



Research article

Context-dependent environmental quality standards of soil nitrate for terrestrial plant communities



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ABSTRACT

Environmental quality standards (EQS) specify the maximum permissible concentration or level of a specific environmental stressor. Here, a procedure is proposed to derive EQS that are specific to a representative species pool and conditional on confounding environmental factors. To illustrate the procedure, a dataset was used with plant species richness observations of grasslands and forests and accompanying soil nitrate-N and pH measurements collected from 981 sampling sites in the Netherlands. Species richness was related to soil nitrate-N and pH with quantile regression allowing for interaction effects. The resulting regression models were used to derive EQS for nitrate conditional on pH, quantified as the nitrate-N concentrations at a specific pH level corresponding with a species richness equal to 95% of the species pool, for both grasslands and forest communities. The EQS varied between 1.8 mg/kg nitrate-N at pH 9–65 mg/kg nitrate-N at pH 4. EQS for forests and grasslands were similar, but EQS based on Red List species richness were considerably lower (more stringent) than those based on overall species richness, particularly at high pH levels. The results indicate that both natural background pH conditions and Red List species are important factors to consider in the derivation of EQS for soil nitrate-N for terrestrial ecosystems.

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1. Introduction

Environmental quality standards (EQS) specify the maximum permissible concentration or level of a specific environmental stressor (Van Straalen and Denneman, 1989; Van Goethem et al., 2015). The permissible level is generally set to be protective for 95% of the species pool (Kefford et al., 2011; Cormier & Suter II, 2013). EQS can be used to identify locations where unacceptable ecological risks may occur due to the stressor of concern. So far, EQS have mainly been established for chemical substances in aquatic ecosystems, typically derived on the basis of lab toxicity tests (Posthuma et al., 2002). However, not all species and environmental stressors lend themselves to laboratory testing, implying that laboratory-based EQS are biased toward easily cultured species and a limited set of environmental factors (Schipper et al., 2014). As species may differ strongly in their sensitivity to environmental

stressors (Pastorok et al., 2002), EQS derived from field monitoring data are expected to be more ecologically relevant, as field monitoring data cover the actual species pool of a particular area (Kefford et al., 2011; Posthuma et al., 2002; Schipper et al., 2014). Yet, in order to derive EQS for a particular stressor from field monitoring data, the influence of confounding effects of other stressors need to be accounted for, as other stressors may obscure the response to the environmental variable of concern.

Anthropogenic emissions of nitrogen (N) have been identified as a major cause of changes in terrestrial plant diversity, resulting in various calls and efforts to quantify effect thresholds (Sala et al., 2000; Bobbink et al., 2010; Simkin et al., 2016). Effects of N on plant community composition and diversity are, however, dependent on soil pH (Pausas and Austin, 2001; Bobbink et al., 2010; Azevedo et al., 2013; Mueller et al., 2013). Several studies identified the influence of acidification and base cation depletion on plant responses to nitrogen deposition and suggest that sensitivity to nitrogen addition is co-determined by soil pH (Clark et al., 2007; Horswill et al., 2008; Simkin et al., 2016). Hence, the response of plant species richness to N is likely influenced by potential

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interaction effects with pH, implying that EQS for N may at least partly depend on background pH conditions (Bobbink et al., 1998, 2010). This type of interaction has, however, not been included in the derivation of EQS up to now. This, in turn, implies a need for EQS for N that are conditional on soil pH (Simkin et al., 2016).

The aim of the present study was to derive EQS for soil nitrate-N, accounting for pH as potentially influencing factor. EQS for soil nitrate-N were derived for grasslands and forests separately, based either on the overall plant species pool or on the Red List species only. Red List species included those that are considered vulnerable, endangered or critically endangered according to the Red List of the Netherlands (Floron, 2012). The dataset used in this analysis comprised presence-absence observations of plant species along gradients of soil pH (3–10) and nitrate-N content of the soil (0.01–210 mg/kg), collected from 981 sampling sites of grassland and forest vegetation across the Netherlands (Wamelink et al., 2012). To derive the EQS, quantile regression models were established relating species richness to soil pH and nitrate-N, thereby accounting for interaction effects between the factors. Quantile regression was used to filter out the confounding influences of other environmental factors, because quantile regression based on one of the upper boundaries of the response variable distribution (e.g. the 0.90 or 0.95 quantile) is expected to show the constraints imposed by the explanatory environmental variables of concern (Cade and Noon, 2003; Iwasaki and Ormerod, 2012; Van Goethem et al., 2015). The resulting regression models were subsequently used to derive EQS for soil nitrate-N conditional on soil pH, with the EQS quantified as the nitrate-N concentration corresponding with 95% of the maximum species richness found along the gradient.

2. Methods

2.1. Data set

Monitoring data for grasslands and forests were selected from the ecological conditions (EC) database compiled by Wamelink et al. (2012). The dataset comprises 505 grassland and 592 forest vegetation relevés from the Netherlands, each accompanied by a soil nitrate-N (mg/kg dry weight) and pH measurement. The vegetation relevés were made according to the Braun-Blanquet method and were classified based on the vegetation classification of Schaminée et al. (1995) using the software tool Associa (Van Tongeren et al., 2008). A description of the selected grassland and forest vegetation types is given in the supporting information (Table S1). Several relevés were repeat visits. To reduce potential confounding effects of temporal autocorrelation, only the most recent relevé per site was included in the dataset, leading to a decrease in the number of grassland relevés from 505 to 469 and forest relevés from 592 to 512. Per relevé, both the overall species richness and the Red List species richness were determined. Red List species were identified using the Red List species database from FLORON (2012) (Table S2). The characteristics of the dataset used are given in Table 1.

2.2. Species richness response models

Before model construction, the nitrate-N measurements were \log_{10} -transformed. Subsequently, the pH and NO₃-N measurements were standardized to zero mean and unit variance. The standardized data were then used to establish regression models relating species richness to soil nitrate-N and pH for each of the four species group (i.e. for grassland species, forest species, Red List grassland species and Red List forest species). Model building was done with quantile regression, using the 95th percentile of the species richness observations in order to filter out the potential

Table 1

Characteristics (minimum, maximum and percentiles) of the grassland and forest datasets, describing both the overall species richness (SR) of the relevés and the measured soil pH and nitrate-N concentrations.

	Grassland			Forest		
	SR	pH	Nitrate-N (mg/kg)	SR	pH	Nitrate-N (mg/kg)
Min	4	3.9	0.1	1	3.4	0.1
0.05	9	4.3	0.2	7	3.8	0.7
0.5	23	5.6	4.2	24	5.0	11.9
0.95	47	8.3	64	46	8.3	101
Max	61	9.0	366	65	9.0	805

confounding effects of environmental variables other than soil nitrate-N and pH (Van Goethem et al., 2015). Because non-linear (unimodal) responses of grassland species richness have been observed for soil nitrate-N as well as pH (Bobbink et al., 2010; Simkin et al., 2016), quadratic terms were added as potential predictors in the regression modelling. All possible combinations of predictors were fitted, including interaction terms for soil nitrate-N and pH, whereby the quadratic predictor terms were only allowed if the linear term was included as well, in order to obtain models insensitive to linear transformations of the predictors (Nelder, 2000). For comparison, for each species group a model was fitted to soil nitrate-N only, including its quadratic term, without soil pH. The most parsimonious models were selected based on the Bayesian Information Criterion (BIC) (Lee et al., 2013). The 95% confidence intervals of the regression lines were calculated based on the covariance matrix of the regression parameter estimates (see Koenker, 1994; Van Goethem et al., 2015). Model building was performed with the quantreg and MuMIn packages in R (Koenker et al., 2013).

2.3. Environmental quality standards

From the quantile regression models, environmental quality standards (EQS) for nitrate-N conditional upon soil pH were derived as the nitrate-N concentration corresponding with a 5% reduction of the maximum species richness at a given pH background level. This was done for each of the four species groups and for pH background levels ranging from 4 to 9 in steps of 1 pH unit (i.e., covering the range of pH values in the dataset, see Table 1). The 5% reduction of species richness is in line with the way EQS are derived in the field of chemical risk assessment (see Posthuma et al., 2002).

For comparison, EQS were also derived from the quantile regression models based on nitrate-N only, where the EQS was defined as the nitrate-N concentration corresponding with a 5% reduction of the maximum species richness. Hence, these EQS do not account for the possible interaction with pH. Again this was done for each of the four species groups.

3. Results

3.1. Species richness response models

The regression models for nitrate-N only showed positive unimodal responses along the nitrate gradient (Fig. 1; Table S3). Optimum nitrate-N values (i.e., values corresponding with maximum species richness) were found in the range of 4.8–7.1 mg/kg, depending on the vegetation type (forest vs. grassland) and species group (all species vs. Red List species only) (Table 2). The response curves differed in their width and amplitude, where width was defined as the nitrate-N range at 75% of the maximum species richness and amplitude as the relative difference between

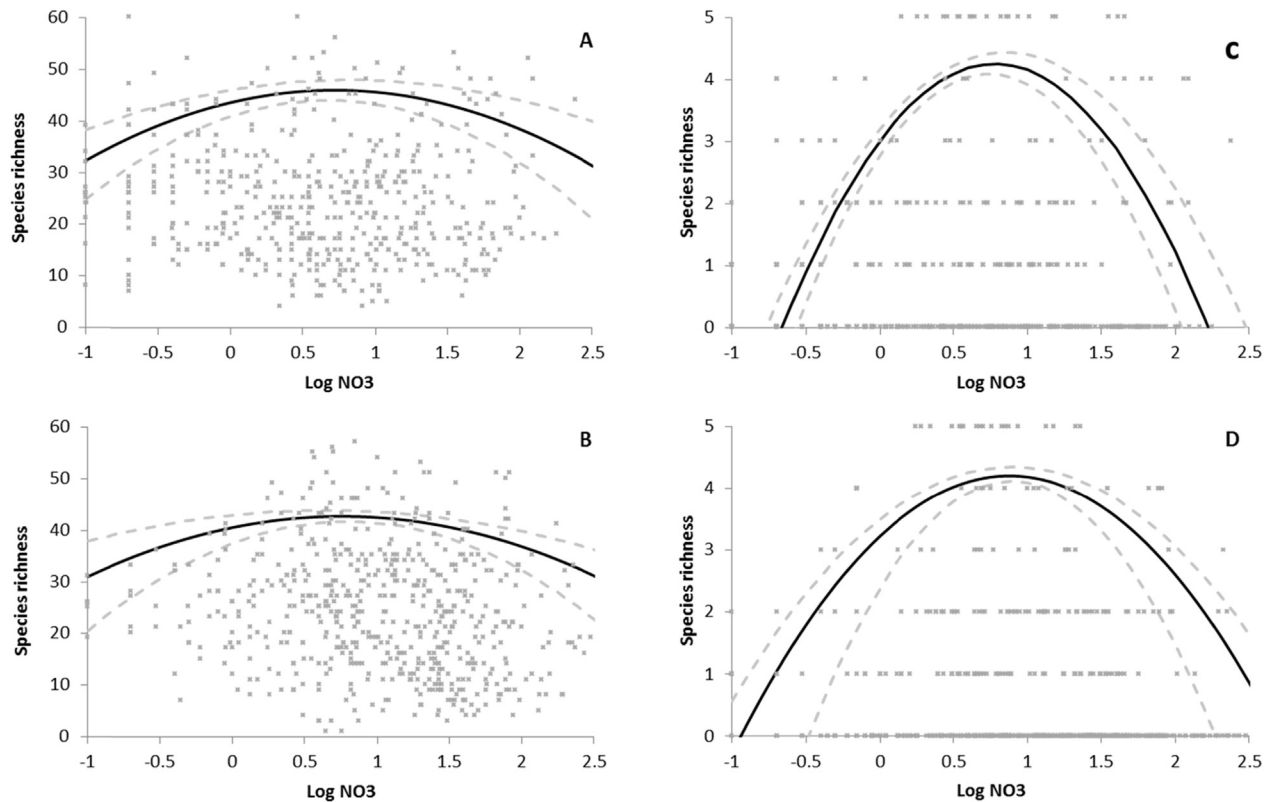


Fig. 1. Nitrate-N species richness response curves for grasslands (A), forests (B), grassland Red List species (C) and forest Red List species (D). The response curves were derived with quantile regression based on the 95th quantile. Grey dots indicate observed species richness; dotted lines indicate the 95% confidence intervals.

Table 2

Characteristics of the grassland and forest response curves for both overall species richness and Red List species richness, including the nitrate-N concentration corresponding with maximum species richness (nitrate- N_{opt} ; in mg/kg), the nitrate-N range at 75% of the maximum species richness (width; in mg/kg) and the amplitude (relative to the maximum species richness) of the species richness response curves. For comparison, values were also derived from a model that did not include pH.

	Grassland						Forest					
	All species			Red List species			All species			Red List species		
	Nitrate- N_{opt}	Width	Amplitude	Nitrate- N_{opt}	Width	Amplitude	Nitrate- N_{opt}	Width	Amplitude	Nitrate- N_{opt}	Width	Amplitude
No pH	4.8	190	0.29	6.3	32	1.0	5.6	251	0.27	7.1	60	1.0
pH 4	16	316	0.30	12.6	88	1.0	12.7	316	0.26	17.0	139	1.0
pH 5	7.9	316	0.18	6.3	51	1.0	5.0	316	0.16	10.2	84	1.0
pH 6	4.0	316	0.15	4.0	29	1.0	2.5	316	0.22	6.0	55	1.0
pH 7	2.5	316	0.19	2.5	16	1.0	1.3	218	0.28	3.8	32	1.0
pH 8	1.3	316	0.24	1.3	9	1.0	0.5	104	0.37	2.0	17	1.0
pH 9	0.8	141	0.33	0.8	5	1.0	0.3	35	0.51	1.4	10	1.0

the maximum and minimum species richness estimated along the nitrate-N gradient (Table 2). The widths ranged from 32 mg/kg for the grasslands Red List species to 251 mg/kg for forests. The amplitude ranged from 0.27 for forests to 1.0 for both grassland and forest Red List species.

The regression models with pH interaction included also showed positive unimodal responses of plant species richness to soil nitrate-N (Fig. 2). Interaction between nitrate and pH was indicated by the regression coefficient for the interaction term (Table S3). Irrespective of vegetation type and species group, the soil nitrate-N concentration corresponding with the highest species richness decreased with increasing pH, up to a factor of 42 along the pH range (Table 2). The same was observed for the widths of the response curves, which decreased up to a factor of 17 from low to high pH. In other words, the optimum nitrate-N concentration was

lowest and the slope of the response curve was steepest at the highest pH levels (Fig. 2, Table 2).

3.2. Environmental quality standards

Environmental quality standards derived from the regression models based on nitrate-N only ranged from 12.6 for Red List grassland species to 32.9 mg/kg for all forest species (Table 3). When accounting for the interaction between nitrate-N and pH, the EQS for nitrate-N became clearly pH-dependent. At pH 4, the EQS for nitrate-N was a factor of 1.4–2.5 higher compared to the EQS independent of pH. In contrast, for pH 9 the EQS for nitrate-N was a factor of 1.6–7 lower (more stringent) compared to the pH-independent EQS (Table 3). The large overlap of the 95% confidence intervals indicated that the EQS for nitrate-N were not

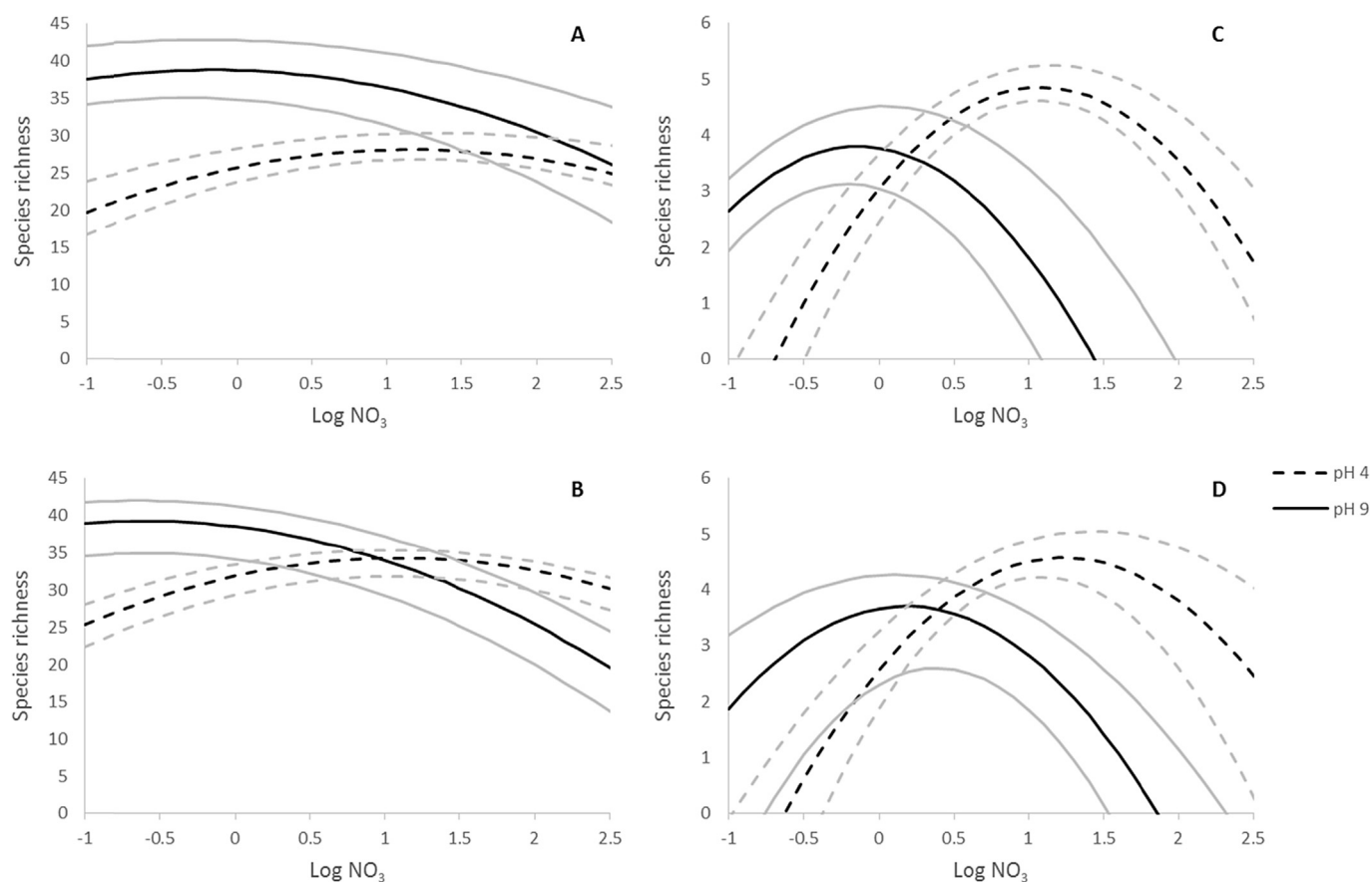


Fig. 2. Species richness response curves for soil nitrate-N conditional on background pH conditions (pH 4 and 9) for grasslands (A), forests (B), grassland Red List species (C) and forest red list species (D). The response curves were derived with quantile regression based on the 95th quantile. Grey lines indicate the 95% confidence intervals.

Table 3

Environmental quality standards for soil nitrate-N (in mg/kg) conditional on background pH conditions for grasslands, forest and their respective Red List species. For comparison, EQS were also derived from a model that did not include pH. Numbers between brackets represent the 95% confidence intervals.

EQS (mg/kg)	Grasslands		Forests	
	All species	Red list	All species	Red list
pH 4	64.6 (42.6–84.0)	29.5 (19.5–38.4)	44.7 (31.3–62.6)	42.2 (27.9–59.1)
pH 5	56.2 (38.2–67.4)	17.0 (11.6–23.8)	39.8 (27.9–59.7)	26.3 (16.8–34.2)
pH 6	46.8 (28.1–65.5)	12.3 (8.6–17.2)	33.1 (21.2–53.0)	17.0 (10.5–27.2)
pH 7	30.2 (18.7–45.3)	5.2 (3.1–6.8)	25.7 (17.0–43.7)	9.5 (6.3–11.4)
pH 8	24.0 (14.4–38.4)	3.2 (2.2–5.1)	17.8 (12.1–24.9)	6.0 (4.1–7.8)
pH 9	16.6 (11.6–28.2)	1.8 (1.2–2.2)	13.2 (7.9–19.8)	3.5 (2.5–4.2)
No pH	26.3 (17.8–49.5)	12.6 (8.3–16.4)	32.9 (23.4–42.7)	17.2 (11.4–24.1)

different between grasslands and forests. In contrast, the EQS for Red List species richness tended to be considerably lower (more stringent) compared to the EQS based on overall species richness, in particular at higher pH levels.

4. Discussion

4.1. Species richness response curves

The response curves obtained with the quantile regression models suggest a unimodal relationship between plant species richness and soil nitrate-N (Figs. 1 and 2). The unimodal response can be explained by the fact that oligotrophic conditions limit the occurrence of species due to nutrient limitation, while eutrophic conditions can lead to an increase in productivity and a subsequent decrease in species diversity due to competitive interactions

(Goldberg and Miller, 1990; Bobbink et al., 2010). Furthermore, the response curve for nitrate-N and pH combined suggest that in alkaline conditions the maximum species richness occurs at lower nitrate-N levels and an increase in nitrate-N beyond the optimum results in a stronger species richness decline. These patterns might be explained by the influence of pH on nitrogen cycles in soils (Bobbink et al., 2010; Bolan et al., 2003). Nitrification is reduced below pH 6 and is almost absent below pH 4.5, resulting in decreased nitrate-N uptake and accumulation of NH₄ (Alexander, 1977; Bolan et al., 2003; Marschner, 1995; Bobbink et al., 2010). Some plant species may use NH₄ as a source of nitrogen, whereas for others the accumulated NH₄ may reach toxic concentrations (Alexander, 1977; Bolan et al., 2003). This may result in a shift in species composition towards species that primarily rely on ammonium as a nitrogen source (Bobbink et al., 2010; Bolan et al., 2003; Van den Berg et al., 2005). Species assemblages occurring at

low pH conditions are, therefore, potentially less sensitive to changes in nitrate-N concentration, resulting in a gentler slope of the response curve and a maximum species richness at higher nitrate-N concentrations. In alkaline soils, on the other hand, NH_4 is nitrified more rapidly due to an increase in the activity of micro-organisms involved in nitrification (Lyngstad, 1992; Puttanna et al., 1999). Alkaline conditions may, therefore, favor species that are specialized in the uptake of nitrate as nitrogen source (Zvereva et al., 2008; Tinsley, 1973; Adams, 1986). These species assemblages are potentially more sensitive to changes in soil nitrate concentrations as there is high resource competition between these efficient nitrate consumers, resulting in a steeper curve and maximum species richness occurring at lower nitrate-N concentrations (Fargione and Tillman, 2006).

Other processes occurring at low or high pH levels may also influence species composition along the nitrate gradient. For instance, at low pH levels the solubility of aluminum is enhanced. Given that aluminum toxicity is a major factor limiting plant production on acid soils, especially affecting forest species (Delhaize and Ryan, 1995; Silva, 2012), this process may, therefore, also affect species composition at low pH levels. Furthermore, the 95% confidence intervals showed considerable uncertainty for the response curves of nitrate-N and pH combined, especially for the Red List species. This might be explained by the fact that the variation in species richness was much lower for Red List species, as there were only a few relevés with more than five Red List species.

5. Environmental quality standards

According to the EQS derived in our study, grasslands and forest are approximately equally sensitive to an increase in soil nitrate (Table 3). This is in line with other studies that showed that both grassland and forest communities in the Netherlands, often dominated by species with low nutrient requirements, are sensitive to eutrophication (Stevens et al., 2011; Gall et al., 2015). Red List species appeared to be more sensitive to eutrophication than plant species in general, according to the more stringent EQS. In general, Red List species have narrower tolerance ranges compared to common species, especially for nutrients (Wamelink et al., 2014). An increase in soil nitrate-N concentration is, therefore, more likely to lead to a decrease in Red List species richness than species richness overall. By deriving environmental response curves and EQS for specific species groups, sensitive species groups can be identified and better protected (see also Wamelink et al., 2003; Azevedo et al., 2014). Among applied biologists concerned with biodiversity indicators, the relation between “species richness” and “conservation value” is a fervently debated issue, and a recurrent source of misunderstandings (Duelli and Obrist, 2003). The overall number of species is still seen as an important indicator of ecosystem quality, while, for instance in this case, it may be more relevant to look specifically at Red List species, as these are species with high conservation priority (Rodrigues et al., 2006; Gotelli and Colwell, 2006).

For 15% of the grassland plots (26% of the plots with Red List species) and 24% of the forest plots (42% for Red List species) in our dataset, measured soil nitrate-N concentrations exceeded the respective EQS based on nitrate-N only. However, the EQS for nitrate-N conditional on pH suggest a more nuanced interpretation. For instance, based on a characteristic pH of 4.5 (Schaminée et al., 1995), fen meadows (*Cirsio dissecti*-*Molinietum*) have an EQS of 60.1 mg/kg nitrate-N, implying that nitrate-N concentrations measured in fen meadows would exceed the EQS in only 3% of the plots. Similarly, hay meadows (*Arrhenatheretum elatioris*) have a characteristic pH of 7.5 (Schaminée et al., 1995) hence an EQS of 26.9 mg/kg nitrate-N, which corresponds to exceedance of the EQS

in 2% of the plots. Hence, although both vegetation types are considered grasslands, they have clearly different EQS for nitrate-N. This suggests that natural background pH conditions are an important factor to consider in the derivation of EQS for soil nitrate. The present study further showed that, in general, context-dependent response curves for environmental stressors can help to derive EQS tailored to specific nature conservation areas, as opposed to broadly used general EQS (Erisman et al., 2003; Simkin et al., 2016). This in turn is a great asset in the prioritization of targeted environmental stressor management measures.

Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.jenvman.2016.08.037>.

References

- Adams, S.N., 1986. The interaction between liming and forms of nitrogen-fertilizer on established grassland. *J. Agric. Sci.* 106, 509–513.
- Alexander, M. (Ed.), 1977. *Introduction to Soil Microbiology*, second ed. Wiley, New York.
- Azevedo, L.B., 2014. Development and application of stressor-response relationships of nutrients. Radboud University.
- Azevedo, L.B., van Zelm, R., Hendriks, A.J., Bobbink, R., Huijbregts, M.A.J., 2013. Global assessment of the effects of terrestrial acidification on plant species richness. *Environ. Pollut.* 174, 10–15.
- Bobbink, R., Hornung, M., Roelofs, J.G.M., 1998. The effects of air-borne nitrogen pollutant on species diversity in natural and semi-natural European vegetation. *J. Ecol.* 86, 717–738.
- Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinnerby, S., Davidson, E., Dentener, F., Emmett, B., Erisman, J.W., Fenn, M., Gilliam, F., Nordin, A., Pardo, L., De Vries, W., 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecol. Appl.* 20 (1), 30–59.
- Bolan, N.S., Adriano, D.C., Curtin, D., 2003. Soil acidification and liming interactions with nutrient and heavy metal transformation and bioavailability. *Adv. Agron.* 78, 215–272.
- Cade, B.S., Noon, B.R., 2003. A gentle introduction to quantile regression for ecologists. *Front. Ecol. Environ.* 1, 412–420.
- Clark, C.M., Cleland, E.E., Collins, S.L., Fargione, J.E., Gough, L., Gross, K.L., Pennings, S.C., Suding, K.N., Grace, J.B., 2007. Environmental and plant community determinants of species loss following nitrogen enrichment. *Ecol. Lett.* 10, 596–607.
- Cormier, S.M., Suter II, G.W., 2013. A method for deriving water-quality benchmarks using field data. *Environ. Toxicol. Chem.* 32, 255–262.
- Duelli, P., Obrist, M.K., 2003. Biodiversity indicators: the choice of values and measures. *Agric. Ecosys. Environ.* 98 (1), 87–98.
- Delhaize, E., Ryan, P.R., 1995. Aluminum toxicity and tolerance in plants. *Plant Physiol.* 107, 315–321.
- Erisman, J.W., Grennfelt, P., Sutton, M., 2003. The European perspective on nitrogen emission and deposition. *Environ. Int.* 29 (2), 311–325.
- Fargione, J., Tillman, D., 2006. Plant species traits and capacity for resource reduction predict yield and abundance under competition in nitrogen-limited grassland. *Funct. Ecol.* 20 (3), 533–540.
- Goldberg, D.E., Miller, T.E., 1990. Effects of different resource additions of species diversity in an annual plant community. *Ecology* 71, 213–225.
- Gall, A.C.I., Hettelingh, J.P., Posch, M., 2015. Mapping Critical Loads for Ecosystems. RIVM.
- Gotelli, N.J., Colwell, R.K., 2006. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecol. Lett.* 4 (4), 379–391.
- Horswill, P., O'Sullivan, O., Phoenix, G.K., Lee, J.A., Leake, J.R., 2008. Base cation depletion, eutrophication and acidification of species-rich grasslands in response to longterm simulated nitrogen deposition. *Environ. Pollut.* 155, 336–349.
- Iwasaki, Y., Ormerod, S.J., 2012. Estimating safe concentrations of trace metals from intercontinental field data on river macroinvertebrates. *Environ. Pollut.* 166, 182–186.
- Kefford, B.J., Marchant, R., Schafer, R.B., Metzeling, L., Dunlop, J.E., Choy, S.C., Goonan, P., 2011. The definition of species richness used by species sensitivity distributions approximates observed effects of salinity on stream macroinvertebrates. *Environ. Pollut.* 159, 302–310.
- Koenker, R., 1994. Confidence Intervals for Regression Quantiles. *Asymptotic Statistics*. Physica-Verlag HD, pp. 349–359.
- Koenker, R., Portnoy, S., Tian, P., Zeileis, A., Grosjean, P., 2013. Package Quantreg: Quantile Regression and Related Methods. <http://cran.r-project.org/web/packages/quantreg/quantreg.pdf>.
- Lee, E.R., Noh, H., Park, B.U., 2013. Model selection via bayesian information criterion for quantile regression models. *J. Am. Stat. Assoc.* <http://dx.doi.org/10.1080/>

- 01621459.2013.836975.
- Lyngstad, I., 1992. Effect of liming on mineralization of soil-nitrogen as measured by plant uptake and nitrogen released during incubation. *Plant Soil* 144, 247–253.
- Marschner, H. (Ed.), 1995. *Mineral Nutrition of Higher Plants*, second ed. Academic Press, London.
- Mueller, K.E., Hobbie, S.E., Tilman, D., Reich, P.B., 2013. Effects of plant diversity, N fertilization, and elevated carbon dioxide on grassland soil N cycling in a long-term experiment. *Glob. Change Biol.* 19, 1249–1261.
- Nelder, J.A., 2000. Functional marginality and response-surface fitting. *J. Appl. Stat.* 27 (1), 109–112.
- Pastorok, R.A., Bartell, S.M., Ferson, S., Ginsberg, L.R. (Eds.), 2002. *Ecological Modeling in Risk Assessment: Chemical Effects on Populations, Ecosystems and Landscapes*. Lewis Publishers, Boca Raton, FL.
- Pausas, J.G., Austin, M.P., 2001. Patterns of plant species richness in relation to different environments: an appraisal. *J. Veg. Sci.* 12, 153–166.
- Posthuma, L., Traas, T., Suter, G.W., 2002. General introduction to species sensitivity distributions. In: Posthuma, L. (Ed.), *Species Sensitivity Distributions in Ecotoxicology*. CRC Press, Boca Raton, 3e10.
- Puttanna, K., Gowda, N.M.N., Rao, E.V.S.P., 1999. Effect of concentration, temperature, moisture, liming and organic matter on the efficacy of the nitrification inhibitors benzotriazole, onitrophenol, m-nitroaniline and dicyandiamide. *Nutr. Cycles Agroecosyst.* 54, 251–257.
- Rodrigues, A.S.L., Pilgrim, J.D., Lamoreux, J.F., Hoffmann, M., Brooks, T.M., 2006. The value of the IUCN Red List for conservation. *Trends Ecol. Evol.* 21 (2), 71–76.
- Sala, O.E., et al., 2000. Global biodiversity scenarios for the year 2100. *Science* 287 (5459), 1770–1774.
- Schaminée, J.H.J., Stortelder, A.H.F., Westhoff, V., 1995. *De Vegetatie Van Nederland*, Deel 1. Uppsala/Leiden.
- Schipper, A.M., Hendriks, A.J., Ragas, A.M.J., Leuven, R.S.E.W., 2014. Disentangling and ranking the influences of multiple environmental factors on plant and soil-dwelling arthropod assemblages in a river Rhine floodplain area. *Hydrobiologia* 729, 133–142.
- Silva, S., 2012. Aluminum toxicity targets in plants. *J. Bot.* <http://dx.doi.org/10.1155/2012/219462>.
- Simkin, S.M., et al., 2016. Conditional vulnerability of plant diversity to atmospheric nitrogen deposition across the United States. *PNAS* 113, 4086–4091.
- Sparrius, L., Ode, B., Beringen, R., 2012. *Rode Lijst Vaatplanten 2012 volgens Nederlandse en IUCN-criteria*. FLORON.
- Stevens, C.J., et al., 2011. The impact of nitrogen deposition on acid grasslands in the Atlantic region of Europe. *Envir. Pollut.* 159 (10), 2243–2250.
- Tinsley, J., 1973. Leaching of nitrogen and other soil constituents in a podzol under scots pine following nitrogen fertilizer and liming treatments. *J. Sci. Food Agric.* 24, 483–483.
- Van Den Berg, L.J., Dorland, E., Vergeer, P., Hart, M.A., Bobbink, R., Roelofs, J.G., 2005. Decline of acid-sensitive plant species in heathland can be attributed to ammonium toxicity in combination with low pH. *New Phytol.* 166 (2), 551–564.
- Van Goethem, T.M., Huijbregts, M.A.J., Wamelink, G.W.W., Schipper, A.M., 2015. How to assess species richness along single environmental gradients? Implications of potential versus realized species distributions. *Environ. Pollut.* 200, 120–125.
- Van Straalen, N.M., Denneman, C.A.J., 1989. Ecotoxicological evaluation of soil quality criteria. *Ecotoxicol. Environ. Saf.* 18, 241–251.
- Van Tongeren, O., Gremmen, N., Hennekens, S., 2008. Assignment of relevés to pre-defined classes by supervised clustering of plant communities using a new composite index. *J. Veg. Sci.* 19 (4), 525–536.
- Wamelink, G.W.W., Ter Braak, C.J.F., Van Dobben, H.F., 2003. Changes in large-scale patterns of plant biodiversity predicted from environmental economic scenarios. *Landsc. Ecol.* 18, 513–527.
- Wamelink, G.W.W., van Adrichem, M.H.C., van Dobben, H.F., Frissel, J.Y., den Held, M.E., Joosten, V., Malinowska, A.H., Slim, P.A., Wegman, R.M.A., 2012. Vegetation relevés and soil measurements in The Netherlands: the ecological conditions database (EC). *Biodivers. Ecol.* 4 (17), 125–132.
- Wamelink, G.W.W., Goedhart, P.W., Frissel, J.Y., 2014. Why some plant species are rare. *PLoS One* 9 (7), e102674. <http://dx.doi.org/10.1371/journal.pone.0102674>.
- Zvereva, E.L., Toivonen, E., Kozlov, M.V., 2008. Changes in species richness of vascular plants under the impact of air pollution: a global perspective. *Glob. Ecol. Biogeogr.* 17 (3), 305–319.