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Improving assessments of coastal ecosystems – Adjusting coastal fish indicators to variation in ambient environmental factors

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A B S T R A C T

The application of ecological indictors for assessing the environmental status of ecosystems play an important role for effective management. However, natural variability may limit the indicators' ability to provide relevant information about anthropogenic pressures and guide management action. Coastal fish species are not only a resource for commercial and recreational fisheries but also key ecosystem components in the Baltic Sea, and is therefore used as management objectives within the EU Marine Strategy Framework Directive and the HELCOM Baltic Sea Action Plan. A challenge, however, is that the distribution and abundance of coastal fish populations in Baltic Sea is also influenced by spatial and temporal variation in ambient environmental factors. Here, using 16 years of monitoring data, over a latitudinal range of 56 – 66°N along the Swedish Baltic Sea coast, we evaluated the effect of variability in water temperature and depth, and wave exposure for three indicators of environmental status assessment in the Baltic Sea: Abundance of perch, Abundance of Cyprinids, and Abundance of Piscivores. Generalized linear mixed models (GLMM) revealed an overall positive linear relationship between water temperature for all indicators, and overall negative linear relationships to depth and wave exposure. When adjusting indicator values using the parameter estimates from the GLMM models, the variability and 95 % confidence interval for all three indicators were reduced. The adjustment, however, did not have a strong impact on the assessment of the ecological state of the indicator. Our results suggest that adjusting coastal fish indicators to variation in local ambient environmental factors will increase their precision, and hence, the confidence in the assessment of environmental status.

1. Introduction

Application of ecological indictors for assessing the environmental status of ecosystems is important to provide guidance for effective management (Bergström et al., 2016b; Muñoz-Colmenares et al., 2021; Shin and Shannon, 2010; Östman et al., 2020). Biological indicators, however, may respond to variation in a wide range of abiotic and biotic environmental variables besides the anthropogenic pressures of primary interest for management (Asmamaw et al., 2021; Bergström et al., 2016a; Borja et al., 2010; Hao et al., 2021; Laurila-Pant et al., 2021; Östman et al., 2017a). Hence, natural variability may limit the indicators' ability to provide confident results and prevent accurate assessments and thereby influence management action (Bergström et al., 2016a; Bergström et al., 2016b; Östman et al., 2017a).

Coastal fish species are key ecosystem components in the Baltic Sea to support environmental status assessment in relation to the objectives within the EU Marine Strategy Framework Directive (MSFD) and the HELCOM Baltic Sea Action Plan (BSAP) (EC, 2008; HELCOM, 2021). Coastal fish species influences the food-web structure and ecological function (Östman et al., 2016). At the same time coastal fish are important for small scaled commercial and recreational fisheries (Hansson et al., 2018; Olsson, 2019). The species composition of coastal fish in the brackish Baltic Sea varies geographically along its salinity gradient (Koehler et al., 2022). However, several other factors can affect the relative abundances of different species of coastal fish (Olsson et al. 2012). In addition to anthropogenic pressures such as eutrophication, habitat deterioration and fishing, the distribution and abundance of local coastal fish assemblages in the Baltic Sea is also influenced by spatial and temporal variation in ambient environmental factors such as water temperature, wave exposure, habitat and depth (Bergström et al.,

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2016a; Bergström et al., 2019; Bergström et al., 2013; Kraufvelin et al., 2018; Linløkken and Haugen, 2006; Olsson et al., 2012; Östman et al., 2012).

With respect to temperature, many Baltic Sea coastal fish species are of freshwater origin (e.g. cyprinids, percids) and favored by higher water temperatures, whereas marine species typically are favored in coastal areas during cooler water temperatures (Olsson et al., 2012). Periods of higher temperatures therefore benefit the growth and reproduction of many coastal species within currently prevailing ranges (Böhling et al., 1991; Heikinheimo et al., 2014; Kjellman et al., 2001; Lappalainen et al., 1996). However, changes in water temperature also affects fish behavior, such as swimming speed and foraging activity (Bergström et al., 2016a; Johansen et al., 2015; Marchand et al., 2002; Östman et al., 2017b). This does not affect abundances of coastal fish directly but influences the catchability of fish in the multi-mesh gillnets used in the monitoring of coastal fish communities in the Baltic Sea (HELCOM, 2018). Thus, increasing water temperatures may exaggerate indicator values based on catch per unit effort of certain coastal fish species and potentially contribute to uncertainty of the true status. Also water depth and wave exposure might affect the small-scale spatial variation in coastal fish distribution and abundance, for example Eurasian perch (*Perca fluviatilis*) and many cyprinid species prefer shallower and more sheltered areas of the coastal zone of the Baltic Sea (Bergström et al., 2016a; Karås and Thoresson, 1992). Hence, even though depth and wave exposure do not change over time within a monitoring area (unless sampling stations are changed), they may inflict spatial variability which, if unaccounted, is difficult to separate from anthropogenic variation among areas.

Here, we evaluate the effects of temporal and spatial variability in water temperature and spatial variability in water depth and wave exposure on three indicators for the environmental status of coastal fish in the Baltic Sea: *Abundance of perch, Abundance of Cyprinids, and Abundance of Piscivores.* For the indicators, catch per unit effort (CPUE) is used as a proxy for "*Abundance*" of the target species in monitoring (HELCOM, 2018). Perch is a predominating piscivorous species in coastal areas of the Baltic Sea (HELCOM, 2018; Olsson, 2019) favored by increasing water temperatures and moderate levels of nutrient concentrations, as well as by vegetated habitats, while being disadvantaged by fishing and habitat degradation (Karås, 1996; Kjellman et al., 2001; Östman et al., 2017a). High abundances of cyprinids are indicative of eutrophic conditions (Bergström et al., 2016a; Eriksson et al., 2011; Östman et al., 2017a), but are also favored by increasing water temperatures and lack of top-down regulation (Härmä et al., 2008; Östman et al., 2016; Östman et al., 2017a). Viable populations of piscivorous species, compared to mesopredators such as cyprinids, are often indicative of an environmental status with few eutrophication symptoms and moderate exploitation(Eriksson et al., 2011; Östman et al., 2016).

We use time series covering 16 years of monitoring data from 11 areas ranging the latitudinal gradient along the Swedish coast of the Baltic Sea, to: i) estimate the quantitative relationship between coastal fish indicators and ambient environmental factors across time and over space; (ii) statistically adjust indicator values to the variability in ambient environmental factors, and (iii) evaluate how adjusted indicator values affect the assessment of environmental state using current

Fig. 1. Location of the sampling areas along the Swedish east coast.

assessment method compared to unadjusted data.

2. Methods

2.1. Data

The study was conducted using standardized monitoring data of coastal fish from eleven reference areas with low direct human impact (Bergström et al., 2016a) at the Baltic Sea coast covering a latitudinal gradient from 56 to 66°N (Fig. 1). Monitoring was done with Nordic coastal multimesh gillnets in August (when water temperature is the highest) (HELCOM, 2018) during 2002 – 2017 in eight areas (Forsmark, Holmön, Kvädöfjärden, Lagnö, Långvindsfjärden, Norrbyn, Råneå, Torhamn), 2004–2017 in two areas (Gaviksfjärden and Kinnbäcksfjärden), and 2005–2017 in one area (Asköfjärden) (Fig. 1). Nordic coastal multimesh gillnets are 1.8 m deep monofilament nets composed of nine 5 m panels with mesh sizes 10, 12, 15, 19, 24, 30, 38, 48 and 60 mm.

Each sampling area consists of 29–50 fixed sampling stations fished every year in a depth-stratified design over the 0–3, 3–6, and 6–10 m depth intervals, and where depth conditions allows, also 10 – 20 m (Bergström et al., 2016a). Each station was fished during one night, from late afternoon (3–6 pm) to the next morning (6–8 am). Catches were registered as numbers of individuals per species and cm-classes. Only fish>12 cm were included in the analyses, as inspection of catch curves (individuals per length interval in catch) showed that fish below this length were not sampled in a representative way. The indicator *Abundance of Perch* (hereafter "Perch") was calculated as the total number of perch caught per station and night (catch per unit effort; CPUE). The indicator *Abundance of Cyprinids* (hereafter "Cyprinids") was represented by the combined total catch of roach, white bream (*Abramis bjoerkna*), ide (*Leuciscus idus*), rudd (*Scardinius erythrophthalmus*), bleak (*Alburnus alburnus*), bream (*Abramis brama*), tench (*Tinca tinca*), crucian carp (*Carassius carassius*), vimba (*Vimba vimba*), and dace (*Leuciscus leuciscus*) per station and night. Last, the *Abundance of Piscivores* (hereafter "Piscivores") was calculated as the total catch of perch, pike (*Esox Lucius*), pikeperch (*Sander lucioperca*), and cod (*Gadus morhua*) per station and night.

For data on ambient environmental variables, water temperature was measured at the bottom of each station in connection to the fish surveys, together with information on sampling depth. Information on wave exposure for each station was derived from a digital sea chart using the Simplified Wave Exposure index (Isæus, 2004) which combines fetch calculations with wind conditions and also accounts for wave refraction and diffraction effects (Bergström et al., 2016a).

2.2. Data analyses

2.2.1. Objective i: Relation between coastal fish indicators and ambient abiotic variables

Collinearity of covariates were assessed using Variance Inflation Factor (VIF) statistic (Allison, 1999; Johnston et al., 2018). According to Johnston et al. 2018, VIFs of 2.5 or greater are generally considered indicative of considerable collinearity (Johnston et al., 2018). We then used generalized linear mixed models (GLMM) fitted by maximum likelihood estimation (Littell et al., 2006) to analyze the contribution of the ambient environmental variables as fixed effects on each of the three indicators "Perch", "Cyprinids", and "Piscivores" as response variable and with "area" as a random effect variable. To improve the fit of our linear mixed model we $\log_{10}(x + 1)$ – transformed the wave exposure data prior to analyses, and to meet the assumption of normally distributed residuals we $\log_{10}(x + 1)$ – transformed all the response (indicator) variables. Hence, the full model applied was:

$$\log_{10}(\text{indicator} + 1) = \beta_0 + \beta_1\,\text{Temperature} + \beta_2 \log_{10}(\text{Wave exposure}) + \beta_3\,\text{Depth} + U_i\,\text{Area}, \quad (1)$$

where β_0 is the model intercept and β_1-β_3 are specific coefficients of fixed abiotic variables and U_i is the random area - specific intercept.

2.2.2. Objective ii: Adjust indicators for ambient environmental factors

To derive estimates of adjusted indicator values, in which (natural) variation in the studied environmental variables are controlled for, we adjusted the observed indicator values according to the difference from mean environment condition among all monitoring stations:

Adjusted $\text{Log}_{10}(I_{adjusted} + 1) = \text{Log}_{10}(I_{observed} + 1) + \beta_1 \times (\text{Temperature}_{mean} - \text{Temperature}_{observed}) - \beta_2 \times (\log_{10}(\text{Wave exposure}_{mean}) / (\text{Wave exposure}_{observed})) - \beta_3 \times (\text{Depth}_{mean} - \text{Depth}_{observed})$ (eq. 2) where I is the "indicator".

Including all areas in the same model allowed us to combine information on environmental variables from the whole study area, and make the estimates from different monitoring sites comparable. To evaluate site-specific effects of these adjustments, we estimated the coefficient of variation ($CV = 100\%\sqrt{e^{[\ln(10)]^2\sigma^2} - 1}$) (Canchola et al., 2017) for both observed and adjusted log-transformed indicator values at each monitoring area and year. For both observed and adjusted log-transformed indicator values, we also estimated the 95 % confidence interval (CI) of the means for each area and year. Because the assumptions about homogeneity of variance were not met for neither the estimated CV nor CI (Levene's test, $p < 0{,}001$), we used a nonparametric Kruskal–Wallis test to statistically assess differences in the CV/CI of observed and adjusted data among areas. In addition, we performed a Mann-Whitney *U* test for each area to assess whether the CV or CI of indicator values differed significantly between observed and adjusted values within areas.

2.2.3. Objective iii: Effect of adjustment on indicator state

To evaluate if the adjustment of indicator values would affect the assessment of status change in each area, we carried out status assessments for data sets based on both observed and adjusted indicator values, respectively. This was done by applying the ASCETS (Analyses of Structural Changes in Ecological Time Series) – method (Östman et al., 2020), which can be used to assess changes in status of coastal fish communities (HELCOM, 2018). From indicator values (observed or adjusted, respectively) during a stationary phase of a reference period, ASCETS enables identification of quantitative boundary levels from pre-decided percentiles of bootstrapped distributions. Median indicator vales (observed or adjusted, respectively) from an assessment period can then be compared to the boundary levels to assess the changes in indicator states (Östman et al., 2020). Here we use indicator values during a stationary phase from a pre-defined reference period (2002/2004/2005–2011) relative to median indicator values during the assessment period of the last six years in each data set (2012–2017), in alignment with the most recent assessment cycle of the MSFD (Commission, 2017). We assesses changes in indicator states in each area from the 5th and 95th percentiles of the resampled indicator distributions from the reference period. Hence, if median indicator values from the six-year assessment period is outside the 5–95 percentile interval this is interpreted as a change in indicator state between reference and assessment periods. See Östman et al. (2020) for a detailed description of the ASCETS-methodology.

2.3. Results

2.3.1. Objective i: Relation between coastal fish indicators and ambient abiotic variables

The VIF values were 1.39 for temperature, 1.12 for wave exposure, and 1.40 for depth suggesting that there was weak collinearity among covariates. The GLMM for all areas combined showed an overall positive linear relationship between water temperature for all indicators ("Perch", "Cyprinids" and "Piscivores"), and an overall negative linear relationships to depth and wave exposure (Table 1, Fig. 2).

Table 1
Results of Generalized Linear Mixed Models to test for the effects of water temperature, wave exposure, and depth on the three coastal fish indicators included in this study.

Variable	Perch			Cyprinids			Piscivores		
	Estimate	Standard Error	*t*-value	Estimate	Standard Error	*t*-value	Estimate	Standard Error	*t*-value
Intercept	0.529	0.082	6.5***	0.438	0.101	4.3***	0.579	0.081	7.1***
Water temperature	0.056	0.002	31.1***	0.044	0.002	20.2***	0.056	0.002	31.0***
Log Wave exposure	−0.054	0.011	−4.7***	−0.051	0.014	−3.7***	−0.060	0.011	−5.3***
Depth	−0.029	0.001	−19.6***	−0.035	0.002	−19.6***	−0.028	0.001	−19.4***

*** $p < 0.001$.

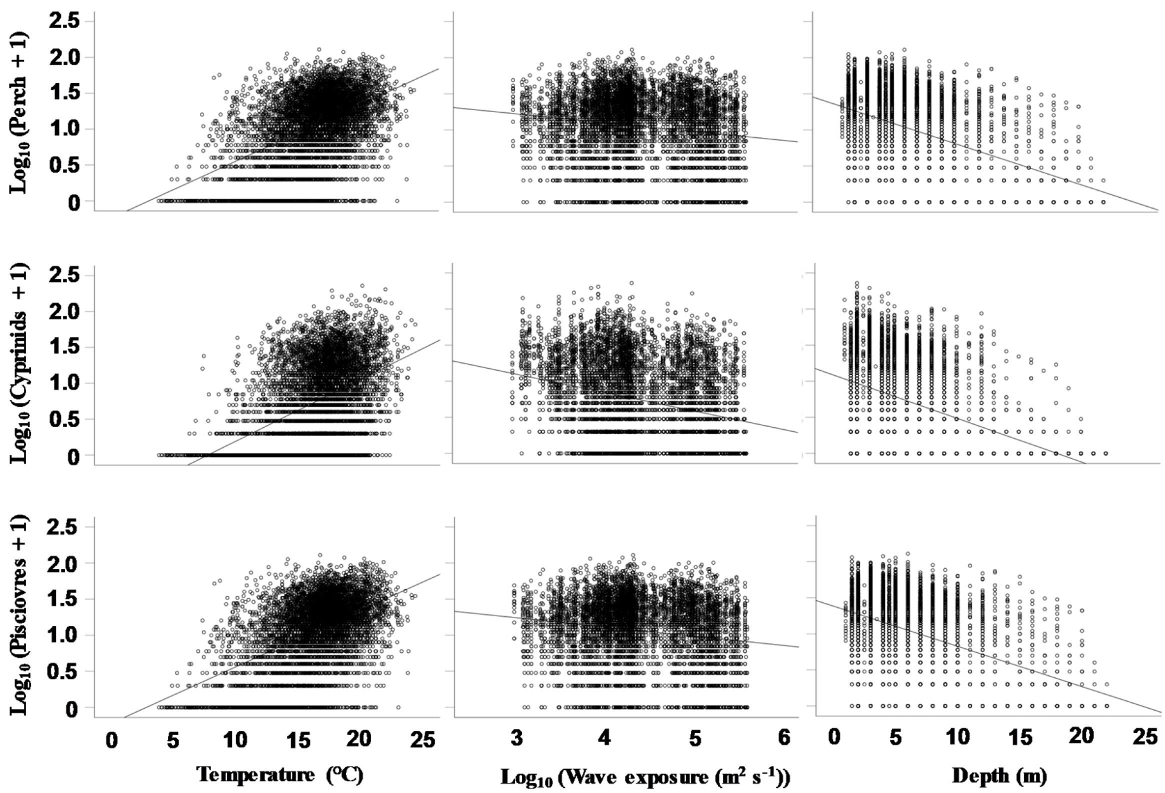

Fig. 2. Relation between ($\log_{10}$ (x + 1)-transformed) coastal fish-based indicators (Perch, Cyprinids, and Piscivores) and ambient environmental variables (temperature, wave exposure, and depth), based on monitoring data during 2002–2017, combined for all areas. Each dot is one station one year.

When addressing the relationships within each area, linear regressions also showed significant positive relationships between water temperature and the indicators in all 11 areas (Appendix 1). Water temperature at fishing ranged from 3.8℃ to 24.4℃, and the strength of the linear relationship (r) between the indicator values and temperature varied between 0.12 and 0.50 (Appendix 1). Indicator values were often

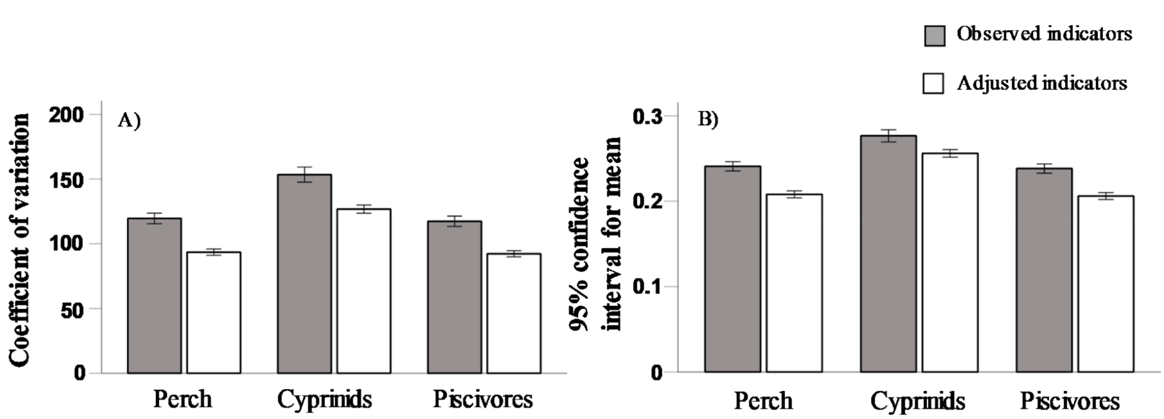

Fig. 3. Mean (±SE) of the coefficient of variation (A) and 95 % confidence interval for mean (B) of observed and adjusted log-transformed indicator values among study areas.

negatively related to water depth, although there were no significant relations between water depth and "Perch" in Kvädöfjärden and Råneå, "Cyprinids" in Torhamn, nor "Piscivores" in Råneå (Appendix 2). Moreover, there was a positive relation between "Cyprinids" and water depth in one area, Forsmark (Appendix 2b). Significant relations between indicator values and wave exposure were negative in most areas (Appendix 3). However, no relationship were observed between wave exposure and the "Perch" and "Piscivores" indicators in Gaviksfjärden, Lagnö, and Råneå, nor "Cyprinids" in Kvädöfjärden and Torhamn (Appendix 3).

2.3.2. Objective ii: A model to adjust the variability in indicators resulting from ambient abiotic variables

For all three indicators, adjusted values generally rendered lower CVs (Kruskal–Wallis test, $n = 338$ and $df = 1$ per indicator, all $p < 0.001$, Fig. 3) and narrower CIs (Kruskal–Wallis test, $n = 338$ and $df = 1$ per

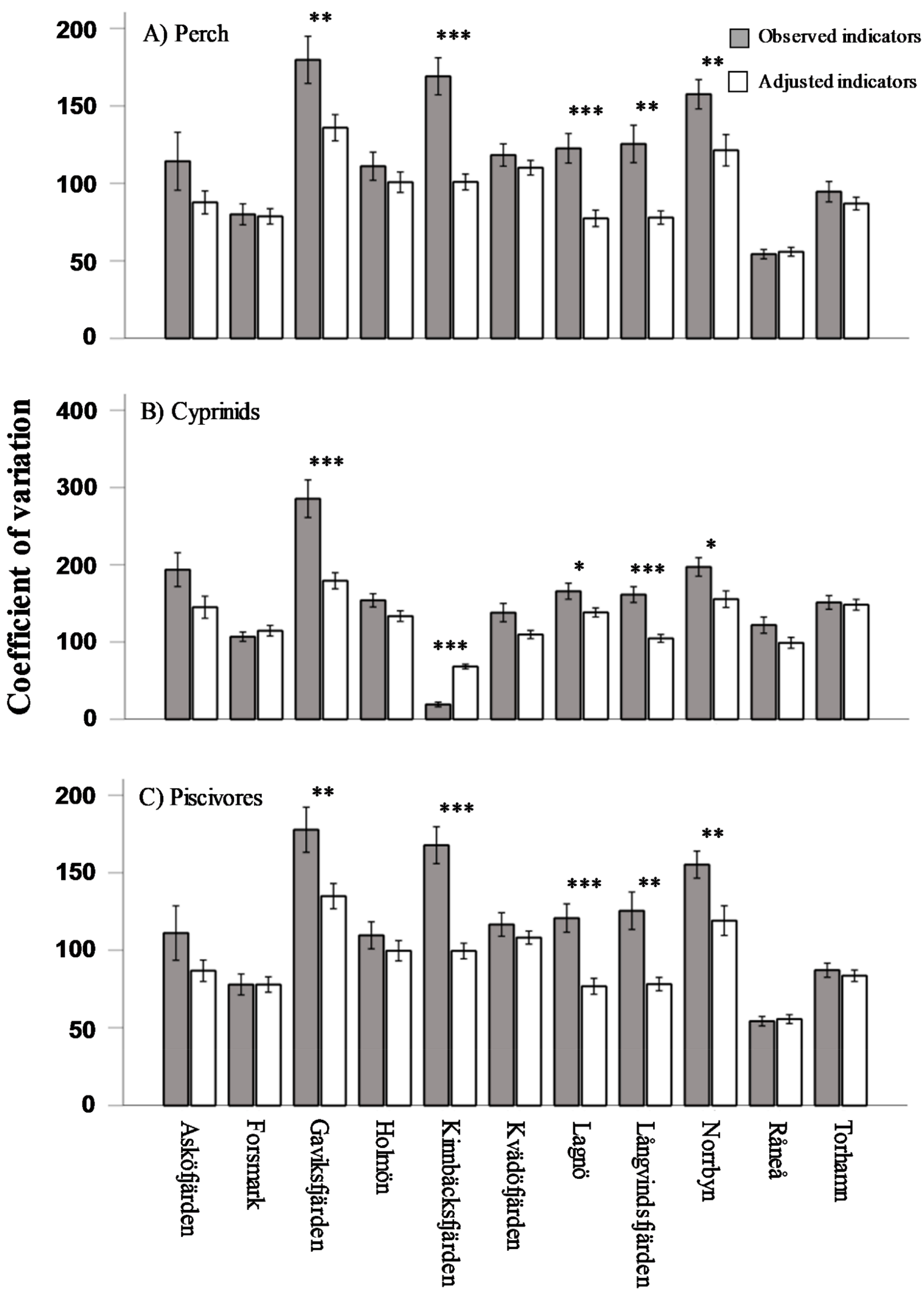

Fig. 4. Mean (±SE) of the coefficient of variation of observed and adjusted log-transformed indicator values in different monitoring areas. Significant differences between adjusted and unadjusted values are indicated by stars (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; Mann-Whitney *U* test). (n (years) per indicator: 26 in Asköfjärden, 28 in Gaviksfjärden, and 32 at each of the other areas).

indicator, all $p < 0.001$, Fig. 3) than observed values. 'Area' also had a significant effect on the CV (Kruskal–Wallis test, $n = 338$ and $df = 10$ per indicator, all $p < 0.001$) and CI of indicator values (Kruskal–Wallis test, $n = 338$ and $df = 10$ per indicator, all $p < 0.001$).

Within areas, CVs and CIs of adjusted indicator values were significantly lower than those of observed values in Gaviksfjärden, Kinnbäcksfjärden (only for perch and piscivores), Lagnö, Långvindsfjärden, Norrbyn (Mann-Whitney *U* test, Fig. 4 and Fig. 5). Adjustments had no significant effect on the CVs or CIs of indicators in Asköfjärden, Forsmark, Holmön, Råneå, Torhamn and Kvädöfjärden (Mann-Whitney *U* test, Fig. 4 and Fig. 5). In Kinnbäcksfjärden, the CV and the CI of cyprinids were significantly lower for observed than adjusted values (Mann-Whitney *U* test, $p < 0.001$, Fig. 4 and Fig. 5).

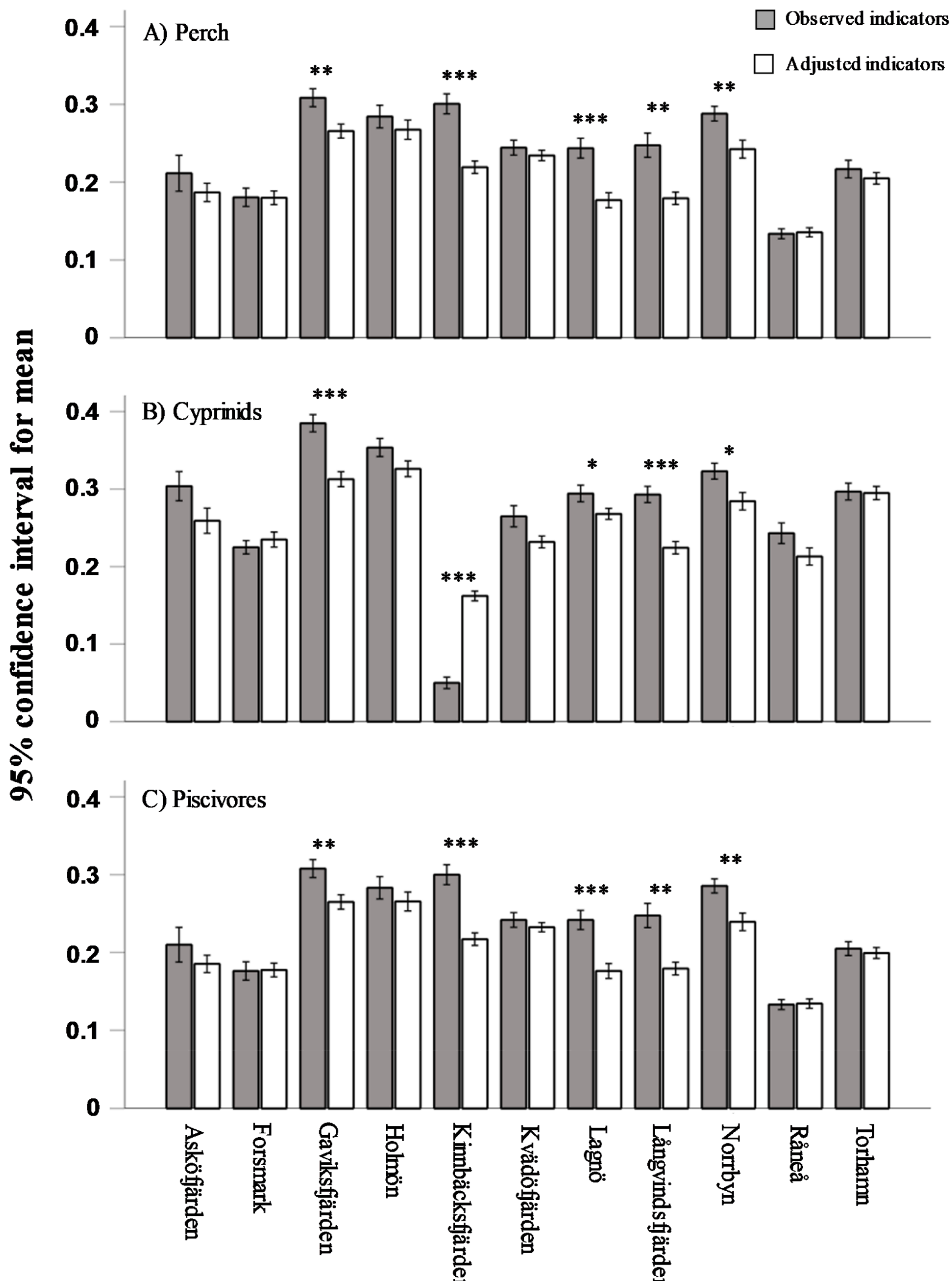

Fig. 5. Mean (±SE) of the 95 % confidence interval for mean of observed and adjusted log-transformed indicator values in different monitoring areas. Significant differences between adjusted and unadjusted values are indicated by stars (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; Mann-Whitney *U* test). (n (years) per indicator: 26 in Asköfjärden, 28 in Gaviksfjärden, and 32 at each of the other areas).

2.3.3. Objective iii: Effect of the adjustment on change in indicator state over time

In most cases, adjusting the indicator values resulted in changes in both the boundary levels for the reference period as well as the indicator distribution during the assessment period (Fig. 6). The adjustment however, did not have a strong impact on the assessment of change in ecological state of the indicator in each area using the ASCETS-methodology. For Cyprinids, boundaries were relatively similar between adjusted and original indicator values in all areas (Fig. 6). For Perch and Piscivores, the adjustment led to changes in boundary levels in four areas. In Forsmark, the adjusted indicator values of both Perch and Piscivores yielded a median value of the assessment period that was outside the boundary values of the reference period, whereas the unadjusted median value of the assessment period was found to be within the boundary values of the reference period (Confidence 0.13 vs 0.7; Fig. 6). In Lagnö, Långvind and Torhamn adjusted indicator values of Perch and Piscivores resulted in median values during the assessment period being within the boundary values of the reference period, whereas unadjusted indicator values resulted in median values being outside the boundary values of the reference period (Fig. 6). The adjustment substantially reduced the range between upper and lower boundary levels in some areas (mainly Forsmark, Holmön, Råneå, and Torhamn), but in Asköfjärden, the range between boundary values increased for Perch and Piscivores (Fig. 6).

3. Discussion

This study demonstrates that three coastal fish indicators of environmental status are influenced by ambient environmental factors and we propose adjusting indicators to consider this environmental variation to reduce overall variability and improve the reliability of the status assessments. Along the study gradient of the Swedish Baltic Sea coast, the observed values for all three indicators (Perch, Cyprinids, and Piscivores) increased with water temperature and decreased with water depth and wave exposure, in agreement with expectations based on previous studies (Bergström et al., 2016a; Olsson et al., 2012; Östman et al., 2017a).

To develop a methodological standard for indicator-based assessments of coastal fish in the Baltic Sea, we used a GLMM based on data for all monitoring areas (including "area" as a random factor) instead of running site-specific models for each area. On average, these "generic" adjustments lowered the CV within areas for the different indicators with around 17–22 %, and decreased the range of CI with on average 7–14 % for the different indicators. However, generic models are seldom fully consistent across areas, and here the reduction in variation was only marginal in some areas and in few cases, it even increased. Hence, site specific models would likely have had reduced variation but our models result in a potential general applicability of the proposed approach, and highlight the general utility of adjusting the coastal fish indicators for variation in ambient environmental factors to decrease temporal variability.

The lack of reduced variability in some areas (Forsmark, Råneå, Torhamn and Kvädöfjärden) were in most cases related to the absence of significant correlations between indicator values and the environmental variables within these monitoring areas. In Råneå, Kvädöfjärden, and Torhamn, one or all three indicators were not related to variability in depth and/or wave exposure (see Appendixes 2 and 3), and in Forsmark both "Perch" and "Piscivores" were positively related to wave exposure, rather than negatively as in most other areas (Appendix 3a, 3c). In Kinnbäcksfjärden, CVs and CIs of cyprinids were lower for the observed values than the adjusted values. The reason for this may be that Kinnbäcksfjärden is not a typical cyprinid area (Appendix 1b, 2b, 3b) and the variation in cyprinid catches is much higher in Kinnbäcksfjärden compared to all other areas (median relative standard error > 50 %; (Appelberg et al., 2020). In Asköfjärden and Holmön, there was a significant correlation between indicators and the environmental variables. The adjustment of indicator values reduced the CV by 22–25 % and 9–13 % and the CI by 12–15 % and 6–8 % in Asköfjärden and Holmön, respectively, but the effect was not statistically significant. Asköfjärden have been monitored three years less than other monitoring area so the lower statistical power at Asköfjärden may partly explain lack of significance. Taken together this suggests that there are other abiotic or biotic environmental variables, or internal dynamics like density dependence or cohort effect can influence the CPUE of these fish species, and further investigation would be needed to find the reasons for this inconsistency.

The adjustment of indicator values did not have any major impact on the change in indicator state over time, as assessed using the ASCETS approach (Östman et al., 2020). In the few cases where there was a difference between adjusted and non-adjusted values (Forsmark, Lagnö, Långvind and Torhamn), these did not show a consistent pattern; in some cases there was no change when using observed data but indicating a change when using adjusted data, but in more cases vice versa.

Whereas the adjustments only rendered minor changes in the resulting assessments of indicator state, a key effect appeared in that the upper boundary of the reference range was lowered when using adjusted compared to observed indicator values. This supports that adjusting indicator values for ambient environmental variables could be beneficial to increase comparability across areas, which are under influence of variability in local environmental conditions (Bergström et al., 2016a; Olsson et al., 2012; Östman et al., 2016; Östman et al., 2017a). However, other processes such as sampling error (stochastic processes) and internal dynamics like autocorrelation as well as variability in other locally important environmental factors such as wind condition and turbidity during fishing may also affect variability in the indicator values.

The ASCETS approach was used in this study due to some advantages over other alternative approaches in the applied setting. For instance, other fishery models such as Status-quo harvest control and Depletion-Corrected Average Catch (ICES, 2012) require catch data that was either not available (due to no fisheries data collection) or highly uncertain (e.g., from recreational fisheries) for the here assessed fish populations. Further, these methods are sensitive to both single observation errors and environmental stochasticity. In contrast, ASCETS focuses on longer time-series (preferably > 10 observations) and encompasses an array of natural and human induced drivers as well as observation errors, and only assumes that random processes are identical across the reference period (Östman et al., 2020). The 'Criteria A' approach used for categorizing species for the IUCN 'Red list', does not explicitly handle uncertainty and stochasticity, other than that referring to a data-deficient class (IUCN, 2001), whereas ASCETS handles uncertainty by resampling of data to provide confidence intervals (Östman et al., 2020). Compared to the TRIM used for assessments of birds (Soldaat et al., 2017), and designed for assessing statistical deviations of linear (linearized) long-term trends, ASCETS is designed to identify breakpoints in time-series and compare pre-defined reference and assessment periods (Östman et al., 2020). Compared to the approach of Greenstreet et al. (2012), which uses a large number of indicators and focuses on assessment at a single occasion (year) that require an aggregated assessment of several (>10) indicators (Greenstreet et al., 2012), ASCETS works better for longer assessment periods and is based on assessments of single indicator states that later can be aggregated (Östman et al., 2020).

4. Conclusion

Based on the results obtained in this study we propose that adjusting coastal fish indicators to local variation in ambient environmental factors will increase their precision, and hence, the confidence in the assessment of environmental status. Such adjustments are particularly useful to improve assessments against empirically derived reference values, as is the case for coastal fish in the Baltic Sea. We applied a

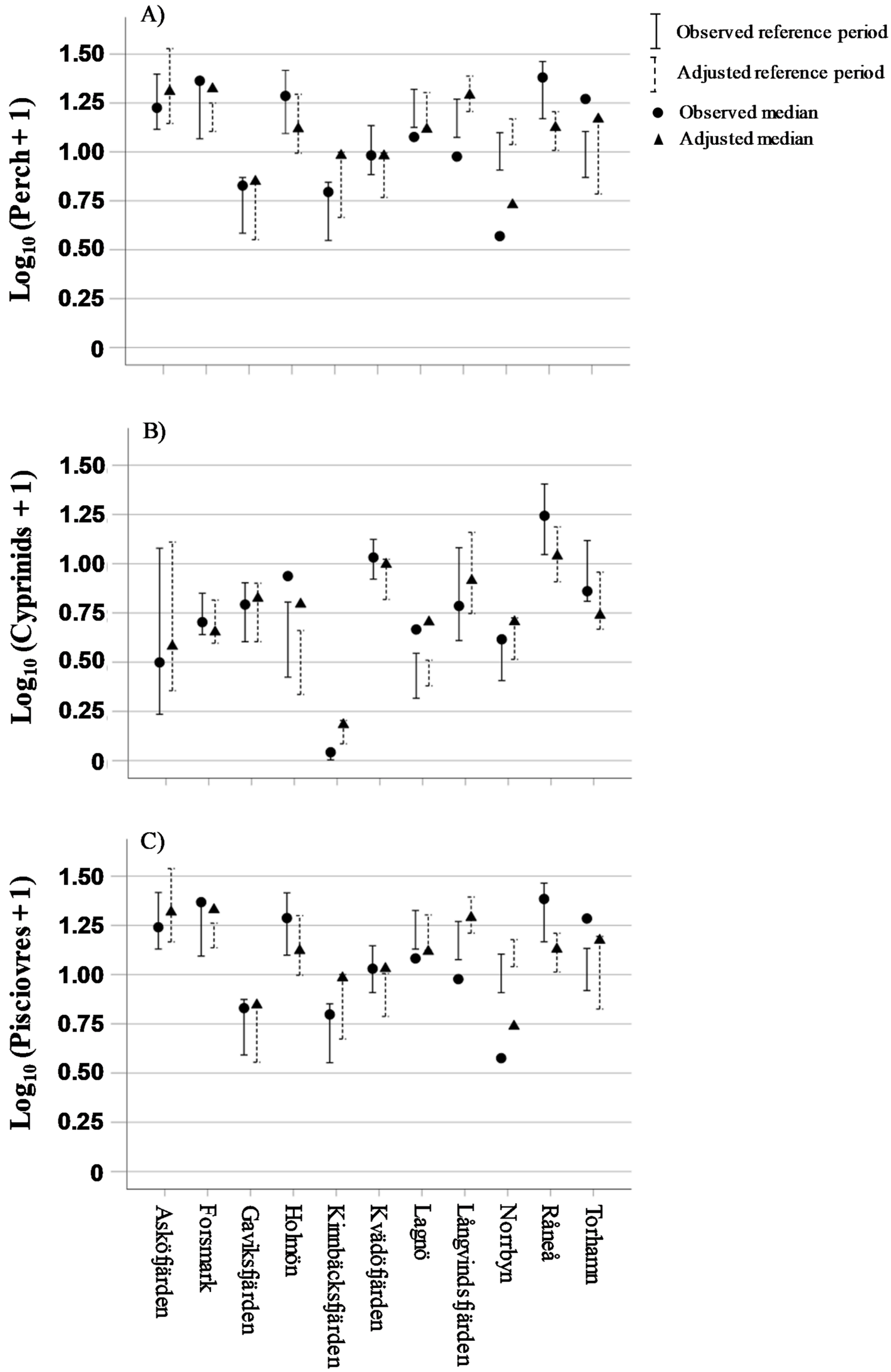

Fig. 6. Observed (dots) and adjusted (triangles) median values of log-transformed indicator values during the assessment period and boundary levels (error bars) from the reference period (solid and hatched for observed and adjusted, respectively) in different monitoring areas as derived from the ASCETS-methodology, for a) perch, b) cyprinids and c) piscivores. If the median values fall outside the boundary levels, it is indicative for a change over time in indicator state.

generalized function for all areas in this study, so the approach did not perform equally well in all cases. A prerequisite for motivating the adjustment appears that locally observed relationships between indicator and environmental variables are directionally similar to those in the generalized function. An alternative would to apply area-specific adjustments. However, whereas such approach might benefit to local assessment of the temporal development of the indicators, it does not support comparability of reference values across areas and excludes the possibility for assessments of shorter time-series. Facilitating comparisons across larger spatial scales is a central challenge in the assessment of local populations, such as coastal fish, which are under strong influence of local environmental conditions. To that end, we suggest to encompass the influence of ambient environmental variables on indicator-based evaluations, is a highly recommended for assessments that rely on environmental monitoring data.

CRediT authorship contribution statement

Rahmat Naddafi: Conceptualization, Data curation, Investigation, Writing – review & editing, Methodology, Writing – original draft, Formal analysis. **Örjan Östman:** Conceptualization, Writing – original draft, Methodology, Writing – review & editing, Formal analysis. **Lena Bergström:** Conceptualization, Data curation, Writing – review & editing, Writing – original draft, Visualization. **Noora Mustamäki:** Investigation, Validation, Writing – review & editing. **Magnus Appelberg:** Investigation, Visualization, Writing – review & editing. **Jens Olsson:** Conceptualization, Writing – review & editing, Writing – original draft, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The authors do not have permission to share data.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2022.109604.

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