Environmental Pollution 200 (2015) 85–92

Contents lists available at ScienceDirect

Environmental Pollution

journal homepage: www.elsevier.com/locate/envpol

Combined ecological risks of nitrogen and phosphorus in European freshwaters

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ARTICLE INFO

Article history:
Received 26 September 2014
Received in revised form
29 January 2015
Accepted 9 February 2015
Available online 18 February 2015

Keywords:
Ecological risk
Nitrogen
Phosphorus
Lake
Stream
River basin

ABSTRACT

Eutrophication is a key water quality issue triggered by increasing nitrogen (N) and phosphorus (P) levels and potentially posing risks to freshwater biota. We predicted the probability that an invertebrate species within a community assemblage becomes absent due to nutrient stress as the ecological risk (ER) for European lakes and streams subjected to N and P pollution from 1985 to 2011. The ER was calculated as a function of species-specific tolerances to NO_3^- and total P concentrations and water quality monitoring data. Lake and stream ER averaged 50% in the last monitored year (i.e. 2011) and we observed a decrease by 22% and 38% in lake and stream ER (respectively) of river basins since 1985. Additionally, the ER from N stress surpassed that of P in both freshwater systems. The ER can be applied to identify river basins most subjected to eutrophication risks and the main drivers of impacts.

1. Introduction

The “limiting nutrient” concept, following Liebig's Law of the Minimum, was based on the effects of added nutrients on crop performance (van der Ploeg et al., 1999). Later, the concept was extended to productivity-based experiments for eutrophication, such as those testing the effects of nutrient surplus (mainly nitrogen – N – and phosphorus – P) on chlorophyll concentration or biomass productivity (Allgeier et al., 2011; Elser et al., 2007). Despite the benefit prompted by the increase in the availability of a resource, such as increase in productivity, a further increase in the same resource level could cause ecosystem damage, such as a shift in species composition (Odum et al., 1979).

Freshwater eutrophication is triggered by agricultural and urban discharges of N and P as well as atmospheric emissions of N (Harrison et al., 2010; van Drecht et al., 2005; van Drecht et al., 2009). On one hand, the increase in nutrient availability generally increases primary production and, thus, the availability of food to planktivores and herbivores (Carpenter et al., 1985). On the other hand, it may also lead to increased predation by secondary consumers and decreases in food quality (Carpenter et al., 1985; Grimm and Fisher, 1989), water transparency and light availability, thereby eliciting competitive exclusion of autotrophic species and the release of allelochemicals by competing phytoplankton, particularly cyanobacteria (Havens et al., 2001, 2003; Leflaive and Ten-Hage, 2007). Furthermore, enhanced decomposition of nuisance algae and macrophytes may generate hypoxic or (in extreme cases) anoxic conditions in aquatic systems (Carpenter et al., 1998). Ultimately, oxygen depleted conditions, the exposure to toxins released by phytoplankton, and shifts in food availability may be harmful to invertebrates (Camargo and Alonso, 2006; Correll, 1998) (Fig. 1). Therefore, the same nutrient stimulating autotrophic productivity and food availability may, in turn, instigate ecosystem damage at increasing concentrations. Accordingly, defining the nutrient as a resource or as a stressor depends as to whether its concentration prompts a benefit or damage to ecosystems.

Ecological theory models detect this dual aspect of N and P. The intermediate disturbance hypothesis (IDH) conveys that species richness is maximized at intermediate levels of stress and

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http://dx.doi.org/10.1016/j.envpol.2015.02.011
0269-7491/

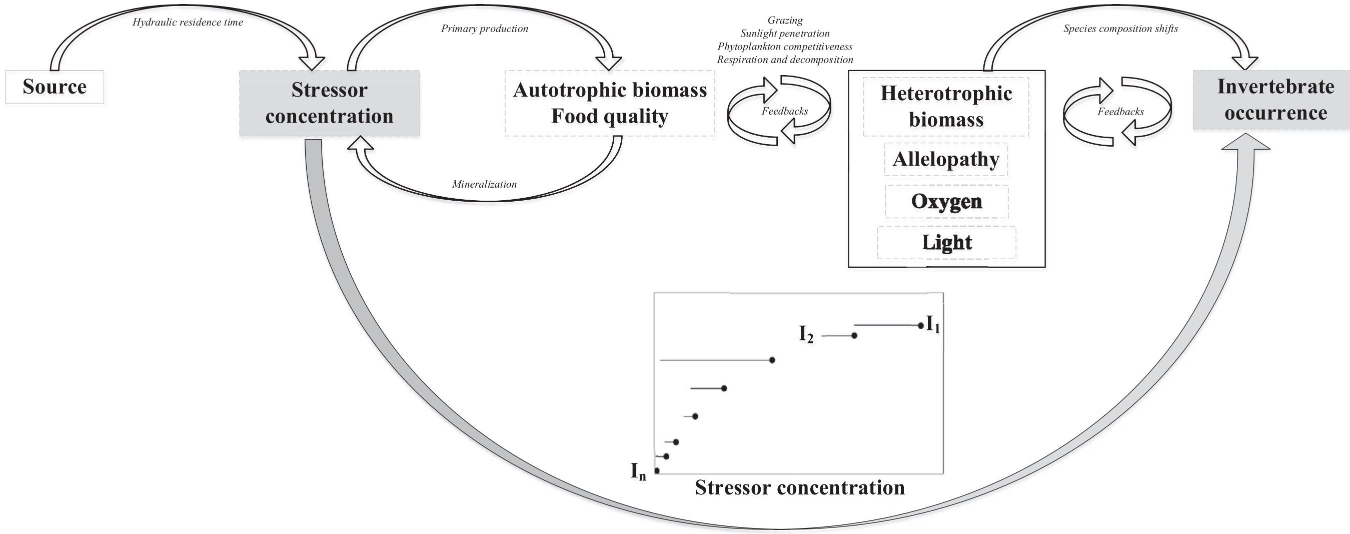

Fig. 1. (2-column figure image) Probabilistic model of our study: the linkage between stressor concentrations and invertebrate species occurrences (grey fill). The full impact pathway of freshwater eutrophication includes the discharge of stressors (nitrogen and phosphorus) into freshwater bodies, thereby influencing N and P levels, primary productivity, and autotrophic biomass which, in turn, may set off various ecological feedbacks, including the occurrence of invertebrates in the field. The graph illustrates the upper boundary of the stressor concentration (x-axis) at which individual invertebrate species occurs under field conditions (y-axis, listed as I_1 to I_n, ordered in descending tolerance to a stressor).

minimized at the two extremes (Grime, 1973). Underlying the IDH, the physiological tolerance hypothesis (Currie et al., 2004) conveys that species richness is the upshot of the tolerance of each individual species to specific local conditions. Currie et al. (2004) use the hypothesis to explain species tolerance to climatic variables and we expand it so as to describe species tolerance to the upper end of nutrient levels, i.e. the level of the stressor which triggers species loss.

Eutrophication is a complex issue as it encompasses potential feedback mechanisms (van Donk and van de Bund, 2002), non-linear responses of primary production to trophic conditions (Genkai-Kato and Carpenter, 2005), and synergistic effects of N and P on primary production (Elser et al., 2007). The extent to which they drive primary productivity can be examined by analyzing past nutrient level patterns (Anderson, 1998) or nutrient stoichiometry changes (Glibert, 2012), ecological modeling (Genkai-Kato and Carpenter, 2005), or via nutrient addition experiments (Schindler, 1977). Nonetheless, the development and the application of eutrophication models which include all the various pathways through which N and P influence individual invertebrate species occurrence may be troublesome due to lack of data and of insights on all relevant mechanisms of impact.

Alternatively to mechanistic models, statistical models coupled with available monitoring data of water bodies may be used to underpin biodiversity effects of eutrophication and provide environmental protection agencies with guidelines for the improvement and the maintenance of water quality (Smith et al., 2007). We circumvent the uncertainties within each of the different ecological mechanisms by developing a probabilistic model of invertebrate species occurrences with the upper observed stressor tolerance in field observations (Fig. 1).

Eutrophication indicators based on the performance of invertebrates may be less certain than those on autotrophs since consumers are not directly affected by N and P concentrations as are photosynthesizing organisms (Johnson et al., 2014). However, invertebrates are convenient to environmental agencies because they are extensively monitored (Growns et al., 1997) and their monitoring can be easily employed as water quality indicators, such as the ecological quality ratio (EQR). In the case of the EQR, the composition of invertebrates is compared with a reference representing minimum impairment (Clarke, 2013). Nevertheless, indicators usually do not detect the main stressor driving the eutrophication impact.

In the case of eutrophication, the estimation of the overall health quality of freshwater needs also to uncover what the main cause of impairment is. Therefore, an ecological indicator that allows for estimation of the ecosystem health as well as for identification of the driving stressor of eutrophication impairment may provide environmental agencies with the tools to recognize impaired areas and to target the stressor of concern. In this study, we propose the ecological risk (ER) to identify the areas and the main drivers of eutrophication impairment. This framework is compatible with risk assessments proposed for toxicants (Beketov et al., 2013; Fedorenkova et al., 2012; Malaj et al., 2014; van Straalen, 2002).

2. Material and methods

2.1. Ecological risk

The ER posed to a group of species depends upon the sensitivity of each of its species and the probability that the group of species is subjected to the stressor (Fig. 2a). Thus, ER (dimensionless) is the definite integral

$$ER_i = \int_{-\infty}^{\infty} CDF_i(x) \cdot PDF_i(x) dx \tag{1}$$

where PDF is the probability density function of the stressor i of ^{10}log concentration x and CDF is the cumulative distribution function of the sensitivity of species to increasing x (Fedorenkova et al., 2012; van Straalen, 2002) (see study outline in Fig. 3). The ER can be interpreted as the probability that an invertebrate species within a