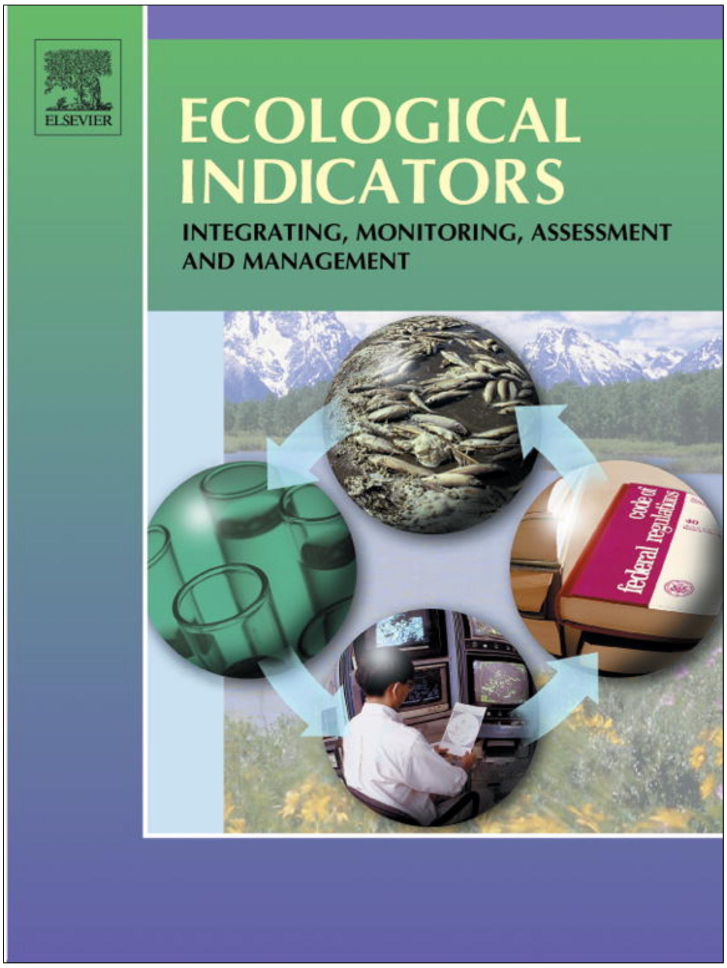
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Letter to the Editor

Uncertainties in biotic indicators and a corrigendum to Ponader et al. (2007): Implications for biomonitoring

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ABSTRACT

Many bioindicators based on species composition have been proposed during the past 30 years in both terrestrial and freshwater ecosystems. Kelly (2011) raised the need to better communicate the usefulness and meaning of the indices and their uncertainties. A survey of European bioindicators of freshwater ecosystems has recently highlighted that most indices have not even considered uncertainties. While recording errors have sometimes been taken into account into the calculation of index uncertainties, the reliability of biotic indices in terms of causality and predictability between a given environmental pressure and bioindicator response have seldom been considered. This has led to serious misunderstanding of what bioindicators may be able to achieve. Here a correction to Ponader et al. (2007) is presented showing that the tolerance (ecological niche) of individual species of diatoms along nutrient enrichment gradients is much wider than formerly presented. The correction, together with the comments by Kelly (2011) and recent whole ecosystem experiments seriously question the reliability of diatoms as nutrient indicators, similarly to recent findings with aquatic macrophytes. A more mechanistic basis for biomonitoring is needed based on current ecological science, rather than early 20th century community ecology.

1. Uncertainties in biotic indicators

Many bioindicators based on species composition have been proposed during the past 30 years in both terrestrial and freshwater ecosystems (including diatom, macroinvertebrate, fish, macrophyte). The recent comment by Kelly (2011) on the comparison of European diatom indicators by Besse-Lototskaya et al. (2011) raised the need to better communicate the usefulness and meaning of the indices and their uncertainties (see also Birks et al., 1990; Clarke et al., 2003). It is quite staggering that a recent survey has highlighted that most indices have not even considered uncertainties, as exemplified for rivers in Fig. 1. When uncertainties have been considered, they were mostly limited to the precision of the indices, such as sampling replicability and taxonomic identification, rather than ecological accuracy such as the causality and predictability between a given environmental pressure and bioindicator response (Demars et al., 2012). Causation may be brought into the equation by using species traits such as growth form, life history, ecophysiology (e.g. macroinvertebrates in lakes, Brodersen et al., 2004; macrophyte in rivers, Demars and Trémolières, 2009) or designing whole ecosystem experiments (e.g. Gudmundsdóttir, 2012). Recent studies have seriously questioned the ability of bioindicators to detect reliably nutrient enrichment using aquatic macrophytes and diatoms (e.g. Demars et al., 2012; Gudmundsdóttir, 2012). Predictability and its associated statistical errors may be assessed by testing the indices on independent datasets using appropriate statistical analyses (e.g. Birks et al., 1990; Clarke et al., 2003; Holden et al., 2008; Demars et al., 2012).

As Kelly (2011) mentioned, these indicators alone may not help a great deal to evaluate the ecological status of our rivers as requested by current legislation (e.g. European Water Framework Directive, WFD or Clean Water Act in the USA). Clews and Ormerod (2009) suggested combining standard biotic indices may improve biodiagnostic monitoring, but because of potential interaction effects of multiple stressors and confounding factors it is unlikely to succeed beyond broad predictions at regional scales (see Demars et al., 2012).

What is really needed in biomonitoring is a shift in thinking towards a more mechanistic approach based on ecological science. It is the integration of multiple trophic interactions and ecosystem functioning at multiple spatial and temporal scales that will likely increase our understanding and ultimately management efficiency (e.g. Woodward et al., 2010; O'Gorman et al., 2012; Palmer and Febria, 2012).

2. Corrigendum to Ponader et al. (2007)

A common approach to building bioindicators based on taxonomic composition (occurrence–abundance) is to determine species scores (optima) and species range (tolerance or niche breadth sensu Hutchinson, 1957) along an environmental gradient. Optima and tolerances have traditionally been derived from expert judgements or more formal statistical analyses (e.g. canonical correspondence analysis or general linear modelling) based on a calibration dataset. Then, from a field survey or sample collection, a weighted average method is used to calculate the biotic index (sum of species optima weighted by species abundance and tolerance,

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Table 1
Species apparent optima and tolerances estimated by weighted averaging regression and calibration techniques.

Taxon name	TP				TN			
	Optima		Tolerance		Optima		Tolerance	
	$\log_{10}$ μg P L^{-1}	μg P L^{-1}	$\log_{10}$ μg P L^{-1}	μg P L^{-1}	$\log_{10}$ μg N L^{-1}	μg N L^{-1}	$\log_{10}$ μg N L^{-1}	μg N L^{-1}
Achnanthidium rivulare Potapova and Ponader	1.40	25	0.42	10–66	2.98	960	0.43	350–2600
Navicula antonii Lange-Bertalot	1.48	30	0.33	14–64	2.96	920	0.24	530–1610
Cocconeis placentula var. *euglypta* (Ehrenberg) Cleve	1.48	31	0.40	12–76	2.92	830	0.38	350–2010
Nitzschia fonticola Grunow	1.49	31	0.42	12–81	2.96	920	0.36	400–2090
Achnanthidium minutissimum (Kützing) Czarnecki	1.51	33	0.42	12–85	3.01	1020	0.39	410–2510
Encyonema minutum (Hilse) Mann	1.54	35	0.37	15–82	3.02	1040	0.33	490–2220
Navicula cryptocephala Kützing	1.57	37	0.40	15–94	3.09	1240	0.42	470–3290
Navicula cryptotenella Lange-Bertalot	1.59	39	0.37	17–91	3.06	1160	0.33	540–2500
Cymbella tumida (Brébisson ex Kützing) Van Heurck	1.60	40	0.41	15–103	2.95	890	0.33	420–1900
Cocconeis pediculus Ehrenberg	1.61	40	0.43	15–110	3.04	1100	0.43	410–2930
Navicula decussis Østrup	1.63	42	0.35	19–94	3.03	1070	0.34	490–2370
Diatoma vulgaris Bory	1.64	43	0.34	20–94	3.09	1220	0.27	660–2250
Nitzschia archibaldii Lange-Bertalot	1.64	44	0.46	15–127	3.06	1150	0.33	530–2490
Nitzschia linearis (Agardh ex W. Smith) W. Smith	1.64	44	0.35	19–98	2.97	930	0.35	410–2090
Amphora inariensis Krammer	1.65	45	0.38	19–109	3.10	1260	0.35	570–2790
Navicula tripunctata (O.F. Müller) Bory	1.66	46	0.40	18–115	3.10	1270	0.32	610–2650
Reimeria sinuata (Gregory) Kociolek and Stoermer	1.67	47	0.35	21–106	3.11	1280	0.31	630–2610
Gomphonema minutum (Agardh) Agardh	1.68	48	0.38	20–115	3.12	1320	0.30	670–2620
Synedra ulna Ehrenberg	1.68	48	0.33	23–103	3.10	1260	0.24	730–2190
Navicula capitatoradiata Germain	1.70	50	0.34	23–109	3.11	1270	0.28	670–2430
Nitzschia liebethruthii Rabenhorst	1.71	52	0.40	21–129	3.18	1500	0.33	710–3180
Nitzschia recta Hantz. ex Rabenhorst	1.72	52	0.42	20–136	3.30	1980	0.35	890–4400
Planothidium lanceolatum (Brébisson ex Kützing) Lange-Bertalot	1.72	52	0.33	24–112	3.17	1470	0.32	700–3080
Gomphonema parvulum (Kützing) Kützing	1.72	53	0.37	23–122	3.12	1330	0.31	650–2720
Amphora pediculus (Kützing) Grunow	1.73	54	0.32	26–113	3.16	1450	0.26	800–2610
Achnanthes subhudsonis var. *kraeuselii* Cholnoky	1.74	55	0.50	18–174	3.17	1470	0.36	650–3360
Navicula perminuta Grunow	1.75	56	0.32	27–118	3.21	1640	0.23	960–2800
Melosira varians Agardh	1.77	58	0.35	26–132	3.19	1540	0.26	840–2810
Caloneis bacillum (Grunow) Cleve	1.77	59	0.40	23–150	3.19	1550	0.33	730–3300
Navicula germainii Wallace	1.78	61	0.31	29–125	3.17	1490	0.23	890–2500
Mayamaea atomus (Kützing) Lange-Bertalot	1.78	61	0.22	37–101	3.20	1600	0.16	1100–2340
Nitzschia dissipata (Kützing) Grunow	1.80	63	0.38	26–151	3.21	1610	0.27	870–2970
Cocconeis placentula var. *lineata* (Ehrenberg) Van Heurck	1.80	63	0.35	28–142	3.20	1600	0.25	900–2850
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	1.81	64	0.35	28–144	3.22	1640	0.31	810–3330
Staurosirella pinnata (Ehrenberg) Williams et Round	1.81	64	0.54	19–223	3.12	1320	0.41	520–3400
Rhoicosphenia abbreviata (Agardh) Lange-Bertalot	1.81	64	0.37	27–152	3.20	1570	0.29	800–3080
Achnanthes conspicua Mayer	1.83	67	0.25	38–119	3.27	1840	0.21	1120–3020
Navicula erifuga Lange-Bertalot	1.83	68	0.35	31–151	3.18	1530	0.25	860–2720
Frustulia vulgaris (Thwaites) De Toni	1.83	68	0.32	33–142	3.16	1440	0.22	870–2370
Navicula canalis Patrick	1.84	69	0.40	28–173	3.32	2090	0.27	1120–3920
Nitzschia palea (Kützing) W. Smith	1.84	70	0.37	30–164	3.24	1720	0.26	950–3110
Navicula rostellata Kützing	1.85	72	0.26	39–130	3.25	1780	0.19	1140–2780
Navicula minima Grunow	1.86	72	0.34	33–159	3.21	1630	0.27	870–3030
Navicula symmetrica Patrick	1.86	72	0.34	33–160	3.21	1610	0.24	930–2810
Gyrosigma acuminatum (Kützing) Rabenhorst	1.86	73	0.38	30–174	3.15	1400	0.25	800–2470
Cyclotella meneghiniana Kützing	1.87	74	0.44	27–202	3.17	1490	0.28	790–2820
Navicula gregaria Donkin	1.87	74	0.35	33–165	3.25	1760	0.25	990–3140
Navicula subminuscula (Manguin)	1.87	74	0.30	37–147	3.21	1620	0.19	1050–2500
Navicula lanceolata (Agardh) Ehrenberg	1.87	75	0.36	33–170	3.29	1930	0.23	1130–3310
Navicula capitata Ehrenberg	1.88	75	0.35	34–167	3.19	1560	0.23	920–2650
Nitzschia amphibia Grunow	1.88	75	0.31	37–155	3.27	1840	0.26	1020–3310
Fragilaria vaucheriae (Kützing) Petersen	1.88	76	0.48	25–228	3.23	1700	0.31	830–3490
Nitzschia inconspicua Grunow	1.88	76	0.27	41–142	3.26	1830	0.20	1140–2930
Nitzschia capitellata Hustedt	1.88	76	0.28	40–144	3.34	2190	0.21	1350–3550
Achnanthidium exiguum (Grunow) Czarnecki	1.88	77	0.34	35–167	3.27	1880	0.26	1050–3390
Bacillaria paradoxa Gmelin	1.89	77	0.36	34–177	3.39	2460	0.24	1400–4300
Navicula ruttnerii var. *capitata* Hustedt	1.90	79	0.28	42–149	3.28	1900	0.21	1170–3100
Navicula agrestis Hustedt	1.91	81	0.33	38–174	3.27	1870	0.23	1100–3150
Staurosira construens (Ehrenberg) Williams et Round	1.92	82	0.47	28–241	3.20	1590	0.42	600–4220
Cyclotella pseudostelligera Hustedt	1.93	86	0.40	34–214	3.32	2080	0.25	1180–3660
Sellaphora seminulum (Grunow) Mann	1.95	88	0.32	42–187	3.26	1840	0.24	1070–3160
Sellaphora pupula (Kützing) Mereschkowsky	1.95	89	0.34	40–196	3.29	1940	0.21	1190–3160
Gomphonema kobayasii Kociolek and Kingston	1.96	91	0.41	35–234	3.25	1790	0.26	990–3210
Navicula ingenua Hustedt	1.96	92	0.25	52–163	3.25	1770	0.11	1360–2310
Navicula recens Lange-Bertalot	2.00	101	0.36	44–231	3.24	1740	0.19	1120–2720
Aulacoseira granulata (Ehrenberg) Simonsen	2.09	124	0.50	40–388	3.42	2630	0.26	1460–4730
Cyclotella atomus Hustedt	2.19	156	0.34	71–339	3.32	2070	0.18	1370–3130
Luticola goeppertiana (Bleisch) Mann	2.21	163	0.33	76–350	3.41	2560	0.15	1830–3590

Table 2
Elodea species optimum and tolerance values for PO_4-P (μg L^{-1}) calculated from a calibration dataset of 74 sites using a Gaussian log-linear regression model (data from Demars and Trémolières, 2009). The standard error (±SE) of the optimum and tolerance are also indicated.

	$\log_{10}(x+1)$		Back-transformed	
	Optimum ± SE	Tolerance ± SE	Optimum	Tolerance
E. canadensis Michx.	1.41 ± 0.10	0.43 ± 0.10	25	9–68
E. callitrichoides (Rich.) Casp.	2.13 ± 0.12	0.44 ± 0.08	134	48–371
E. nuttallii (Planch.) H. St. John.	2.12 ± 0.30	0.81 ± 0.27	131	19–850

divided by the sum of the weights). The uncertainties in species abundance, optima and tolerance are however rarely propagated in the calculations of biomonitoring indices (Demars et al., 2012), despite the availability of methods and softwares (e.g. Clarke et al., 2003; Juggins, 2003).

While Ponader et al. (2007, 2008) studies count among the best efforts to estimate statistical errors in predictability of diatom bioindicators; there is an error in the calculation of the back-transformed values for the diatom species tolerance to total phosphorus (TP) and total nitrogen (TN) in Ponader et al. (2007). Ponader et al. (2007) estimated the species apparent optima and tolerances by weighted averaging (WA) calibration and regression techniques. Since TP and TN were initially $\log_{10}$ transformed, the optima and tolerances reported by the C^2 software (Juggins, 2003) were also on a $\log_{10}$ scale. Ponader et al. (2007) back-transformed the optimum (u) and tolerance (t) as 10^u and 10^t. This is not correct for tolerance. The ecological tolerance of the species should instead be back-transformed as the range $10^{(u-t)}$ to $10^{(u+t)}$. This also gives incidentally more realistic ranges (much wider), than what was originally published (see Table 1).

This correction shows that many individual diatom taxa have a wide ecological niche along the nutrient gradients, and therefore may not be as good an indicator as was formally thought (see Kelly, 2011). This is particularly relevant to a recent study showing no response of the TDI (Trophic Diatom Index, Kelly and Whitton, 1995) to whole stream nutrient addition experiments in nutrient poor sub-arctic streams (Gudmundsdóttir, 2012). The reasons are that it may be far too simplistic to try to relate single pressures to a community based index because of all the biotic interactions in the ecosystem (see O'Gorman et al., 2012) and other confounding factors such as electric conductivity or pH (Stevenson et al., 2008; Demars et al., 2012).

Ponader et al. (2008), in a similar study, correctly reported the tolerance in $\log_{10}$ μg P L^{-1} units. Demars and Trémolières (2009) also reported untransformed tolerances in $\log_{10}(x+1)$ units for PO_4-P and NH_4-N concentrations (also in μg L^{-1}). While these

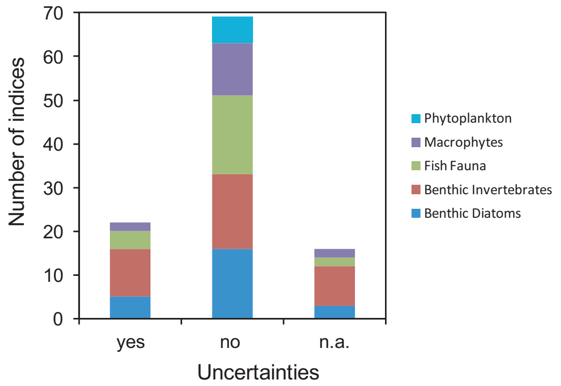

Fig. 1. Consideration of uncertainties in 107 river biomonitoring methods covering five biological quality elements (BQEs) of the WFD. Data extracted from Birk et al. (2010). WISER methods database. Version: March 2011. Available at http://www.wiser.eu/results/method-database/ (Birk et al., 2012). n.a. = not available.

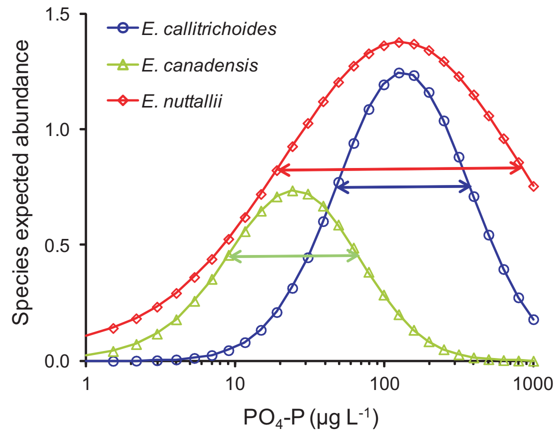

Fig. 2. Expected abundance, with back-transformed PO_4-P model results in three species of *Elodea* (aquatic vascular plant) calculated with a Gaussian log-linear regression model from a calibration dataset of 74 sites (data from Demars and Trémolières, 2009). Horizontal arrows represent the back-transformed range in species tolerance (see Table 2). Note the $\log_{10}$ scale of the x axis. The species optimum for PO_4-P corresponds to the species maximum expected abundance.

transformed values are necessary to calculate the indices, they are not very meaningful for the reader. Tolerance values should also be reported as a range of actual values.

An example is given below (Table 2; Fig. 2), with data and models from Demars and Trémolières (2009). The expected abundance of three species of *Elodea* (aquatic vascular plants) from rivers along a PO_4-P concentration (water column) gradient was estimated with a Gaussian log-linear regression model.

In this letter I have drawn attention to the multi-facets of uncertainties: sampling errors (precision), causality (meaning) and predictability (reliability) in biotic indices. Poor communication may also introduce another layer of uncertainty. Uncertainties may only be ignored at the risk of later disappointing the users of biotic indices (see Demars and Edwards, 2009).

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