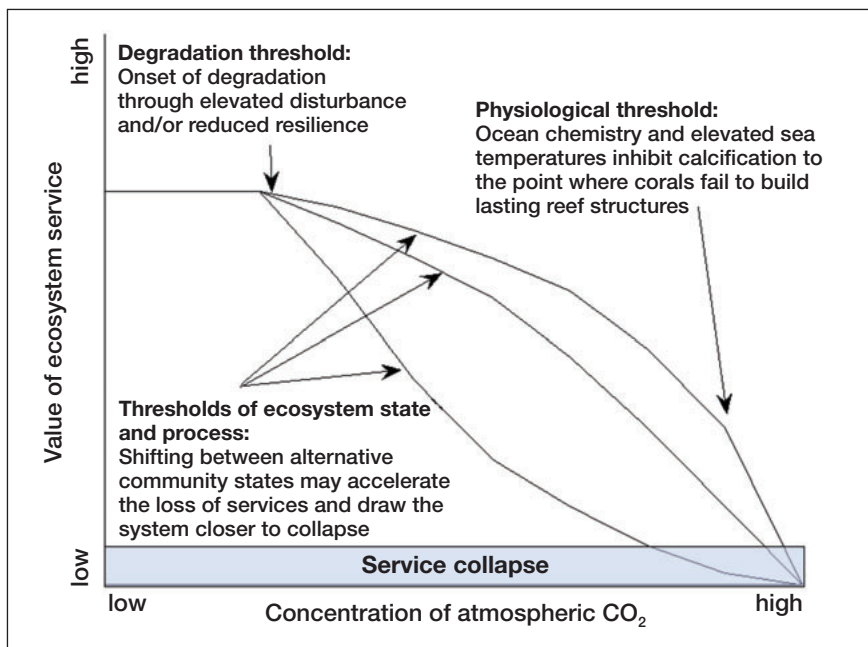Second, many ecosystems, including coral reefs, exhibit phase shifts and

Figure 1. Multiple thresholds acting on the ecosystem services of coral reefs.

even multiple stable community states, if resilience is lacking. For example, heavy depletion of herbivores on some Caribbean reefs reduced the ability of coral communities to recover after disturbance and resulted in a switch of community state to one dominated by seaweed. Dynamics of this type are usually driven by ecological feedbacks and are a major focus of ecological science. In this case, *thresholds of ecosystem state and process* might be defined as sets of conditions that determine whether the ecosystem will exhibit natural recovery or shift inexorably toward a different and less desirable state (Mumby *et al.* 2007).

Yet a third type of threshold can occur at even higher levels of CO_2 (Figure 1). Here, the environment becomes so hostile that fundamental physiological functions are grossly impaired. Coral reefs provide an excellent example of such a *physiological threshold* because of their vulnerability to rising levels of CO_2 (Hoegh-Guldberg *et al.* 2007). There remains great uncertainty over precisely when and where such physiological functions effectively cease. Rising sea temperatures stress corals and reduce their growth, and extreme events cause mass mortality. Ocean acidification may depress growth and other functions further, but the degree to which corals are

negatively affected is highly uncertain.

It is conceptually useful to distinguish different types of threshold because the processes involved are different, their onset may occur sequentially, and it forces us to distinguish ecosystem degradation from collapse (Figure 1). System "collapse" depends on the particular ecosystem service or attribute being considered and will vary from one function to another. For example, collapse of the ability of corals to form accreting reef structures may arise through persistent algal phase shifts (ecosystem state thresholds), physiological stress leading to low calcification rates exacerbated by erosion (physiological thresholds), or a combination of both. Modeling the interactions among thresholds requires a multidisciplinary approach and remains a considerable challenge.

Failure to recognize that not all threshold trangressions imply ecosystem collapse could have profound consequences for global poverty and human health. Within several months of a widely publicized report being published that called for long-term CO_2 levels of 350 ppm (Veron *et al.* 2009), we observed identical responses by a senior government legislator to the UN Framework Convention on Climate Change (COP 15) and a senior representative of the UN

Convention on Biodiversity who questioned the value of conservation efforts for coral reefs based on a misunderstanding of thresholds. Both observed that since the threshold of 350 ppm CO_2 for coral reefs has already been exceeded and may be an infeasible target, why should investments in reef management continue? Both representatives had implicitly assumed that exceeding the 350-ppm threshold meant inevitable ecosystem collapse (even though this was not stated in the paper), such that further conservation investments would yield negligible benefits. This assumption was incorrect because coral reefs did not collapse once 350 ppm was exceeded and they still support vital ecosystem services, including coastal defense and food security, albeit at potentially reduced levels. Indeed, investments in reef management are more important than ever in order to minimize the loss of services and stem the rise of poverty in coastal communities dependent on reefs. A more transparent means of communicating the consequences of crossing thresholds is needed to help busy policy makers use scientific advice effectively.

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Dams and afforestation plans in Chilean Patagonia

Two large hydropower projects (involving seven dams in total) in Chilean Patagonia will, upon acceptance of their environmental impact studies (EISs), lead to the inundation and clear-cutting of almost 12 000 ha of natural forest. The EISs presented by the hydropower companies acknowledge that vegetation (including forest cover) will be heavily impacted by each project. A companion project will also include construction of a 100-m wide, 2300-km-long power-line corridor – resulting in further clear-cutting of forest. To fulfill Chilean environmental legislation, the companies behind such projects must compensate for the area of removed forest by afforesting a similar area of land with tree species of the same type as those that have been removed. The companies claim that their afforestation plans will be highly successful, but we believe the plans associated with these projects contain important flaws that undermine their viability.

First, seedling establishment is the major bottleneck in any afforestation process. Although a precise knowledge of species' ecological requirements is recognized as essential for establishment success, the requirements of native Patagonian forest species are largely unknown. Moreover, the submitted plans that propose afforestation of Patagonian sites – where growing seasons are typically short and temperatures cold – follow tree plantation protocols developed for use in forests located northern to Patagonia, which are subject to a cli-

Figure 1. *(a) Evergreen rainforest and (b) steppe ecosystems in Chilean Patagonia. Hydropower companies plan to compensate for inundation and clear-cutting of natural forest by afforesting the Patagonian steppe with rainforest tree species.*

mate with a Mediterranean influence (Luebert and Pliscoff 2006).

Second, these protocols, which have demonstrated success with fast-growing, exotic tree species (eg *Pinus radiata*, *Eucalyptus globulus*), require the elimination of competing vegetation within plantations – thereby ignoring the importance of ecological processes such as facilitation (ie positive plant interactions). Indeed, it is the presence of other plant species (facilitation) rather than their absence that determines whether slow-growing species will establish successfully at less productive sites with relatively harsher environmental conditions (Callaway 2007). In this regard, there are alternatives to traditional tree-planting protocols that may better ensure successful afforestation. For example, survival of *Nothofagus pumilio* – a widespread Patagonian tree species – can be greatly increased if seedlings are planted in groups (Fajardo and McIntire 2011) instead of in isolation, as dictated by protocols.

Third, some of the sites that the companies have already acquired for afforestation are located within the steppe belt of Patagonia, where few trees occur naturally and where selected surface conditions are quite different from those of the evergreen rainforest that will be affected by the dams' construction. Thus, attempting to afforest the Patagonian steppe (<500 mm annual precipitation) with species belonging to the evergreen rainforest (>2000 mm annual precipitation; Figure 1) seems likely to fail because of the mismatch

between regional conditions and the physiological requirements of the associated species.

Finally, we believe that these afforestation plans are based on a "trial and error" approach, which is an inefficient use of time and money.

It has been suggested that large dams result in mostly negative impacts on ecosystems, and that early cooperation between ecologists, dam engineers, and affected people is essential to improve the effectiveness of compensatory measures (WCD 2000). Unfortunately, such collaboration was not considered when the afforestation plans for the Patagonian hydropower projects were being drawn up. If the current proposals are implemented as intended, another failure in environmental mitigation may be added to the list.

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