

# A new method for jointly assessing effects of climate change and nitrogen deposition on habitats

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## ARTICLE INFO

### Keywords:

Climate change  
Habitats  
N deposition  
Projections  
Risk assessment  
Species distribution models  
Synergistic effects

## 1. Introduction

Biodiversity is under pressure globally from multiple drivers including land use change, overexploitation, climate change, atmospheric nitrogen (N) deposition, and other environmental stressors (Ceballos et al., 2015; Sala et al., 2000). Concern about the decline of biodiversity under these multiple threats and the consequences for ecosystem functioning and services has motivated considerable research effort (e.g. Barnosky et al., 2011; Bellard et al., 2012; Estes et al., 2011; Tittensor et al., 2014) as well as internationally coordinated policy response (e.g. Convention on Biological Diversity, Intergovernmental Platform for Biodiversity & Ecosystem Services). Scenarios and projections of how biodiversity may change under plausible future pathways of drivers are key to proactive environmental policies and management (e.g. Guiot and Cramer, 2016; Pereira et al., 2010; Thuiller et al., 2013). A shortcoming of most of these projections is, however, that they exclusively consider one particular driver and neglect the simultaneous and possibly interacting effects of others (e.g. Pereira et al., 2010; Titeux et al., 2016).

Climate change and N deposition represent a pair of drivers that is

known to have separate and interactive effects on biodiversity and ecosystems (Bernhardt-Römermann et al., 2015; Greaver et al., 2016; Porter et al., 2013). Adverse effects of N deposition mainly stem from eutrophication, which fosters the growth of opportunistic plant species and, eventually, the exclusion of less competitive ones (Bobbink et al., 2010a, 2010b; Gilliam, 2006; Hautier et al., 2009; McClean et al., 2011), and from acidification, which leads to cation imbalances, associated physiological stresses and loss of sensitive plant species from communities (Roem et al., 2002; Simkin et al., 2016a; Stevens et al., 2010). Both of these effects can propagate through the food web and alter the composition and diversity of heterotrophic groups (de Sassi et al., 2012; Wallisdevries and Van Swaay, 2006). Climate change may modify N supply to biota by influencing atmospheric N deposition through the amount and temporal pattern of precipitation, which in turn leads to modifications in soil chemical and microbial processes. Moreover, temperature and moisture conditions control the availability of soil N for plants via their effects on microbial transformation rates of reactive N (Butler et al., 2012; Guntiñas et al., 2012). Interactions are complex and only partly understood (Greaver et al., 2016), but both empirical observations and modelling studies indicate that a warmer

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and wetter climate may enhance detrimental effects of N deposition on biodiversity (Porter et al., 2013; Zavaleta et al., 2003) while a drier climate may render plant communities less sensitive to N effects because of reduced net N mineralization as well as reduced biological activity (Bobbink et al., 2010b; Simkin et al., 2016a).

Despite these potential interactions, the effects of climate change and N deposition have only recently begun to be considered together in large-scale assessments of their effect on biodiversity (De Vries et al., 2010). We assume that methodological issues have hindered such joint assessments. At larger spatial (and temporal) scales, the possible impact of climate change on biodiversity is mainly evaluated by means of species or habitat distribution models (SDMs) (e.g. Guisan and Thuiller, 2005; Thuiller et al., 2005). Spatial variation in N availability to biota cannot be easily mapped and, hence, is just emerging to be included into such models (Rowe et al., 2015). Instead, evaluations of biodiversity risk from N deposition for larger areas have mainly been based on the critical load (CL) approach, i.e. the definition of system or habitat specific thresholds beyond which negative effects on biodiversity, among other system attributes, are to be expected in the long run (Nilsson, 1988). This approach has, by contrast, not been applied in climate impact research and ‘climatic critical loads’ of habitats are hence not defined so far.

As a step forward to bridging the gap between these different metrics, we present a method to express future threats to biodiversity from N deposition and climate change on a common scale. We therefore adapt the CL approach by defining critical ‘climatic loads’, or – more accurately – climatic thresholds of ecosystems. We derive these climatic thresholds from SDMs of the species that typically occur in these ecosystems. The risk of an ecosystem from either climate change or nitrogen deposition can then be compared in terms of exceedance of the CL for N and the climatic threshold, respectively, and a combined risk from the exceedance of both thresholds can be calculated. We emphasize that the aim of our approach is not to improve mechanistic understanding of the interactive effects of climate and N deposition on biodiversity. Rather, we want to provide a consistent screening procedure to identify and compare areas and habitats under risk from both of these components of global environmental change. We illustrate the method by an application to the natural and semi-natural habitat types of Austria.

## 2. Material and methods

### 2.1. Conceptual approach

The exceedance of system-specific CL of airborne pollutants has been quantified by critical load functions which provide distances between measured deposition values and potentially interdependent effect thresholds in one or more dimensions (Posch et al., 2001). These thresholds, i.e. the CLs of habitat types, are derived by syntheses of experimental and observational studies (e.g. Bobbink et al., 2010a, 2010b). Whether and how much deposition exceeds the CL can be mapped in geographical space by combining ecosystem or habitat maps with deposition maps (e.g. Henry and Aherne, 2014; Posch et al., 2015). Here, we transfer this approach to climate impact evaluation by defining analogous ‘critical loads of climate’ for habitat types, henceforth called climatic thresholds. We then overlay both climatic maps and N-deposition maps with habitat maps to produce a map of combined climatic and N deposition exceedance.

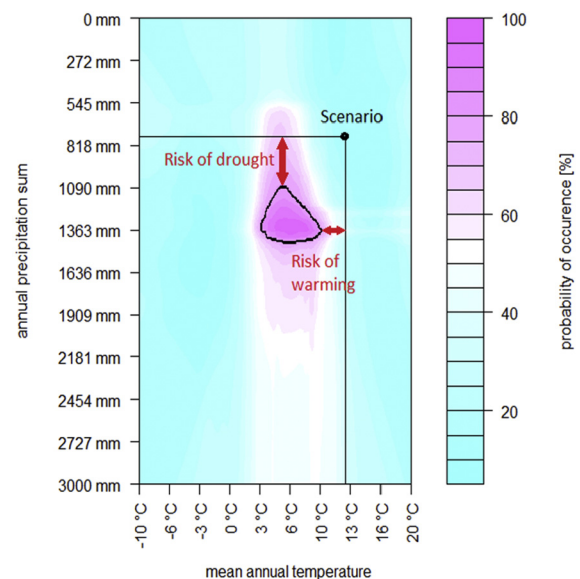
Our definition of climatic thresholds for particular habitat types is based on the idea that such habitat types can be characterized by the co-occurrence of a set of ‘characteristic’ species which are special to or especially abundant under the environmental conditions typical for these habitats. This idea underlies the phytosociological approach to habitat classification (Dengler et al., 2008) which is, in turn, the basis of the conservation-oriented legislation and administration in the European Union (e.g. Habitat Directive, Directive 92/43/EEC; Rodwell

et al., 2002). The close relationship between the typical environmental conditions and the characteristic species of a habitat type (Willner et al., 2009) implies that the climatic niche of a habitat type can be derived from the climatic requirements of its characteristic species.

SDMs have been specifically developed for quantifying species’ realized niches from geographical distribution patterns (Franklin, 2010). The predictions of an SDM relate species’ occurrence probability to one or more ecological gradients. Here, we use this modelling technique for defining a climatic threshold of a particular habitat type by (1) projecting occurrence probabilities of all of the characteristic species of this habitat type into a two-dimensional (mean annual temperature, annual precipitation sum) climatic space; (2) calculating, from these projections, the average occurrence probability of all characteristic species for each XY-value in this climatic space; and (3) defining a threshold of this averaged occurrence probability below which decline or loss of characteristic species, and hence significant habitat alteration, is to be expected. The climatic exceedance, or the climatic risk for a particular habitat at a particular site in the real-world landscape can then be calculated as the two-dimensional Euclidean distance of the temperature and precipitation values at this site and the climatic (= temperature and precipitation) thresholds of the respective habitat type (cf. Fig. 1).

### 2.2. Study area

The study area covers Austria, a landlocked country of Central Europe spanning approximately 84,000 km<sup>2</sup>. The mean annual temperature in Austria ranges between −9 °C at the highest peaks of the Alps and 10 °C in the Eastern lowlands, and the annual precipitation sum between a minimum of approximately 500 mm and a maximum of 2100 mm (www.worldclim.org). High climatic diversity is mainly due to the rugged terrain and elevational gradient of the Austrian Alps, which cover about two thirds of the country and are responsible for the high habitat diversity.



**Fig. 1.** Representation of the niche of a habitat type (Fagion sylvaticae = European beech forests) in temperature-precipitation space. Colours represent probability of occurrence as calculated from averaging SDM projections of all characteristic plant species of the habitat type. The black line delimits the climatic niche as defined by an averaged probability of occurrence of characteristic species of 80%. The precipitation axis was reversed as we assumed drought stress to be a more important challenge for Austrian plants under climate change than excess of water.

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