Status of aquatic and riparian biodiversity in artificial lake ecosystems with and without management for recreational fisheries: implications for conservation

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Abstract

1. Humanity is facing a biodiversity crisis, with freshwater-associated biodiversity being in particularly dire state. Novel ecosystems created through human use of mineral resources, such as gravel pit lakes, can provide substitute habitats for conservation of freshwater and riparian biodiversity. However, many of these artificial ecosystems are managed for recreational fisheries and may exhibit high recreational use intensity, which may limit the biodiversity potential of gravel pit lakes.

2. We assessed the species richness of several taxa present in gravel pit lakes and compared a range of taxonomic biodiversity metrics of lakes managed for recreational fisheries (N = 16) and unmanaged reference lakes (N = 10), while controlling for non-fishing related environmental variation.

3. The average species richness of all examined taxa (plants, amphibians, dragonflies, damselflies, waterfowl, songbirds) was similar among both lake types and no faunal or floral breaks were revealed when examining the pooled species inventory of managed and unmanaged lakes. Similarly, there were no differences among management types in the presence of rare species and in the Simpson diversity index across all the taxa that we assessed.

4. Variation in species richness among lakes was correlated with woody habitat, lake morphology (size and steepness) and land use, but not with the presence of recreational fisheries. Thus, non-fishing related environmental variables have stronger effects on local species presence than recreational-fisheries management or the presence of recreational anglers.

5. Collectively, we found no evidence that anglers and recreational-fisheries management constrain the development of aquatic and riparian biodiversity in gravel pits of the study region.

Keywords

amphibians, biodiversity, birds, disturbance, fishing, lake, littoral, recreation, riparian, vegetation

1. Introduction

Globally, biodiversity is in steep decline, creating a biodiversity crisis of unprecedented scale (Díaz et al., 2019; WWF, 2018). An estimated 1 million species are currently threatened by extinction (Díaz et al., 2019), and current species extinction rates are about 1000 times higher than the calculated background rate (Pimm et al., 2014). The biodiversity decline is particularly prevalent in freshwaters compared to marine and terrestrial environments (Sala et al., 2000; WWF, 2018). From 1970 to today, freshwater biodiversity has, on average, declined by 83% across thousands of populations (WWF, 2018). Although manifold reasons contribute to the freshwater biodiversity crisis (Reid et al., 2019), habitat alteration and fragmentation, pollution, overexploitation, invasive species and climate change constitute key drivers (Dudgeon et al., 2006; IPBES, 2019).

Artificially created aquatic habitats, such as gravel pit lakes or ponds, can play an important role in maintaining and increasing native biodiversity by providing refuge and secondary habitat for rare or endangered species (Damnjanović et al., 2018; Santoul, Gaujard, Angélibert, Mastrorillo, & Céréghino, 2009). Artificial lake ecosystems are often relatively recent in origin (less than 100 years of age; Zhao, Grenouillet, Pool, Tudesque, & Cucherousset, 2016) and created by mining of sand, clay, gravel and other mineral resources (Saulnier-Talbot & Lavoie, 2018). More than one billion tons of sand, gravel and other earthen materials were excavated in more than 24,500 quarries and pits within the EU-28 in 2017 alone (UEPG, 2017). The resulting numerous artificial lakes (for simplicity henceforth referred to as gravel pit lakes) have become common elements in many cultural landscapes across the industrialized world (Blanchette & Lund, 2016).

Lakes, including gravel pit lakes, provide a bundle of ecosystem services to humans (Reynaud & Lanzanova, 2017). These include provisioning services, such as fish yield, as well as a range of cultural services, in particular recreation (Meyerhoff, Klefoth, & Arlinghaus, 2019; Venohr et al., 2018). Although the benefits of water-based recreation can be substantial, the activity can also negatively impact on the biodiversity of freshwater ecosystems (Venohr et al., 2018). For example, human activities can reduce littoral and riparian habitat quality and thereby negatively affect associated taxa (Spyra & Strzelec, 2019). Water-based recreation has also been found to negatively impact birds and other mobile wildlife through fear reactions to humans (Dear, Guay, Robinson, & Weston, 2015; Frid & Dill, 2002), dogs (Randler, 2006) or pleasure boats (McFadden, Herrera, & Navedo, 2017; Wolter & Arlinghaus, 2003). Management and conservation of gravel pit lakes and other artificial waterbodies would benefit from jointly considering the well-being aquatic recreation produces to humans, while balancing these benefits with the possible negative biodiversity impacts that aquatic recreation can induce (Lemmens et al., 2013; Lemmens, Mergeay, Van Wichelen, De Meester, & Declerck, 2015).

Many gravel pit lakes located in central Europe are used by recreational fisheries (Emmrich, Schälicke, Hühn, Lewin, & Arlinghaus, 2014; Matern et al., 2019; Zhao et al., 2016). Anglers are not only users, but in some regions of the world also managers of fish populations and habitats (Arlinghaus, Alós, et al., 2017). This particularly applies to Germany, where organizations of anglers, usually angling clubs and associations, are leaseholders or owners of freshwater fishing rights, and in this position are also legally entitled to manage fish stocks (Arlinghaus, Alós, et al., 2017). This entails the sovereignty to stock fish, to manage littoral habitat and introduce access and harvest regulations (Arlinghaus, Lorenzen, Johnson, Cooke, & Cowx, 2016). Because stocking of fish is particularly prevalent in freshwater recreational-fisheries management (Arlinghaus et al., 2015), key impacts of the presence of recreational fisheries and associated management activities can be expected at the fish stock and fish community levels (Matern et al., 2019; Zhao et al., 2016). Angler-induced changes of fish community composition and species richness typically include fostering elevated fish species richness through the release and introduction of large-bodied "game" fishes of high fisheries interest (Matern et al., 2019; Zhao et al., 2016). These changes might in turn affect submerged macrophytes, (e.g., due to introduction of benthivorous fish that uproot macrophytes; Bajer et al., 2016), and other taxa through predation (e.g., amphibians: Hecnar & M'Closkey, 1997; Miró, Sabás, & Ventura, 2018; invertebrates: Knorp & Dorn, 2016; Miller & Crowl, 2006; or birds: Cucherousset et al., 2012; Syväranta et al., 2009). In addition, anglers may modify littoral habitats through access to angling sites (Dustin & Vondracek, 2017), thereby affecting plant richness (O'Toole, Hanson, & Cooke, 2009), dragonflies (Z. Müller et al., 2003), or directly affect mobile taxa, such as birds, through angler-induced disturbances (Bell, Delany, Millett, & Pollitt, 1997; Cryer, Linley, Ward, Stratford, & Randerson, 1987). Indirectly, angler presence can also inadvertently kill non-targeted wildlife, e.g., through lost fishing gear that is ingested by birds or where birds become entangled (Franson et al., 2003; Sears, 1988). Therefore, anglers can both be seen as stewards of aquatic ecosystems (Granek et al., 2008) as well as a potential threat to certain aquatic taxa depending on the local angling intensity and other conditions (Reichholf, 1988; Venohr et al., 2018).

In Germany, fisheries (including recreational angling) are regulated by Federal state-specific fisheries law, while protection of species and habitat types is regulated by Federal and state-specific nature conservation legislation. Conflicts with angler interests regularly occurs when nature conservation authorities implement rules to achieve conservation goals that partially or fully constrain access to water bodies (Arlinghaus, 2005). Conservation-motivated constraints of angling or recreational-fisheries-management actions (e.g., stocking) are not only motivated by the implementation of national or international conservation law (e.g., Natura 2000 legislation of the European Community; 92/43/EWG, 1992) in natural ecosystems, but are increasingly commonplace in relation to artificial lake ecosystems. For example, in some regions of Germany recreational fisheries have been excluded from potential use of newly created gravel pit lakes already during the process of licensing the sand or gravel extraction (H. Müller, 2012). Such bans of the future angling use is typically justified by the assumption that angling is particularly impactful for disturbance-sensitive taxa (e.g., waterfowl) or habitats of special conservation concern (H. Müller, 2012; Reichholf, 1988; Wichmann, 2010).

To contribute to this ongoing conservation debate, here the taxonomic biodiversity present in and around gravel pit lakes was studied using a space-for-time substitution design comparing lakes managed and used by recreational fisheries with lakes that do not experience recreational fisheries actions and lack angler presence. Our study goal was to examine the aggregate impact of recreational fisheries on the taxonomic aquatic and riparian biodiversity detectable at typical gravel pit lakes in north-western Germany. The specific objective was to estimate the effect of recreational fisheries on species richness, faunal and floral composition, community diversity and conservation value across a range of aquatic and riparian taxa that are protected by national and European conservation legislation (e.g., birds, amphibians, dragonflies). Because the absence of recreational fisheries in a given gravel pit lake does not mean the ecosystems remain undisturbed from other recreational uses (e.g., swimming, walking), it was hypothesized that the presence of recreational fisheries and associated management activities will, on average, not affect species richness and conservation value of taxa that are not specifically targeted by anglers (Odonata, amphibians, submerged and riparian vegetation, waterfowl and songbirds). This hypothesis was formulated as a statistical null hypothesis to be falsified by empirical data.

2. Methods

Study area and lake selection

Our study was conducted in the Central Plain ecoregion of Lower Saxony in north-western Germany (Figure 1). Lower Saxony has 8 million inhabitants at a population density of 167 inhabitants per km^2. The total area of the Federal state encompasses 47,710 km^2, of which 27,753 km^2 constitute agricultural farmland and 10,245 km^2 managed forests (Landesamt für Statistik Niedersachsen (LSN), 2018). Natural lentic waters are scarce. In fact, of 35,048 ha total standing waters in Lower Saxony 73 % by area and more than 99 % by number are artificial lakes, mainly ponds and small gravel pit lakes with less than 10 ha surface area (Manfrin et al., unpublished data).

Most gravel pit lakes in Lower Saxony, and Germany as a whole, are managed for recreational fisheries by angler associations and clubs. These lakes are thus exposed to regular stocking with species of fisheries interests and experience access and harvest rules, regular controls by fisheries inspectors, and fishing club activities like collecting litter and cleaning and development of the littoral zone (Theis, 2016). Similar activities are absent in gravel pit lakes not used for recreational fisheries, which are much rarer in number but still occur in Lower Saxony and elsewhere across Germany. For our study, we selected a set of gravel pit lakes managed by recreational fisheries (defined as managed lakes) and another set of lakes not experiencing any form of legal angling and recreational fishing-related management (defined as unmanaged lakes, Table 1).

We identified managed lakes through a survey of all angling clubs organized in the Angler Association of Lower Saxony. Lakes were selected according to the following criteria. The lake shall be in ownership of a fishing club, of small size (1-20 ha) and with no dredging in the last ten years ("old age"). This approach yielded N = 16 managed lakes as study sites spread across Lower Saxony in 10 angling clubs (Table 1, Figure 1).The angler density (number of anglers per unit water area) for these clubs ranged from 8 to 43 anglers per hectare (mean ± SE: 21 ± 3.6 anglers per ha). These average angler densities correspond to averages known for German and Lower Saxonian angling clubs with 24 ± 2.5 and 22 ± 10.8 anglers per ha, respectively. All selected angler-managed lakes experienced regular angling activities and fisheries-management actions, including annual stocking of a range of fish species and regular shoreline development activities.

Gravel pits not managed by and for anglers were identified in close vicinity to the managed lakes (Figure 1). The number of unmanaged lakes available was much smaller than the number of managed lakes. Overall, we identified N = 10 unmanaged lakes that were of similar age, size and other environmental conditions to the managed lakes, but differed from the managed ones by the absence of an angling club and any form of legal angling and fisheries-management for at least 5 years prior to the onset of our study (Table 1, Figure 1). Both lakes types were accessible to non-angling recreation as they were not fenced.

In a subset of the selected lakes, Matern et al. (2019) previously conducted fish faunistic surveys revealing identical fish abundances and biomasses in both lakes types, but larger local fish species richness and significantly more abundant game fishes (particularly predators and large-bodied cyprinids such as carp, *Cyprinus carpio*) in managed lakes compared to unmanaged ones. These data showed that the angler-managed lakes included in our study were indeed more intensively managed in terms of fish stocking and hosted a substantially different fish community. This finding was a relevant precondition of the study design to show that managed and unmanaged lakes particularly differed in traces left by fisheries-management and fisheries use both in terms of fish community composition and angler presence in the littoral zone.

Despite our attempt to select lakes as similar as possible in terms of the environment (e.g., age, size, trophic state), just differentiated by the presence or not of recreational fisheries, we statistically controlled for possible confounding factors affecting the taxonomic biodiversity at both lake types. To that end, a set of environmental variables were assessed and integrated into the statistical analyses to more carefully isolate the possible impact of recreational-fisheries management on biodiversity, while controlling for other key environmental differences among lakes that could also affect the community composition of specific taxa (e.g., morphometry, land use). We in turn describe the assessment of the various environmental indicators.

Land use

Several indicators of land use and spatial arrangement in Lower Saxony across catchments were assessed to account for potential land use effects on the studied aquatic and riparian taxa. Shortest-path distances of lakes to nearby cities, villages, lakes, canals and rivers were calculated in Google Maps (© 2017). Subsequently, a share of different land use categories within 100 m around each lake (buffer zone) was calculated in QGIS 3.4.1 with GRASS 7.4.2 using ATKIS® land use data with a 10 x 10 meter grid scale (© GeoBasis-DE/BKG 2013; AdV, 2006). The ATKIS®-object categories were merged to seven land use classes: (1) urban (all anthropogenic infrastructures like buildings, streets, railroad tracks etc.), (2) agriculture (all arable land like fields and orchards but not meadows or pastures), (3) forest, (4) wetland (e.g., swamp lands, fen, peat lands), (5) excavation (e.g., open pit mine), (6) water (e.g., lakes, rivers, canals) and (7) other (not fitting in previous classes like succession areas, grass land, boulder sites etc.).

Recreational use intensity

The lake-specific recreational use intensity was assessed by counting the type and number of recreationists during each site visit (between six and nine visits per lake, see biodiversity sampling below). Indirect use intensity metrics encompassed measures of accessibility and litter, which were assessed as follows: the pathways around every lake were enumerated with a measuring wheel (NESTLE-Cross-country Model 12015001, 2 m circumference, 0.1% accuracy), measuring the length of all trails and paths at each lake. These variables were summed and normalized to shoreline length. Angling sites and other open spaces accessible to other recreationists (e.g., swimmers) along the shoreline were counted, and all litter encountered along paths and sites was counted and assigned to (1) angling-related (e.g., lead weight, nylon line, artificial bait remains) and (2) other litter not directly angling-related (e.g., plastic packaging, beer bottles, cigarette butts). More intensively used lakes were expected receiving larger amount of litter and being more easily accessible through paths and trampled sites, which could affect biodiversity negatively.

Age and morphology

The age of each lake was assessed through records in the angling clubs and by interviewing owners of lakes and regional administration or municipalities. Bathymetry and the size of each lake was mapped with a SIMRAD NSS7 evo2 echo sounder paired with a Lawrence TotalScan transducer mounted on a boat driving at 3 – 4 km/h along transects spaced at 25-45 m depending on lake size and depth. The data were processed using BioBase (Navico), and the post-processed data (depth and gps-position per ping) were used to calculate depth contour maps using ordinary kriging with the gstat-package in R (Gräler, Pebesma, & Heuvelink, 2016; R Core Team, 2013). Maximum depth and relative depth ratio (Damnjanović et al., 2018) were extracted from the contour maps. Shoreline length and lake area were estimated in QGIS 3.4.1 and used to calculate the shoreline development factor (Osgood, 2005).

Water chemistry and nutrient levels

During spring overturn, epilimnic water samples were taken for analyzing total phosphorus concentrations (TP), total organic carbon (TOC), ammonium and nitrate concentrations (NH_4, NO_3) and chlorophyll a (Chl-a) as a measure of algal biomass. TP was determined using the ammonium molybdate spectrophotometric method (EN ISO 6878, 2004; Murphy & Riley, 1962), TOC was determined with a nondispersive infrared detector (NDIR) after combustion (DIN EN 1484, 1997), ammonium and nitrate were assessed using the spectrometric continuous flow analysis (DIN EN ISO 13395, 1996; EN ISO 11732, 2005), and Chl-a was quantified using high performance liquid chromatography (HPLC), where the phaeopigments (degradation products) were separated from the intact chlorophyll a and only the concentration of the latter was measured (Mantoura & Llewellyn, 1983; Wright, 1991). Also during spring overturn, the lake's conductivity and pH were measured in epilimnic water with a WTW Multi 350i sensor probe (WTW GmbH, Weilheim, Germany), and turbidity was assessed using a standard Secchi-disk.

Littoral and riparian habitat assessment

Riparian structures and littoral dead wood was assessed using a plot design inspired by Kaufmann & Whittier (1997). Each plot consisted of a 15 x 4 meter riparian sub-plot, a 1 x 4 meter shoreline band, and a 4-meter-wide littoral transect extending into the lake to a maximum of ten meters or a water depth of three meters. At each lake the position of the first plot was randomly selected and subsequent plots placed every 100 (or 150 meters for larger lakes) apart along the shoreline until the lake was surrounded, resulting in 4-20 plots per lake (depending on lake size). In each riparian sub-plot and shoreline band, all plant structures (e.g., trees, tall herbs, reed) were assessed following the protocol of Kaufmann & Whittier (1997) as 0 - absent, 1 - sparse (< 10 % coverage), 2 - moderate (10-39 % coverage), 3 - dominant (40-75 % coverage), and 4 - very dominant (> 75 % coverage). In each littoral transect all dead wood was counted, and length and bulk diameters measured. Additionally, the width and height of each coarse woody structure was assessed, and each piece assigned to either (1) simple dead wood (bulk diameter < 5 cm and length < 50 cm, no or very low complexity), or (2) coarse woody structure (bulk diameter > 5 cm and/or length > 50, any degree of complexity) following the criteria of DeBoom & Wahl (2013). Further, for each dead wood structure the volume was calculated using the formula for a cylinder for simple dead wood and for an ellipsoid for coarse woody structure.

Riparian plant species

All lakes were sampled for riparian plant species at four transects (one per cardinal direction) in May. Each transect was 100 m long and contained five evenly spaced (20 m distance) 1 m^2-plots. Along the transects trees (>2 m high) were identified using Spohn, Golte-Bechtle, & Spohn (2015) and counted. Within each sampling plot, riparian vascular plants (<2 m high) were identified following the same key (Spohn et al., 2015) and their abundance assessed following Braun-Blanquet (1964): “r” = 1 individual, “+” = 2 – 5 individuals but < 5 % coverage, “1” = 6 – 50 individuals but < 5 % coverage, “2m” = > 50 individuals but < 5 % coverage, “2a” = 5 – 15 % coverage, “2b” = 16 – 25 % coverage, “3” = 26 – 50 % coverage, “4” = 51 – 75 % coverage, “5” = 76 – 100 %. The regional species pool was estimated from the Red Lists of Lower Saxony (Garve, 2004) in combination with their expected occurrence according to habitat type and species habitat preferences.

Submerged macrophytes

All lakes were sampled for submerged macrophytes between late June and late August, following the sampling protocol of Schaumburg et al. (2014). Every lake was scuba dived and snorkeled along transects set perpendicular to the shoreline from the bank (depth = 0 m) to the middle of the lake until the deepest point of macrophyte growth was reached. The position of the first transect was randomly chosen and all other transects spaced evenly along the shoreline at 80-150 m distances depending on lake size, resulting in 4-20 transects sampled per lake. Along each transect, in every depth stratum (0-1 m, 1-2 m, 2-4 m, 4-6 m) the dominance of submerged macrophyte species was visually estimated following the Kohler scale: "0 – absent", "1 – very rare", "2 – rare", "3 – widespread", "4 – common", "5 – very common" (Kohler, 1978). No macrophytes were found below 6 m depth. Macrophytes were identified under water according to Van de Weyer & Schmitt (2011). If this was not possible, samples were taken and identified under a stereomicroscope following Van de Weyer & Schmitt (2011). Stonewort species were only identified to the genus level (Chara spec. or Nitella spec.), thus exact species numbers might be underestimated. Macrophytes dominance was transformed to percent coverage for each transect (Van der Maarel, 1979). The average coverage per stratum was extrapolated to the total lake using the contour maps. The total macrophyte coverage in the littoral zone was calculated using the extrapolated coverage from strata between 0 m and 3 m depth. The regional species pool was estimated from the Red Lists of Lower Saxony in combination with the expected species for gravel pit lakes following the list of plant species associations in Lower Saxony (Garve, 2004; Korsch, Doege, Raabe, & van de Weyer, 2013; Preising et al., 1990).

Amphibians

Amphibians were sampled during the mating-seasons (from March to May). Every lake was sampled twice: (1) during the day with an inflatable boat driving slowly along the shore searching for adults, egg-balls (frogs) and egg-lines (toads), (2) after sunset by feet around the lake searching for calling adults. Each observation (adult or eggs) was marked with a GPS (Garmin Oregon 600), identified in the field or photographed for later identification following Schlüpmann (2005). Numbers were recorded (adults) or estimated (eggs), assuming 700 to 1500 eggs per egg-ball (frogs) or 10,000 eggs per (100 % covered) m^2 of egg-line-assemblages (toads). The egg numbers were calculated from pictures taken in the field and verified with literature (Trochet et al., 2014). The regional species pool was estimated from the Red List of Lower Saxony in combination with their expected distribution (Podloucky & Fischer, 2013).

Odonata

Dragonflies and damselflies were sampled once per lake between early- and mid-summer. At each lake, the whole shoreline was intensively searched during mid-day. Sitting or flushing imagines were caught with a hand net (butterfly net, 0.2 mm mesh size, bioform), identified using Lehmann & Nüss (2015), and released without being harmed. The regional species pool was estimated from the Red List of Lower Saxony in combination with their expected habitat preferences (Altmüller & Clausnitzer, 2010; Hein, 2018).

Waterfowl and songbirds

Waterfowl were identified following Dierschke (2016), counted and protocoled at every visit (between six and nine visits per lake). Songbirds were sampled once per lake between early- and mid-summer using a point-count sampling combined with a bioacoustics approach which was also used in other studies (Rempel, Hobson, Holborn, Van Wilgenburg, & Elliott, 2005; Wilson, Barr, & Zagorski, 2017). We took 2-minutes audio-recordings (ZOOM Handy Recorder H2, Surround 4-Channel setting, 44.1kHz sampling frequency, 16 bit quantification) at sampling points placed 200 m apart around the whole lake, assuming each sampling point covers a radius of 100 m. Sampling points were marked with GPS. At each point all birds seen or heard were protocoled when identified following Dierschke (2016). The audio-records were analyzed in the lab, and singing species were identified using reference audio samples (www.deutsche-vogelstimmen.de; www.vogelstimmen-wehr.de) and a birdsong-identifying software (BirdUp - Automatic Birdsong Recognition, developed by Jonathan Burn, Version 2018). The regional species pools for waterfowl and songbirds were estimated from the Red List of Lower Saxony (e.g. waders; Krüger & Nipkow, 2015) in combination with their expected occurrence according to habitat type and preferences.

Diversity metrics

We focused on the analysis of species presence-absence data to arrive at measures of taxonomic species richness to serve as an aggregate index of species diversity. Additionally, using relative abundance data by species we computed the Simpson diversity index (Pielou, 1969) to consider the dominance of certain species within the taxa-specific community. We were not interested in whether a particular species detected actually recruits in a given gravel pit lake, rather we were interested in the species inventory present, assuming our estimates represented a minimal estimate of local richness as rare species likely remained undetected. To weigh rare and threatened species more, we additionally computed the richness of threatened species and estimated an index of taxon-specific conservation value for the study region following Oertli et al. (2002). To that end, each species was ranked according to its threat status on the Red Lists of Lower Saxony (Altmüller & Clausnitzer, 2010; Garve, 2004; Korsch et al., 2013; Krüger & Nipkow, 2015; Podloucky & Fischer, 2013). Species of least concern were ranked lowest ($c(0) = 2^0 = 1$). All species classified with an increasing threat status category r according to the regional Red List were weighted exponentially more strongly as $c(r) = 2^r$ (Table 2) following Oertli et al. (2002)**Error! Reference source not found.**. For each lake, the final taxon-specific conservation value (CV) was calculated as the sum of all values for the observed species s_i ($s_1, s_2, s_3, \dots, s_n$) divided by the total number of species (n) for a given taxon:

$$CV = \frac{1}{n} * \sum_{s_{i=1}}^{s_n} c(r_{s_i}).$$

The conservation index value increases with more species of a given taxon being threatened or rare. We tested a range of different allocations of threat status to estimate the conservation value, using also national and European red lists; however, the results remained robust. For space reasons, we just report the regional index here.

Finally, to test for differences in species composition across all lakes, the pooled species inventory by lake type (managed and unmanaged) was used and the Sørensen index (Sørensen, 1948) as a measure of community similarity was calculated. The Sørensen index ranges from 0 (no species in common among the two lakes types) to 1 (all species the same) and is calculated as $\frac{2a}{2a+b+c}$, with a being the number of shared species and b and c being the numbers of unique species to each lake type, respectively. Following Matthews (1986), faunal or floral breaks, i.e. substantial (i.e., biologically meaningful) differences in species composition, among lake types were assumed to occur when the Sørensen index was < 0.5.

Statistical analysis

The impact of the presence of recreational-fisheries management on aquatic and riparian biodiversity was tested in two steps.

First, differences in taxon-specific species richness, Simpson diversity index, richness of threatened species, conservation value and as well as key environmental variables between lake types (managed and unmanaged gravel pits) were assessed with univariate statistics. To that end, mean differences among lake types were tested using Student's t (in case of variance homogeneity) or Welch-F (in case of variance heterogeneity) whenever the error term was normally distributed (Shapiro-Wilk-test). Otherwise, a Mann-Whitney-U-test of median differences was used. P-values were Sidak-corrected (Šidák, 1967) for multiple comparisons. Significance was assessed at $p < 0.05$.

Second, we modelled the among-lake variation in species richness as a function of management type and a set of lake-specific environmental descriptors. These analyses aimed at further isolating an impact of fisheries management and type of recreational uses on species inventory across all taxa and lakes in a joint model that included other predictor variables of the lake environment. To reduce the dimensionality of the environmental variables, a Principal Component Analyses (PCA) was conducted without rotations (PCA; Mardia, Kent, & Bibby, 1979) in "classes of environmental variables" (e.g., morphology, productivity, habitat structure, land use, recreational use). Environmental variables forming Principal Components (PC) were considered correlated, their loadings identified, the axis meanings interpreted, and the PC scores used as indicator variables. We then conducted multivariate Redundancy Analyses (RDA; Legendre & Legendre, 2012) to examine if recreational fisheries management explains variation in either environmental variables or species richness across multiple taxa in the multivariate space. In addition to management type, all relevant environmental variables (e.g., trophic state, lake size/steepness, land use, riparian/littoral habitat structure, water chemistry), recreational use intensity, gravel pit age, and catchment were included in the multivariate analysis of species richness. With the RDA, we used a forward selection process (Blanchet, Legendre, & Borcard, 2008) to identify the most explaining environmental predictors of species richness across different taxa and lakes, including management as a key variable of interest in this study. Using the variance inflation factor (VIF; Neter, Kutner, Nachtsheim, & Wasserman, 1996) correlated environmental variables were removed before model building. All data were scaled and centered (z-transformation) prior to analyses. The degree of explanation was expressed using the adjusted coefficient of multiple determination ($R^2_{adj.}$). Variables significantly explaining variation in richness across lakes were also assessed using ANOVA (Analysis of variance, Chambers & Hastie, 1992) at a significance level of $p < 0.05$. All calculations and analyses were carried out in R using the vegan-package (Oksanen et al., 2018; R Core Team, 2013).

3. Results

Description of lake types in relation to the environment

The studied lakes were, on average, small (mean ± SD, area 6.5 ± 5.2 ha, range 0.9 – 19.5 ha), shallow (maximum depth 9.6 ± 5.2 m, range 1.1 – 23.5 m) and mesotrohic (TP 26.3 ± 30.9 µg/l, range 8 - 160 µg/l) with moderate visibility (Secchi depth 2.4 ± 1.4 m, range 0.5 – 5.5 m) (Table 3). The land use in a 100 m buffer around the lake was, on average, characterized by low degree of forestation (mean 16 ± 21 %, range 0 – 72.6 %) and high degree of agricultural land use (mean 27 ± 22 %, range 2.4 – 79 %). Lakes were, on average, situated close to both human settlements (mean distance to the next village 618.3 ± 533.4 m, range 20 – 1810 m) and other water bodies (mean distance to next lake, river, or canal 55.8 ± 84.7 m, range 1 – 305 m). Gravel pit lakes were all in an advanced stage of succession and on average 27.3 ± 13.3 years old (range 6 – 54 years, see Tables S1-S4 for detailed lake-specific environmental variables). The study lakes belong to four different watersheds (small North Sea tributaries and the catchments of the rivers Ems, Weser and Elbe; Table 1, Figure 1).

Environmental characteristics of managed and unmanaged gravel pit lakes

Both lake types did not statistically differ in age, size, trophic state, and land use (Table 3). A similar result was revealed in a multivariate RDA, which confirmed the absence of significant differences between managed and unmanaged lakes in "classes of environmental variables" (i.e., PC scores, for details see Tables S5, S6) representing morphology (an index of steepness and water body size; $R^2_{adj.} = -0.005$, $F = 0.86$, $p = 0.470$), trophic state ($R^2_{adj.} = -0.006$, $F = 0.86$, $p = 0.544$), proximity to alternative water bodies ($R^2_{adj.} = -0.023$, $F = 0.45$, $p = 0.867$), proximity to human presence ($R^2_{adj.} = 0.035$, $F = 1.90$, $p = 0.143$) and land use variables ($R^2_{adj.} = 0.033$, $F = 1.85$, $p = 0.135$). However, in multivariate space the habitat structure differed significantly among managed and unmanaged lakes along the first PC axis (Dim 1), which represented a vegetation gradient below and above water (Figure 2). Along this axis, managed lakes were found to be more vegetated than unmanaged ones in both the riparian and the littoral zones ($R^2_{adj.} = 0.056$, $F = 2.48$, $p = 0.022$).

Recreational uses of managed and unmanaged lakes

The two lake types differed strongly in terms of recreational use intensity, particularly in relation to the observed angling intensity. As intended by our study design, managed lakes revealed, on average, significantly higher angling use intensity indexed by a diverse set of variables like angling litter density, extension of open sites, paths and trails and number of anglers observed (Table 4). By contrast, the average recreational use intensity of managed and unmanaged lakes by non-angling recreationists (e.g., swimmers) did not statistically differ when analyzed by univariate statistics on a variable-by-variable level (Table 4). However, when all indicator variables of the recreational use, both angling and non-angling, were combined in a multivariate RDA analysis as a function of management type, managed lakes separated from unmanaged lakes along PC axis 1. This axis represented differences in recreational use intensity by both anglers and other recreationists (particularly swimmers) and extension of trails and paths (Figure 3, $R^2_{adj.} = 0.16$, $F = 5.76$, $p < 0.001$). Note that there was no differentiation among lake types along the second PC axis of the recreational variables (Figure 3), which represented shoreline inaccessibility.

Species diversity and taxon-specific conservation value in managed and unmanaged gravel pit lakes

In total 41 submerged macrophytes species were detected, 191 riparian vascular plants, 44 trees, 3 amphibians, 33 Odonata, 36 songbirds and 34 waterfowl species. This species inventory represented a substantial fraction of the regional species pool of trees (59 %), Odonata (56 %) and waterfowl (45 %). By contrast, only one third or less of the regional species pool of vascular plant species (12 %), submerged macrophytes (33 %), songbirds (33 %) and amphibians (38 %) was detected. Only few species non-native to Lower Saxony or Germany were found: 4 submerged macrophyte species (e.g., *Elodea nuttallii*, [Planch.] H. St. John, which is invasive), 4 riparian tree species, 2 waterfowl species (e.g., *Alopochen aegyptiaca*, L., which is invasive), 1 riparian vascular plant species, and 1 dragonfly species.

Based on the pooled species inventories (gamma diversity), unique species (i.e. species present in only one lake or only one lake type) were found in all taxa except amphibians (Table 5). Managed lakes hosted more unique species within most taxa than unmanaged lakes, while unmanaged lakes had more unique Odonata. No faunal or floral breaks were detected between managed and unmanaged lakes using the Sørensen index (all indices ≥ 0.5; Table 5). The average taxon-specific species richness (alpha-diversity), the Simpson diversity index, the average number of threatened species and the average taxon-specific conservation value were statistically similar in managed and unmanaged lakes across all taxa when analyzed using univariate statistics (Table **6**).

Environmental correlates of among-lake variation in species richness

Across lakes, species richness of amphibians, Odonata, songbirds and riparian vascular plant species covaried along the first axis (Figure 4), collectively representing riparian diversity (for full PCA results, see Table S8). The second PCA axis represented mainly submerged macrophytes (Figure 4). The third axis was related to the diversity of riparian tree species and the forth mainly to waterfowl diversity (Figure 5). Therefore, lakes offering high riparian species richness were not necessarily rich in in-lake biodiversity (represented by submerged macrophytes and waterfowl) or tree biodiversity. The RDA analysis to explain the among-lake variation in species richness as a function of management type alone revealed no influence of this factor on among-lake richness across several taxa (RDA, $R^2_{adj.} = 0.028$, $F = 1.73$, $p = 0.114$).

All environmental variables subsumed by PC-scores into environmental predictors and lake age had acceptable inflation factors (VIF < 5, maximum: 4.98, supplement Table S7) and were used along with watershed association and management type in the full RDA analysis to explain among-lake species richness jointly across all taxa. The RDA-based forward model selection retained a few environmental variables as key correlates of species richness of multiple taxa across lakes, but management type dropped from the best model (Table 7). Therefore, among lake variation in richness across several aquatic and riparian taxa was solely explained by environmental factors unrelated to either management type or recreation-related variables. Specifically, the coverage of woody habitat along the littoral was negatively correlated with riparian species richness and positively correlated with tree diversity along the first axis in Figure 4. The extent of agricultural land use (representing also more rural conditions, supplementary Table S6) was positively associated with riparian species richness (Figure 4). Lake steepness (representing also small lake size and low shoreline development factor, supplementary Table S5) was negatively correlated with waterfowl species richness (Figure 5). All other environmental variables, including lake age and watershed were not significant (Table 7). The best model explained 36 % of the total variance in the multivariate species richness. In this model, neither management type nor any of the recreational use variables explained variation in species richness of a range of aquatic and riparian taxa among lakes.

4. Discussion

We found support for our study hypotheses. Specifically, in line with initial expectations we found no differences in species richness, Simpson diversity and conservation value across all examined taxa among managed and unmanaged gravel pit lakes, and we found a similar species pool to be present in both lake types. Collectively, our study did not reveal that recreational-fisheries management (through impacts on fish communities) or the presence of anglers (through disturbance effects on shoreline habitat and wildlife or lethal impacts through lost fishing gear) significantly constrains the development of diverse communities of amphibians, birds, submerged macrophytes, terrestrial plants and Odonata relative to those expected at lakes that are not managed for recreational fisheries. Instead, we found the best predictors of the variation in species richness among lakes related to land use variables, the extent of woody habitat at the lake shores and the lake morphology (size and steepness). Therefore, our study suggests that for the taxa and lake types that we examined broader environmental factors and land use, and not the presence of recreational fisheries and its management of fish stocks and littoral zones, shape taxonomic diversity of plants, birds, amphibians, and dragonflies.

Biodiversity potential of gravel pit lakes

We found gravel pit lakes in Lower Saxony, Germany, to host a substantial species diversity and fraction of the regional species' pools of aquatic and riparian taxa, in particular trees, Odonata and waterfowl. This finding supports related work in other areas of Europe (Damnjanović et al., 2018; Santoul et al., 2009; Spyra & Strzelec, 2019). Yet, only small fractions of the regional species' pools were detected for vascular plant species, submerged macrophytes, songbirds and amphibians. In particular amphibians are considered very sensitive to predation from fish (Hecnar & M'Closkey, 1997; Miró et al., 2018). Many amphibian species depend on shallow water and best develop in small, temporary waters (Porej & Hetherington, 2005; Shulse, Semlitsch, Trauth, & Williams, 2010; Temple & Cox, 2009). None of our study lakes, however, were free of fish (Matern et al., 2019). Our gravel pits were also relatively steeply-sloped with small fractions of littoral areas, disconnected from rivers, placed in agricultural landscapes and close to anthropogenic infrastructure. All of these factors are negative for amphibian diversity and can explain the low species richness we detected for this taxon (Porej & Hetherington, 2005; Shulse et al., 2010; Temple & Cox, 2009). Importantly, with our study results however, management by recreational fisheries and the substantially different fish communities in managed and unmanaged lakes can be excluded as an additional stressor.

Environmental differences among managed and unmanaged lakes

The gravel pit lakes we studied were similar in the majority of the environmental factors that we examined (including age) except the coverage of submerged macrophytes, which was more prevalent in managed gravel pit lakes compared to unmanaged ones. Submerged macrophytes have been reported to be strongly affected by fish stocking of benthivorous species, such as common carp (Bajer et al., 2016; Miller & Crowl, 2006). However, in a subset of the same gravel pit lakes presented in this paper, Matern et al. (2019) found similar biomasses of fishes in managed and unmanaged lakes. Due to the sampling gear used in the study by Matern et al. (2019), the authors likely underestimated the abundance and biomass of common carp and other large benthivorous fish (Ravn et al., 2019). Although no absolute biomass data of carp or other species in our study lakes are available, the fact that submerged macrophytes were more diverse and more developed in the angler-managed lakes suggests that co-existence of carp and other game fish with a species rich submerged macrophyte community, also in terms of threatened stonewort species (*Chara sp., Nitella sp.*), is possible. This disagrees with expectations expressed elsewhere that managing lakes with benthivorous fish necessarily harms submerged macrophytes (Van de Weyer, Meis, & Krautkrämer, 2015). The more developed submerged macrophytes in managed lakes revealed in this study instead suggests that critical biomass thresholds for benthivorous fish after which macrophytes typically vanish or strongly decline (about 100 kg/ha; Vilizzi, Tarkan, & Copp, 2015) might not have been reached in our study lakes. Alternatively, the transferability of typically mesocosm studies that have reported substantial impacts of carp on macrophytes to occur after reaching about 100 kg/ha may not hold under conditions in the wild (Arlinghaus, Hühn, et al., 2017). We speculate the regular shoreline development activities by anglers and angling clubs to maintain access to angling sites may create “disturbances” (Dustin & Vondracek, 2017; O’Toole et al., 2009) that regularly interrupt the succession of tree stands, thereby reducing the shading effects in the littoral zone (Balandier et al., 2008; Monk & Gabrielson, 1985). Reduced shading of the shallow littoral can promote growth of submerged macrophytes. Another factor promoting macrophyte growth could be angler-induced nutrient inputs (Niesar, Arlinghaus, Rennert, & Mehner, 2004), although this factor is unlikely, given that the trophic variables did not differ among managed and unmanaged lakes in our study.

Lake shorelines managed by anglers were previously reported to be heavily modified to accommodate angling sites and access to anglers (Dustin & Vondracek, 2017; O’Toole et al., 2009). Although improved accessibility in angler-managed lakes was supported in our study, the amount of aquatic and riparian vegetation was nevertheless significantly larger in angler-managed systems compared to unmanaged lakes. This indicates that maintaining accessibility of lakeshores to anglers does not necessarily mean degraded riparian or littoral habitat quality. In fact, anglers have an interest to maintain access to lakes to be able to fish, but there is also an interest in developing suitable habitats for fish (Meyerhoff et al., 2019) and maintaining sites that promise solitude during the experience (Beardmore, Hunt, Haider, Dorow, & Arlinghaus, 2015), which indirectly may also support biodiversity. The littoral zone is the most productive habitat of lakes (Winfield, 2004), and fish species depend on submerged macrophytes and other structures for spawning and refuge (Lewin, Mehner, Ritterbusch, & Brämick, 2014; Lewin, Okun, & Mehner, 2004). At the same time, crowding is a severe constraint that reduces angler satisfaction (Beardmore et al., 2015). Therefore, although anglers regularly engage in shoreline development activities and angling site maintenance, the data of this study suggest they do so to a degree that may maintain or even foster aquatic and riparian vegetation.

Differences in recreational use of managed and unmanaged lakes

Managed lakes were found to have more developed tracks, paths, parking places and other facilities that attract anglers and other recreationists. Thus, angler-managed lakes were generally more accessible to water-based recreationists, although these differences were not always statistically significant among the two lake types for recreational uses other than angling. Importantly, despite managed lakes receiving regular fisheries-management activities such as stocking and angler use, neither “management type” nor the index of general recreational use intensity were related with species richness across multiple taxa and lakes. Thus, for the diversity metrics and the taxa we examined (vegetation, odonata, amphibians, birds), our study does not suggest that the use of gravel pits by recreational fisheries significantly constraints the development of aquatic and riparian biodiversity across a range of taxa. Clearly, species-specific effects on disturbance-sensitive species (e.g., selected bird species; Knight, Anderson, & Marr, 1991) may still occur, which our aggregate metrics of taxonomic richness or the Simpson community diversity index might have been too insensitive to detect. Further work on community differences among managed and unmanaged lakes is warranted and will form the basis of a future paper.

Differences in biodiversity among managed and unmanaged lakes

Across all the taxa we examined, no statistical differences were found in species richness, number of threatened species, conservation value and Simpson diversity between managed and unmanaged lakes. This result was unexpected. Recreational-fisheries management can affect aquatic and riparian biodiversity through various pathways, first through supporting and enhancing fish stocks that exert predation pressure (e.g., on tadpoles and Odonata larvae; Hecnar & M’Closkey, 1997; Knorp & Dorn, 2016), second through indirect fish-based effects (e.g., uprooting macrophytes through benthivorous feeding; Bajer et al., 2016), third through direct removal or damage of submerged and terrestrial plants during angling activities (O’Toole et al., 2009), which may have knock-on effects on dragonflies (Z. Müller et al., 2003), and forth through activity-based disturbance effects or lethal impacts through lost fishing sinkers in particular on birds (Cryer et al., 1987; Sears, 1988). Our study design was not tailored towards directly measuring disturbance effects on particular species; instead we choose to aggregately look at a range of taxonomic richness indices for communities present at gravel pit lakes that are associated with recreational fisheries compared to ecologically similar lakes that do not. When judged against these aggregate biodiversity metrics, our work does not support the idea that recreational-fisheries management and angler presence has important impacts that modify species inventories to a degree that strongly depart from situations expected at unmanaged lakes without anglers. Previous work has reported relevant reductions in bird biodiversity from lakes exposed to human disturbances due to recreation including angling (Bell et al., 1997; Newbrey, Bozek, & Niemuth, 2005). However, we found similar species richness and conservation value of both waterfowl and riparian songbirds in managed and unmanaged lakes. This does not exclude the possibility that for example the breeding success of specific disturbance-sensitive taxa might have been impaired in angler-managed lakes (Park, Park, Sung, & Park, 2006; Reichholf, 1988). But if such effects were present, they were not strong enough to substantially alter species richness, not to be confused with species identity. Overall, against the metrics we choose, our findings supported the initial hypothesis of no impacts from recreational fishing on non-targeted taxa in gravel pits situated in agricultural landscapes.

Environmental determinants of aquatic and riparian biodiversity in gravel pit lakes

The species richness of different taxa did not uniformly vary among lakes, in contrast to a study of managed shallow ponds by Lemmens et al. (2013). While examining strictly aquatic taxa (zooplankton, submerged and emerged aquatic macrophytes, benthic invertebrates), Lemmens et al. (2013) revealed uniform responses in species richness across taxa and ponds. The much broader trophic and habitat requirements of aquatic and riparian taxa studied in our work resulted in significantly more variable biotic responses. For example, lakes rich in riparian biodiversity were not necessarily rich in submerged macrophytes and waterfowl. The reason is that the biodiversity of aquatic and riparian biodiversity respond to many more variables than in-lake variables. Our multivariate analyses revealed the variation in species richness across multiple taxa was driven by structural variables of the habitat quality, lake morphometry (size and steepness) and land use in a buffer zone around the lake, but not by recreational use intensity or the presence of recreational-fisheries management activities. Thus, environmental factors unrelated to recreational fishing seem to overwhelm any specific impacts of angling, at least for the taxonomic diversity metrics and the taxa examined in our work.

Mosaics of different habitats (reeds, overhanging trees etc.) along the shoreline support species richness and diversity for most taxa (Kaufmann, Hughes, Whittier, Bryce, & Paulsen, 2014), and the presence of endangered biodiversity increase the recreational value of gravel pit as perceived by anglers (Meyerhoff et al., 2019). Extended woody habitat both in water and particularly in the riparian zone was correlated with increased tree diversity, but reduced riparian species richness of vascular plants, amphibians, Odonata and songbirds. This might be explained by the shading effect of trees on herbal vegetation (Balandier et al., 2008; Monk & Gabrielson, 1985). Odonata, songbirds and amphibian species benefited from more vegetated littoral habitats, in agreement with previous work (Paracuellos, 2006; Remsburg & Turner, 2009; Shulse et al., 2010). The species richness of waterfowl was strongly governed by lake area and steepness of the littoral, with larger and shallower lakes having a higher waterfowl species richness, confirming earlier findings by Elmberg, Nummi, Poysa, & Sjoberg (2006) and Paszkowski & Tonn (2000). The three dominant waterfowl species (occurring on 85 % or more of sampled lakes) were either omnivorous (mallard, *Anas platyrhynchos*, L.) or herbivorous-invertivorous (common coot, *Fulica atra*, L., and tufted duck, *Aythya fuligula*, L.). In addition, 77 % of the lakes were used by grey goose (*Anser anser*, L.), which feeds on terrestrial plants. Thus, it can be concluded that the dominant waterfowl detected at our studied lakes benefit from submerged macrophytes or riparian plants, both found to be more abundant at managed lakes.

Collectively, our data do not support substantial negative impacts of recreational fisheries management on the species richness and community diversity of waterfowl and songbirds present at gravel pit lakes. In a related study from Welsh reservoirs Cryer, Linley, Ward, Stratford, & Randerson (1987) observed only distributional changes of waterfowl to the presence of anglers and no changes in abundance. Similarly negligible effects of anglers on piscivorous birds at Canadian natural lakes were reported by Somers, Heisler, Doucette, Kjoss, & Brigham (2015). Specifically for gravel pit lakes, Bell et al. (1997) failed to find evidence for impacts of recreational fishing on the community structure of waterfowl, although in particular diving waterfowl reduced their abundance during presence of anglers and other recreationists. In that study, similar to ours, habitat quality and lake size were more important for waterfowl diversity than the bank use by anglers, and their shoreline management rather supported grazing waterfowl by opening up sites (Bell et al., 1997). This does not mean that recreational fishing will not impact bird populations at all as for example the breeding success of disturbance sensitive species might still be impaired (Park et al., 2006; Reichholf, 1988). Our study design was not designed to examine the breeding success of particular species and rather focused on aggregate diversity metrics.

Limitations

The strength of our study design is the focus on multiple taxa, which is rare in the recreational ecology literature related to freshwaters. The limitation is that we did not focus on specific species, and our sampling design does not answer whether the detected mobile species (e.g., birds or Odonata) recruited in the study lakes or only used them temporarily as feeding or resting habitat. Moreover, because of adjustments in taxa-specific sampling schemes, in our sampling we might have underestimated seasonal taxa (amphibians, Odonata) and we likely missed rare species (Yoccoz, Nichols, & Boulinier, 2001; Zhang et al., 2014). However, even if this is the case the conclusions of our work are robust because this systematic error affected both management types that we compared in our study.

Our study used a comparative approach where lakes were not randomly allocated to either angler-managed lakes or controls. However, because the sample of our lakes were all from the same geographical area and the age of our lakes and the wider environmental factors were mainly similar, the key differences among lake types related to the presence of recreational fishing. Therefore, we content that our design would have been able to differentiate strong angling-induced biodiversity effects would they exist in reality.

A further limitation is that our design did not include entirely unused lakes where recreation whatsoever is prohibited. Our data have to be interpreted against the possibility that gravel pits situated in reserves with strictly no human access might show higher species diversity than revealed in our sample lakes. All the lakes were situated in agricultural environments and all were exposed to a certain recreational use. Background disturbance (Liley, Underhill-Day, Panter, Marsh, & Roberts, 2015) might have affected the observed species pool, affecting the detectability of species in our study region.

The conclusions of our work are also confined to the environmental gradients that we could observe. For example, higher angler use intensities than found in the present work might reveal different results.

Finally, the recreational use intensity was mainly recorded during weekdays when the field visits were done. Thus, potential high intensity phases at weekends might be unrepresented. However, this would even strengthen our conclusions, if the real recreational use of managed lakes was well beyond what was considered in our analyses.

Conclusions

Our study shows that recreational-fisheries management does not constrain the establishment of a rich biodiversity of aquatic and riparian taxa in and around gravel pit lakes relative to conditions expected at ecologically similar unmanaged lakes. Environmental variables related to habitat quality, land use and lake morphometry rather than recreation-related drivers were key in driving the species richness for multiple taxa across lakes. Our study thus shows that co-existence of recreational fisheries and aquatic and riparian biodiversity of high conservation value and richness is possible, at least under the specific ecological conditions of gravel pit lakes in agricultural landscapes.

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References

AdV - Working Committee of the Surveying Authorities of the States of the Federal Republic of Germany. (2006). *Documentation on the Modelling of Geoinformation of Official Surveying and Mapping (GeoInfoDok)* (S. Afflerbach & W. Kunze, Eds.). Munich, Germany.

Altmüller, R., & Clausnitzer, H.-J. (2010). Rote Liste der Libellen Niedersachsens und Bremens: 2. Fassung, Stand 2007. *Informationsdienst Naturschutz Niedersachsen*, *30*, 211–238.

Arlinghaus, R. (2005). A conceptual framework to identify and understand conflicts in recreational fisheries systems, with implications for sustainable management. *Aquatic Resources, Culture and Development*, *1*(2), 145–174.

Arlinghaus, R., Alós, J., Beardmore, B., Daedlow, K., Dorow, M., Fujitani, M., … Wolter, C. (2017). Understanding and managing freshwater recreational fisheries as complex adaptive social-ecological systems. *Reviews in Fisheries Science and Aquaculture*, *25*(1), 1–41. https://doi.org/10.1080/23308249.2016.1209160

Arlinghaus, R., Cyrus, E.-M., Eschbach, E., Fujitani, M., Hühn, D., Johnston, F., … Riepe, C. (2015). *Hand in Hand für eine nachhaltige Angelfischerei* (Band 28). Berichte des IGB.

Arlinghaus, R., Hühn, D., Pagel, T., Beck, M., Rapp, T., & Wolter, C. (2017). Fischereilicher Nutzen und gewässerökologische Auswirkungen des Besatzes mit Karpfen (Cyprinus carpio) in stehenden Gewässern: Ergebnisse und Schlussfolgerungen aktueller Ganzseeexperimente und Meta-Analysen. *Fischerei Und Fischmarkt in Mecklenburg-Vorpommern*, *1*, 36–46.

Arlinghaus, R., Lorenzen, K., Johnson, B. M., Cooke, S. J., & Cowx, I. G. (2016). Management of freshwater fisheries: Addressing habitat, people and fishes. In J. F. Craig (Ed.), *Freshwater Fisheries Ecology* (pp. 557–579). https://doi.org/10.1002/9781118394380.ch44

Bajer, P. G., Beck, M. W., Cross, T. K., Koch, J. D., Bartodziej, W. M., & Sorensen, P. W. (2016). Biological invasion by a benthivorous fish reduced the cover and species richness of aquatic plants in most lakes of a large North American ecoregion. *Global Change Biology*, *22*(12), 3937–3947. https://doi.org/10.1111/gcb.13377

Balandier, P., Marquier, A., Dumas, Y., Gaudio, N., Philippe, G., Da Silva, D., … Sinoquet, H. (2008). Light sharing among different forest strata for sustainable management of vegetation and regeneration. In *Forestry in achieving millennium goals* (Nov, pp. 81–86). Retrieved from https://hal.archives-ouvertes.fr/hal-00468830/document

Beardmore, B., Hunt, L. M., Haider, W., Dorow, M., & Arlinghaus, R. (2015). Effectively managing angler satisfaction in recreational fisheries requires understanding the fish species and the anglers. *Canadian Journal of Fisheries and Aquatic Sciences*, *72*, 500–513. https://doi.org/10.1139/cjfas-2014-0177

Bell, M. C., Delany, S. N., Millett, M. C., & Pollitt, M. S. (1997). Wintering waterfowl community structure and the characteristics of gravel pit lakes. *Wildlife Biology*, *3*(3/4), 65–78. https://doi.org/10.2981/wlb.1997.009

Blanchet, F. G., Legendre, P., & Borcard, D. (2008). Forward selection of explanatory variables. *Ecology*, *89*(9), 2623–2632. Retrieved from https://www.jstor.org/stable/27650800

Blanchette, M. L., & Lund, M. A. (2016). Pit lakes are a global legacy of mining: An integrated approach to achieving sustainable ecosystems and value for communities. *Current Opinion in Environmental Sustainability*, *23*, 28–34. https://doi.org/10.1016/j.cosust.2016.11.012

Braun-Blanquet, J. (1964). *Pflanzensoziologie: Grundzüge der Vegetationskunde* (3rd ed.). https://doi.org/10.1007/978-3-7091-4078-9

Chambers, J. M., & Hastie, T. J. (1992). *Statistical models in S* (Vol. 251). Pacific Grove, CA: Wadsworth & Brooks/Cole; Advanced Books & Software.

Cryer, M., Linley, N. W., Ward, R. M., Stratford, J. O., & Randerson, P. F. (1987). Disturbance of overwintering wildfowl by anglers at two reservoir sites in south wales. *Bird Study*, *34*(3), 191–199. https://doi.org/10.1080/00063658709476961

Cucherousset, J., Boulêtreau, S., Azémar, F., Compin, A., Guillaume, M., & Santoul, F. (2012). “Freshwater killer whales”: Beaching behavior of an alien fish to hunt land birds. *PLoS ONE*, *7*(12), 1–6. https://doi.org/10.1371/journal.pone.0050840

Damnjanović, B., Novković, M., Vesić, A., Živković, M., Radulović, S., Vukov, D., … Cvijanović, D. (2018). Biodiversity-friendly designs for gravel pit lakes along the Drina River floodplain (the Middle Danube Basin, Serbia). *Wetlands Ecology and Management*, *27*, 1–22. https://doi.org/10.1007/s11273-018-9641-8

Dear, E. J., Guay, P.-J., Robinson, R. W., & Weston, M. A. (2015). Distance from shore positively influences alert distance in three wetland bird species. *Wetlands Ecology and Management*, *23*(2), 315–318. https://doi.org/10.1007/s11273-014-9376-0

DeBoom, C. S., & Wahl, D. H. (2013). Effects of coarse woody habitat complexity on predator-prey interactions of four freshwater fish species. *Transactions of the American Fisheries Society*, *142*(6), 1602–1614. https://doi.org/10.1080/00028487.2013.820219

Díaz, S., Settele, J., Brondizio, E. S., Ngo, H. T., Agard, J., Arneth, A., … Zayas, C. N. (2019). Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science*, *1327*(December). https://doi.org/10.1126/science.aaw3100

Dierschke, V. (2016). *Welcher Vogel ist das?: 170 Vögel einfach bestimmen*. Kosmos.

DIN EN 1484. (1997). *Water analysis - Guidelines for the determination of total organic carbon (TOC) and dissolved organic carbon (DOC)*.

DIN EN ISO 13395. (1996). *Water quality - Determination of nitrite nitrogen and nitrate nitrogen and the sum of both by flow analysis (CFA and FIA) and spectrometric detection (ISO 13395:1996)*.

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Lévêque, C., … Sullivan, C. A. (2006). Freshwater biodiversity: Importance, threats, status and conservation challenges. *Biological Reviews of the Cambridge Philosophical Society*, *81*(2), 163–182. https://doi.org/10.1017/S1464793105006950

Dustin, D. L., & Vondracek, B. (2017). Nearshore habitat and fish assemblages along a gradient of shoreline development. *North American Journal of Fisheries Management*, *37*(2), 432–444. https://doi.org/10.1080/02755947.2017.1280567

Elmberg, J., Nummi, P., Poysa, H., & Sjoberg, K. (2006). Relationships between species number, lake size and resource diversity in assemblages of breeding waterfowl. *Journal of Biogeography*, *21*(1), 75. https://doi.org/10.2307/2845605

Emmrich, M., Schälicke, S., Hühn, D., Lewin, W.-C., & Arlinghaus, R. (2014). No differences between littoral fish community structure of small natural and gravel pit lakes in the northern German lowlands. *Limnologica*, *46*, 84–93. https://doi.org/10.1016/j.limno.2013.12.005

EN ISO 11732. (2005). *Water quality - Determination of ammonium nitrogen - Method by flow analysis (CFA and FIA) and spectrometric detection (ISO 11732:1997)*.

EN ISO 6878. (2004). *Water quality - Determination of phosphorus - Ammonium molybdate spectrometric method (ISO 6878:1998)*.

European Aggregates Association (UEPG). (2017). Estimates of aggregates production data 2017. Retrieved May 2, 2019, from http://www.uepg.eu/statistics/estimates-of-production-data/data-2017

Franson, J. C., Hansen, S. P., Creekmore, T. E., Brand, C. J., Evers, D. C., Duerr, A. E., & DeStefano, S. (2003). Lead fishing weights and other fishing tackle in selected waterbirds. *Waterbirds: The International Journal of Waterbird Biology*, *26*(3), 345–352. Retrieved from https://www.jstor.org/stable/1522416

Frid, A., & Dill, L. M. (2002). Human-caused disturbance stimuli as a form of predation risk.

Conservation Ecology, *6*(1), 11[online]. Retrieved from https://www.jstor.org/stable/26271862

Garve, E. (2004). Rote Liste und Florenliste der Farn- und Blütenpflanzen in Niedersachsen und Bremen, 5. Fassung, Stand 1.3.2004. *Informationsdienst Naturschutz Niedersachsen*, *24*(1), 1–76.

Gräler, B., Pebesma, E., & Heuvelink, G. (2016). Spatio-temporal interpolation using gstat. *The R Journal*, *8*(1), 204–218. Retrieved from http://journal.r-project.org/archive/2016-1/na-pebesma-heuvelink.pdf

Granek, E. F., Madin, E. M. P., Brown, M. A., Figueira, W., Cameron, D. S., Hogan, Z., … Arlinghaus, R. (2008). Engaging recreational fishers in management and conservation: Global case studies. *Conservation Biology*, *22*(5), 1125–1134. https://doi.org/10.1111/j.1523-1739.2008.00977.x

Hecnar, S. J., & M'Closkey, R. T. (1997). The effects of predatory fish on amphibian species richness and distribution. *Biological Conservation*, *79*(2–3), 123–131. https://doi.org/10.1016/s0006-3207(96)00113-9

Hein, T. (2018). Libellenwissen. Retrieved June 6, 2018, from https://libellenwissen.de/

IPBES. (2019). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services – unedited advance version* (S. Díaz, J. Settele, E. Brondízio, H. T. Ngo, M. Guèze, J. Agard, … H. Mooney, Eds.). Paris, France: Office of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES).

Kaufmann, P. R., Hughes, R. M., Whittier, T. R., Bryce, S. A., & Paulsen, S. G. (2014). Relevance of lake physical habitat indices to fish and riparian birds. *Lake and Reservoir Management*, *30*(2), 177–191. https://doi.org/10.1080/10402381.2013.877544

Kaufmann, P. R., & Whittier, T. R. (1997). Habitat assessment. In J. R. Baker, D. V. Peck, & D. W. Sutton (Eds.), *Environmental monitoring and assessment program – Surface waters: Field operations manual for lakes* (pp. 5–1 to 5–26). Washington, D.C.: U.S. Environmental Protection Agency.

Knight, R. L., Anderson, D. P., & Marr, N. V. (1991). Responses of an avian scavenging guild to anglers. *Biological Conservation*, *56*(2), 195–205. https://doi.org/10.1016/0006-3207(91)90017-4

Knorp, N. E., & Dorn, N. J. (2016). Mosquitofish predation and aquatic vegetation determine emergence patterns of dragonfly assemblages. *Freshwater Science*, *35*(1), 114–125. https://doi.org/10.1086/684678

Kohler, A. (1978). Methoden der Kartierung von Flora und Vegetation von Süßwasserbiotopen. *Landschaft Und Stadt*, *10*, 73–85.

Korsch, H., Doege, A., Raabe, U., & van de Weyer, K. (2013). Rote Liste der Armleuchteralgen (Charophyceae) Deutschlands 3. Fassung, Stand: Dezember 2012. *Hausknechtia*, *17*(November), 1–32.

Krüger, T., & Nipkow, M. (2015). Rote Liste der in Niedersachsen und Bremen gefährdeten Brutvogelarten, 8. Fassung, Stand 2015. *Informationsdienst Naturschutz Niedersachsen*, *35*(4), 181–256. Retrieved from http://www.nlwkn.niedersachsen.de/naturschutz/veroeffentlichungen/rote-liste-der-in-niedersachsen-und-bremen-gefaehrdeten-brutvoegel-141167.html

Landesamt für Statistik Niedersachsen (LSN). (2018). *LSN-Online - Regionaldatenbank*. Hannover, Germany.

Legendre, P., & Legendre, L. (2012). *Numerical ecology* (3rd Editio). Elsevier.

Lehmann, A. W., & Nüss, J. H. (2015). *Libellen - Bestimmungsschlüssel für Nord- und Mitteleuropa* (6. Auflage). Göttingen, Germany: Deutscher Jugendbund für Naturbeobachtungen.

Lemmens, P., Mergeay, J., De Bie, T., Van Wichelen, J., De Meester, L., & Declerck, S. A. J. (2013). How to maximally support local and regional biodiversity in applied conservation? Insights from pond management. *PLoS One*, *8*, e72538. https://doi.org/10.1371/journal.pone.0072538

Lemmens, P., Mergeay, J., Van Wichelen, J., De Meester, L., & Declerck, S. A. J. (2015). The impact of conservation management on the community composition of multiple organism groups in eutrophic interconnected man-made ponds. *PLoS One*, *10*, e0139371. https://doi.org/10.1371/journal.pone.0139371

Lewin, W.-C., Mehner, T., Ritterbusch, D., & Brämick, U. (2014). The influence of anthropogenic shoreline changes on the littoral abundance of fish species in German lowland lakes varying in depth as determined by boosted regression trees. *Hydrobiologia*, *724*, 293–306. https://doi.org/10.1007/s10750-013-1746-8

Lewin, W.-C., Okun, N., & Mehner, T. (2004). Determinants of the distribution of juvenile fish in the littoral area of a shallow lake. *Freshwater Biology*, *49*, 410–424. https://doi.org/10.1111/j.1365-2427.2004.01193.x

Liley, D., Underhill-Day, J., Panter, C., Marsh, P., & Roberts, J. (2015). *Morecambe Bay bird disturbance and access management report. Unpublished report by Footprint Ecology for the Morecambe Bay Partnership.*

Mantoura, R. F. C., & Llewellyn, C. A. (1983). The rapid determination of algal chlorophyll and carotenoid pigments and their breakdown products in natural waters by reverse-phase high-performance liquid chromatography. *Analytica Chimica Acta*, *151*(C), 297–314. https://doi.org/10.1016/S0003-2670(00)80092-6

Mardia, K. V., Kent, J. T., & Bibby, J. M. (1979). *Multivariate Analysis*. London: Academic Press.

Matern, S., Emmrich, M., Klefoth, T., Wolter, C., Nikolaus, R., Wegener, N., & Arlinghaus, R. (2019). Effect of recreational-fisheries management on fish biodiversity in gravel pit lakes, with contrasts to unmanaged lakes. *Journal of Fish Biology*, *94*, 865–881. https://doi.org/10.1111/jfb.13989

Matthews, W. J. (1986). Fish faunal "breaks" and stream order in the eastern and central United States. *Environmental Biology of Fishes*, *17*(2), 81–92. https://doi.org/10.1007/BF00001739

McFadden, T. N., Herrera, A. G., & Navedo, J. G. (2017). Waterbird responses to regular passage of a birdwatching tour boat: Implications for wetland management. *Journal for Nature Conservation*, *40*, 42–48. https://doi.org/10.1016/J.JNC.2017.09.004

Meyerhoff, J., Klefoth, T., & Arlinghaus, R. (2019). The value artificial lake ecosystems provide to recreational anglers: Implications for management of biodiversity and outdoor recreation. *Journal of Environmental Management*, *252*(September), 109580. https://doi.org/10.1016/j.jenvman.2019.109580

Miller, S. A., & Crowl, T. A. (2006). Effects of common carp (Cyprinus carpio) on macrophytes and invertebrate communities in a shallow lake. *Freshwater Biology*, *51*(1), 85–94. https://doi.org/10.1007/s00198-005-1915-3

Miró, A., Sabás, I., & Ventura, M. (2018). Large negative effect of non-native trout and minnows on Pyrenean lake amphibians. *Biological Conservation*, *218*, 144–153. https://doi.org/10.1016/j.biocon.2017.12.030

Monk, C. D., & Gabrielson, F. C. J. (1985). Effects of shade, litter and root competition on old-field vegetation in South Carolina. *Bulletin of the Torrey Botanical Club*, *112*(4), 383–392. Retrieved from https://www.jstor.org/stable/2996039

Müller, H. (2012). *Zulässigkeit und Grenzen der Ausgestaltung/Einschränkung von Fischereirechten an Baggerseen; Rechtsgutachten*. Bezirksfischereiverband Oberfranken e.V., Landesfischereiverband Bayern e.V.

Müller, Z., Jakab, T., Tóth, A., Dévai, G., Szállassy, N., Kiss, B., & Horváth, R. (2003). Effect of sports fisherman activities on dragonfly assemblages on a Hungarian river floodplain. *Biodiversity and Conservation*, *12*, 167–179.

Murphy, J., & Riley, J. P. (1962). A modified single solution method for the determination of phosphate in natural water. *Analytica Chimica Acta*, *27*, 31–36.

Neter, J., Kutner, M. H., Nachtsheim, C. J., & Wasserman, W. (1996). *Applied linear statistical models* (Fourth Edi). Chicago, Illinois, USA: Irwin.

Newbrey, J. L., Bozek, M. A., & Niemuth, N. D. (2005). Effects of lake characteristics and human disturbance on the presence of piscivorous birds in Northern Wisconsin, USA. *Waterbirds*, *28*(4), 478–486. https://doi.org/10.1675/1524-4695(2005)28[478:eolcah]2.0.co;2

Niesar, M., Arlinghaus, R., Rennert, B., & Mehner, T. (2004). Coupling insights from a carp, Cyprinus carpio, angler survey with feeding experiments to evaluate composition, quality and phosphorus input of groundbait in coarse fishing. *Fisheries Management and Ecology*, *11*(3-4), 225–235. https://doi.org/10.1111/j.1365-2400.2004.00400.x

O'Toole, A. C., Hanson, K. C., & Cooke, S. J. (2009). The effect of shoreline recreational angling activities on aquatic and riparian habitat within an urban environment: Implications for conservation and management. *Environmental Management*, *44*(2), 324–334. https://doi.org/10.1007/s00267-009-9299-3

Oertli, B., Joye, D. A., Castella, E., Juge, R., Cambin, D., & Lachavanne, J.-B. (2002). Does size matter? The relationship between pond area and biodiversity. *Biological Conservation*, *104*, 59–70.

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., … Wagner, H. (2018). *vegan: Community Ecology Package*. Retrieved from https://cran.r-project.org/package=vegan

Osgood, R. A. (2005). Shoreline density. *Lake and Reservoir Management*, *21*(1), 125–126. https://doi.org/10.1080/07438140509354420

Paracuellos, M. (2006). Relationships of songbird occupation with habitat configuration and bird abundance in patchy reed beds. *Ardea*, *94*(1), 87–98.

Park, J.-H., Park, H.-W., Sung, H.-C., & Park, S.-R. (2006). Effect of fishing activity on nest selection and density of waterfowls in Namyang Lake. *Journal of Ecology and Field Biology*, *29*(3), 213–217.

Paszkowski, C. A., & Tonn, W. M. (2000). Effects of lake size, environment, and fish assemblage on species richness of aquatic birds. *Internationale Vereinigung Für Theoretische Und Angewandte Limnologie: Verhandlungen*, *27*(1), 178–182. https://doi.org/10.1080/03680770.1998.11901222

Pielou, E. C. (1969). *An introduction to mathematical ecology.* Retrieved from https://www.cabdirect.org/cabdirect/abstract/19729701428

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., … Sexton, J. O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection. *Science*, *344*(6187), 1246752. https://doi.org/10.1126/science.1246752

Podloucky, R., & Fischer, C. (2013). Rote Listen und Gesamtartenlisten der Amphibien und Reptilien in Niedersachsen und Bremen, 4. Fassung. *Informationsdienst Naturschutz Niedersachsen*, *4*(13).

Porej, D., & Hetherington, T. E. (2005). Designing wetlands for amphibians: The importance of predatory fish and shallow littoral zones in structuring of amphibian communities. *Wetlands Ecology and Management*, *13*, 445–455. https://doi.org/10.1007/s11273-004-0522-y

Preising, E., Vahle, H.-C., Brandes, D., Hofmeister, H., Tüxen, J., & Weber, H. E. (1990). Die Pflanzengesellschaften Niedersachsens - Bestandsentwicklung, Gefährdung und Schutzprobleme: Salzpflanzengesellschaften der Meeresküste und des Binnenlandes // Wasser- und Sumpfpflanzengesellschaften des Süßwassers. In B. Pilgrim (Ed.), *Naturschutz Landschaftspflege Niedersachsens* (20/7 & 8). Hannover, Germany: Niedersächsisches Landesamt für Ökologie - Naturschutz -.

R Core Team. (2013). *R: A language and environment for statistical computing*. Vienna, Austria.

Randler, C. (2006). Disturbances by dog barking increase vigilance in coots Fulica atra. *European Journal of Wildlife Research*, *52*(4), 265–270. https://doi.org/10.1007/s10344-006-0049-z

Rat der europäischen Gemeinschaft. Richtlinie 92/43/EWG des Rates vom 21.Mai 1992 zur Erhaltung der natürlichen Lebensräume sowie der wildlebenden Tiere und Pflanzen. , Amtsblatt der Europäischen Union § (1992).

Ravn, H. D., Lauridsen, T. L., Jepsen, N., Jeppesen, E., Hansen, P. G., Hansen, J. G., & Berg, S. (2019). A comparative study of three different methods for assessing fish communities in a small eutrophic lake. *Ecology of Freshwater Fish*, *28*(2), 341–352. https://doi.org/10.1111/eff.12457

Reichholf, J. H. (1988). Effects of anglers on the breeding of water birds in the internationally important wetland "Lower Inn River." *Vogelwelt*, *109*(Heft 5/6), 206–221.

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T. J., … Cooke, S. J. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews*, *94*(3), 849–873. https://doi.org/10.1111/brv.12480

Rempel, R. S., Hobson, K. A., Holborn, G., Van Wilgenburg, S. L., & Elliott, J. (2005). Bioacoustic monitoring of forest songbirds: interpreter variability and effects of configuration and digital processing methods in the laboratory. *Journal of Field Ornithology*, *76*(1), 1–11. https://doi.org/10.1648/0273-8570-76.1.1

Remsburg, A. J., & Turner, M. G. (2009). Aquatic and terrestrial drivers of dragonfly (Odonata) assemblages within and among north-temperate lakes. *Journal of the North American Benthological Society*, *28*(1), 44–56. https://doi.org/10.1899/08-004.1

Reynaud, A., & Lanzanova, D. (2017). A global meta-analysis of the value of ecosystem services provided by lakes. *Ecological Economics*, *137*, 184–194. https://doi.org/10.1016/j.ecolecon.2017.03.001

Riedmüller, U., Hoehn, E., & Mischke, U. (2013). *Auswerte-Tool für die Trophie-Klassifikation von Seen. Trophie-Index nach LAWA.*

Sala, O. E., Chapin III., F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., … Wall, D. H. (2000). Global biodiversity scenarios for the year 2100. *Science*, *287*, 1770–1774. Retrieved from https://www.ncbi.nlm.nih.gov/pubmed/10710299

Santoul, F., Gaujard, A., Angélibert, S., Mastrorillo, S., & Céréghino, R. (2009). Gravel pits support waterbird diversity in an urban landscape. *Hydrobiologia*, *634*(1), 107–114. https://doi.org/10.1007/s10750-009-9886-6

Saulnier-Talbot, É., & Lavoie, I. (2018). Uncharted waters: The rise of human-made aquatic environments in the age of the "Anthropocene." *Anthropocene*, *23*, 29–42. https://doi.org/10.1016/j.ancene.2018.07.003

Schaumburg, J., Schranz, C., Stelzer, D., & Vogel, A. (2014). *Verfahrensanleitung für die ökologische Bewertung von Seen zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos* (Version Ok). Länderarbeitsgemeinschaft Wasser (LAWA).

Schlüpmann, M. (2005). Bestimmungshilfen: Faden- und Teichmolch-Weibchen, Braunfrösche, Wasser- oder Grünfrösche, Eidechsen, Schlingnatter und Kreuzotter, Ringelnatter- Unterarten. *Rundbrief zur Herpetofauna von Nordrhein-Westfalen*, *28*, 1–38. Retrieved from http://www.herpetofauna-nrw.de

Sears, J. (1988). Regional and seasonal variations in lead poisoning in the Mute Swan (Cygnus olor) in relation to the distribution of lead and lead weights, in the Thames area, England. *Biological Conservation*, *46*, 115–134.

Shulse, C. D., Semlitsch, R. D., Trauth, K. M., & Williams, A. D. (2010). Influences of design and landscape placement parameters on amphibian abundance in constructed wetlands. *Wetlands*, *30*, 915–928. https://doi.org/10.1007/s13157-010-0069-z

Šidák, Z. (1967). Rectangular confidence regions for the means of multivariate normal distributions. *Journal of the American Statistical Association*, *62*(318), 626–633. https://doi.org/10.1080/01621459.1967.10482935

Somers, C. M., Heisler, L. M., Doucette, J. L., Kjoss, V. A., & Brigham, R. M. (2015). Lake use by three avian piscivores and humans : Implications for angler perception and conservation. *Journal of Open Ornithology*, *8*, 10–21.

Sørensen, T. A. (1948). A method of establishing groups of equal amplitude in plant sociology based on similarity of species content and its application to analyses of the vegetation on Danish commons. *Biol. Skar.*, *5*(4), 1–34.

Spohn, M., Golte-Bechtle, M., & Spohn, R. (2015). *Was blüht denn da?* (59. Auflag). Stuttgart, Germany: Kosmos-Verlag.

Spyra, A., & Strzelec, M. (2019). The implications of the impact of the recreational use of forest mining ponds on benthic invertebrates with special emphasis on gastropods. *Biologia*, 1–12. https://doi.org/10.2478/s11756-019-00221-2

Syväranta, J., Cucherousset, J., Kopp, D., Crivelli, A., Céréghino, R., & Santoul, F. (2009). Dietary breadth and trophic position of introduced European catfish Silurus glanis in the River Tarn (Garonne River basin), Southwest France. *Aquatic Biology*, *8*(2), 137–144. https://doi.org/10.3354/ab00220

Temple, H. J., & Cox, N. A. (2009). *European Red List of Amphibians*. Luxembourg: Office for Official Publications of the European Communities.

Theis, S. (2016). *Typology of German angling clubs in relation to means to manage local fisheries. Student project at the Humboldt-Universität zu Berlin* (Humboldt-Universität zu Berlin). Retrieved from https://www.ifishman.de/fileadmin/user_upload/Publikationen/Studie/Student_Project_Theis.pdf

Trochet, A., Moulherat, S., Calvez, O., Stevens, V., Clobert, J., & Schmeller, D. (2014). A database of life-history traits of European amphibians. *Biodiversity Data Journal*, *2*, e4123. https://doi.org/10.3897/BDJ.2.e4123

Van de Weyer, K., Meis, S., & Krautkrämer, V. (2015). Die Makrophyten des Großen Stechlinsees, des Wummsees und des Wittwesees. *Fachbeitrag Des LUGV*, *145*, 1–92.

Van de Weyer, K., & Schmitt, C. (2011). Bestimmungsschlüssel für die aquatischen Makrophyten (Gefäßpflanzen, Armleuchteralgen und Moose) in Deutschland: Band 1: Bestimmungsschlüssel. *Fachbeiträge Des LUGV Brandenburg*, *119*, 164pp. https://doi.org/10.1109/CDC.2012.6427112

Van der Maarel, E. (1979). Transformation of cover-abundance values in phytosociology and its effects on community similarity. *Vegetatio*, *39*(2), 97–114. https://doi.org/10.1007/bf00052021

Venohr, M., Langhans, S. D., Peters, O., Hölker, F., Arlinghaus, R., Mitchell, L., & Wolter, C. (2018). The underestimated dynamics and impacts of water-based recreational activities on freshwater ecosystems. *Environmental Reviews*, *26*, 199–213. https://doi.org/10.1139/er-2017-0024

Vilizzi, L., Tarkan, A. S., & Copp, G. H. (2015). Experimental evidence from causal criteria analysis for the effects of Common Carp (Cyprinus carpio) on freshwater ecosystems: A global perspective. *Reviews in Fisheries Science and Aquaculture*, *23*(3), 253–290. https://doi.org/10.1080/23308249.2015.1051214

Wichmann, G. (2010). Disturbance from angling during the breeding season: Its influence on waterbirds and reedbed birds in the Untere Lobau (Danube Forests National Park). *Egretta*, *51*, 108–113. Retrieved from http://www.landesmuseum.at/pdf_frei_baende/31364.pdf#page=110

Wilson, A. M., Barr, J., & Zagorski, M. (2017). The feasibility of counting songbirds using unmanned aerial vehicles. *The Auk*, *134*(2), 350–362. https://doi.org/10.1642/auk-16-216.1

Winfield, I. J. (2004). Fish in the littoral zone: Ecology, threats and management. *Limnologica*, *34*, 124–131. https://doi.org/10.1016/S0075-9511(04)80031-8

Wolter, C., & Arlinghaus, R. (2003). Navigation impacts on freshwater fish assemblages: The ecological relevance of swimming performance. *Reviews in Fish Biology and Fisheries*, *13*, 63–89. https://doi.org/10.1023/A:1026350223459

Wright, S. W. (1991). Improved HPLC method for the analysis of chlorophylls and carotenoids from marine phytoplankton. *Marine Ecology Progress Series*, *77*(2–3), 183–196. https://doi.org/10.3354/meps077183

WWF. (2018). *Living planet report - 2018: Aiming higher* (M. Grooten & R. E. . Almond, Eds.). Retrieved from https://www.wwf.de/fileadmin/user_upload/living-planet-report/2018/WWF_Living_Planet_Report_Englische_Version.pdf

Yoccoz, N. G., Nichols, J. D., & Boulinier, T. (2001). Monitoring of biological diversity in space and time. *Trends in Ecology & Evolution*, *16*(8), 446–453. https://doi.org/10.1016/S0169-5347(01)02205-4

Zhang, J., Nielsen, S. E., Grainger, T. N., Kohler, M., Chipchar, T., & Farr, D. R. (2014). Sampling plant diversity and rarity at landscape scales: Importance of sampling time in species detectability. *PLoS ONE*, *9*(4), e95334. https://doi.org/10.1371/journal.pone.0095334

Zhao, T., Grenouillet, G., Pool, T., Tudesque, L., & Cucherousset, J. (2016). Environmental determinants of fish community structure in gravel pit lakes. *Ecology of Freshwater Fish*, *25*(3), 412–421. https://doi.org/10.1111/eff.12222

Tables

Table 1: Descriptors of gravel pits sampled in Lower Saxony. Trophic state was determined using Riedmüller, Hoehn, & Mischke (2013).

No.	Lake name	Lake type	water-shed	End of mining	Trophic state	Maximum depth [m]	Lake area [ha]	pH value	Conductivity [S/cm]	Main recreationists and activities identified during on-site visits
01	Chodhemster Kolk	Managed	Ems	1971	Mesotrophic	10.1	3.2	6.7	208	dog walkers
02	Collrunge	Managed	SNST[1]	1982	Mesotrophic	8.6	4.3	7.9	200	Anglers, dog walkers
03	Donner Kiesgrube 3	Managed	Weser	2000	Eutrophic	5.2	1.0	7.8	632	cyclists
04	Kiesteich Brelingen	Managed	Weser	1999	Mesotrophic	8.7	8.5	7.5	299	Anglers, dog walkers
05	Kolshorner Teich	Managed	Weser	1980	Mesotrophic	16.1	4.3	7.6	547	Anglers, horses
06	Linner See	Managed	Ems	2000	Mesotrophic	11.2	17.7	8.5	301	Anglers
07	Meitzer See	Managed	Weser	2006	Oligotrophic	23.5	19.5	7.9	618	Anglers, dog walkers, swimmers
08	Neumanns Kuhle	Managed	Ems	1970	Polytrophic	6.2	6.9	8.6	585	Anglers, dog walkers
09	Plockhorst	Managed	Weser	1998	Eutrophic	8.2	14.3	8.0	338	Anglers, dog walkers, horses
10	Saalsdorf	Managed	Weser	1995	Mesotrophic	9.2	9.0	9.0	566	Anglers, dog walkers
11	Schlepturper See	Managed	Ems	1965	Mesotrophic	10.1	4.0	8.2	502	Anglers, dog walkers
12	Stedorfer Baggersee	Managed	Weser	1983	Eutrophic	2.8	1.9	7.5	352	Anglers, dog walkers
13	Steinwedeler Teich	Managed	Weser	1978	Mesotrophic	9.1	10.4	7.5	742	Anglers, dog walkers, cyclists
14	Wahle	Managed	Weser	1990	Mesotrophic	12.1	8.1	7.6	728	Anglers, dog walkers, cyclists, horses
15	Weidekampsee	Managed	Weser	1994	Oligotrophic	4.3	2.9	8.0	461	Anglers, dog walkers
16	Wiesedermeer	Managed	SNST[1]	1990	Mesotrophic	9.2	2.9	8.0	136	Anglers, dog walkers
17	Bülstedt	Unmanaged	Weser	1991	Polytrophic	1.1	2.4	7.6	285	Birdwatchers
18	Goldbeck	Unmanaged	Elbe	1992	Eutrophic	5.0	2.3	7.9	293	Owner/friends, horses, swimmers
19	Handorf	Unmanaged	Weser	2004	Eutrophic	23.0	13.6	9.2	666	Horses, dogs (commercial)
20	Hänigsen	Unmanaged	Weser	2011	Mesotrophic	12.3	6.2	8.6	437	Swimmers, dog walkers, anglers, campfires, dumping ground
21	Heeßel	Unmanaged	Weser	1963	Eutrophic	7.4	0.9	8.2	537	No
22	Hopels	Unmanaged	SNST[1]	1998	Mesotrophic	14.5	5.5	7.7	215	Swimmers, dog walkers
23	Lohmoor	Unmanaged	Weser	1991	Eutrophic	7.4	4.1	8.2	159	Birdwatchers
24	Pfütze	Unmanaged	Weser	2000	Mesotrophic	7.3	10.6	7.7	342	Dog walkers, canoes, swimmers, cyclists
25	Schwicheldt	Unmanaged	Weser	2007	Mesotrophic	10.0	1.7	8.4	895	Owner/hunter, dog walkers
26	Xella	Unmanaged	Weser	1975	Mesotrophic	7.3	2.1	7.5	957	No (restricted access)

[1] Small North Sea Tributaries

Table 2: Ranking of Red List categories used for calculation of conservation values.

Red List categories of Lower Saxony	**Rank r**	**Weight c**
1 – critically endangered	4	16
2 – endangered R – rare	3	8
3 – vulnerable G – indeterminate	2	4
V – near threatened	1	2
* – least concern - – data deficient	0	1

Table 3: Univariate comparison of environmental variables between managed and unmanaged gravel pit lakes. P-values are Sidak-corrected to account for multiple comparisons within classes of environmental variables, **significant ones ($p < 0.05$) are bolded**, *statistical trends ($p < 0.1$) are in italics*.

Class	Environmental variable (abbreviation)	mean ± standard deviation (minimum - maximum) Managed (N = 16)	Unmanaged (N = 10)	Comparison Test *	Statistic	p-value
Morphology	maximum depth in m (MaxDep)	9.7 ± 4.9 (2.8-23.5)	9.5 ± 6.0 (1.1-23.0)	U-test	W = 89	0.986
	lake area in ha (LArea)	7.4 ± 5.6 (1.0-19.5)	4.9 ± 4.2 (0.9-13.6)	U-test	W = 105	0.592
	shoreline development factor (SDF)	1.5 ± 0.3 (1.1-2.2)	1.6 ± 0.3 (1.3-2.2)	t-test	t = -1.0	0.824
	relative depth ratio (RelDepR)	0.04 ± 0.01 (0.02-0.07)	0.04 ± 0.02 (0.01-0.07)	t-test	t = -1.1	0.754
Trophic state	total phosphorus in µg/l (TP)	25.7 ± 36.5 (8-160)	27.2 ± 20.9 (12-72)	U-test	W = 63.5	0.983
	total organic carbonin mg/l (TOC)	6.6 ± 2.8 (2.5-13)	6.0 ± 2.5 (2.9-12.4)	U-test	W = 93	0.997
	mean chlorophyll a in µg/l (CHLa)	11.6 ± 15.9 (2.05-65.3)	21.2 ± 26.1 (2.6-90.6)	U-test	W = 53	0.765
	Secchi depth in m (Secchi)	2.6 ± 1.5 (0.5-5.5)	2.0 ± 1.1 (0.5-4.5)	t-test	t = 0.9	0.972
	ammonium in µg/l (NH4)	56.9 ± 71.2 (15.0-240.0)	84.0 ± 164.8 (15.0-550.0)	U-test	W = 75	1.000
	nitrate in µg/l (NO3)	283.8 ± 380.4 (5.0-1040.0)	733.0 ± 984.1 (5.0-2940.0)	U-test	W = 50	0.604
	conductivity in mS/cm (Con)	0.5 ± 0.2 (0.1-0.7)	0.5 ± 0.3 (0.2-1.0)	t-test	t = -0.3	1.000
	pH value (pH)	7.9 ± 0.5 (6.7-9)	8.1 ± 0.5 (7.5-9.2)	t-test	t = -1	0.967
Habitat structure	volume-% of simple dead wood (SDW_Vol)	0.005 ± 0.009 (0-0.035)	0.008 ± 0.010 (0.001-0.028)	U-test	W = 65.5	0.973
	volume-% of coarse woody structure (CWS_Vol)	1.5 ± 1.4 (0.02-5.6)	1.7 ± 2.0 (0.02-6.2)	U-test	W = 86.5	1.000
	mean riparian tree coverage on an ordinal scale from 0 to 4 (Rip_Trees)	1.0 ± 0.2 (0.4-1.5)	0.9 ± 0.3 (0.4-1.2)	t-test	t = 0.9	0.942
	mean litoral reed coverage on an ordinal scale from 0 to 4 (Reed)	1.3 ± 0.9 (0-2.5)	0.8 ± 0.7 (0-1.7)	t-test	t = 1.25	0.779
	mean riparian vascular plants coverage on an ordinal scale from 0 to 4 (Rip_Herbs)	1.7 ± 0.4 (1.1-3.0)	1.0 ± 0.7 (0.1-1.9)	U-test	W = 118	0.225
	submerged macrophyte coverage in the littoral zone in % (MP_Cov)	*39.3 ± 19.9 (12.5-82.3)*	*21.1 ± 27.5 (0-85.2)*	*U-test*	*W = 126*	*0.083*
Age	Lake age in years by 2017 (Age)	29.4 ± 12.4 (11-52)	23.8 ± 14.7 (6-54)	t-test	t = 1.05	0.303

* t-test = Student's t-test / U-test = Mann-Whitney-U-test

Table 4: Univariate comparison of environmental variables between managed and unmanaged gravel pit lakes. P-values are Sidak-corrected to account for multiple comparisons within classes of environmental variables, **significant ones (p < 0.05) are bolded**, *statistical trends (p < 0.1) are in italics*.

Class	**Environmental variable (abbreviation)**	**mean ± standard deviation (minimum - maximum)**		**Comparison**		
		Managed (N = 16)	**Unmanaged (N = 10)**	**Test** *	**Statistic**	**p-value**
Land use	excavation within 100m-buffer in % (Excav)	4.8 ± 7.2 (0-21.3)	9.4 ± 14.6 (0-39.0)	U-test	W = 63	0.718
	agriculture within 100m-buffer in % (Agric)	22.8 ± 19.9 (2.4-55.9)	33.8 ± 25.4 (3.5-79.0)	U-test	W = 58	0.598
	forest within 100m-buffer in % (Forest)	22.0 ± 25.1 (0-72.6)	5.6 ± 6.0 (0-15.5)	U-test	W = 118	0.132
	urban area within 100m-buffer in % (Urban)	27.8 ± 29.2 (0-87.5)	17.4 ± 24.5 (0-59.5)	U-test	W = 99	0.767
Water	wetland within 100m-buffer in % (Wetland)	0.9 ± 3.6 (0-14.4)	5.1 ± 14.1 (0-45.1)	U-test	W = 61.5	0.504
	water surface within 100m-buffer in % (Water)	9.7 ± 12.5 (0.9-50.4)	8.7 ± 8.8 (0.5-30.1)	U-test	W = 80	1.000
	distance to next lake in m (DistLake)	164.1 ± 236.4 (5-850)	264.1 ± 440.6 (1-1280)	U-test	W = 87	0.999
	distance to next river in m (DistRiver)	5226.1 ± 9805.2 (25-29,900)	3999.5 ± 9841.6 (220-31,920)	U-test	W = 92	0.980
	distance to next canal in m (DistCanal)	312.4 ± 462.3 (1-1630)	224.5 ± 367.9 (5-1180)	U-test	W = 84	1.000
Human presence	distance to next road in m (DistRoad)	265.3 ± 314.4 (15-1010)	558.0 ± 510.1 (30-1530)	U-test	W = 50.5	0.416
	distance to next settlement in m (DistVille)	504.1 ± 407.8 (20-1400)	801.0 ± 673.0 (60-1810)	t-test	t = -1.4	0.530
	distance to next city in m (DistCity)	7135.0 ± 4087.6 (170-13,130)	5859.0 ± 4488.3 (1070-15,110)	t-test	t = 0.8	0.917
Recreational use	**litter related to angling in numbers per m shore (A_Lit)**	**0.05 ± 0.05 (0-0.20)**	**0.002 ± 0.007 (0-0.021)**	**U-test**	**W = 140.5**	**0.007**
	litter unrelated to angling in numbers/m shore (NonA_Lit)	0.70 ± 0.50 (0.02-1.48)	0.34 ± 0.71 (0-2.29)	U-test	W = 126	0.124
	angling-sites and open spaces in % of shoreline(open_sites)	**18.5 ± 19.8 (3.6-87.7)**	**8.4 ± 14.4 (0-48.6)**	**U-test**	**W = 133**	**0.044**
	trails and paths per shoreline in m/m (Trails)	**0.9 ± 0.1 (0.6-1.1)**	**0.4 ± 0.5 (0-1.4)**	**U-test**	**W = 138**	**0.019**
	anglers per lake per visit (Anglers)	**1.6 ± 1.6 (0-5.1)**	**0.1 ± 0.2 (0-0.8)**	**U-test**	**W = 143**	**0.006**
	dog walkers per lake per visit (Dogs)	1.7 ± 1.9 (0-6)	0.5 ± 1.0 (0-3.3)	U-test	W = 123.5	0.154
	swimmers per lake per visit (Swimmers)	*2.9 ± 2.6 (0-10)*	*0.7 ± 1.0 (0-3.1)*	*U-test*	*W = 129.5*	*0.075*
	other recreationists per lake per visit (other_people)	*2.9 ± 3.2 (0.3-11.9)*	*0.9 ± 1.4 (0-3.8)*	*U-test*	*W = 128.5*	*0.087*

* t-test = Student's t-test / U-test = Mann-Whitney-U-test

Table 5: Overview about unique species of different taxa found at managed and unmanaged gravel pits in Lower Saxony, Germany. The numbers in brackets refer to single-lake observations, i.e. the number of species found at only one lake each.

Taxon	**Species number found only in …**		**Sørensen index (similarity)**
	…managed lakes	**…unmanaged lakes**	
submerged macrophytes	15 (13)	10 (9)	0.58
riparian vascular plants	55 (35)	31 (23)	0.73
riparian trees	6 (4)	5 (4)	0.86
amphibians	0 (0)	0 (0)	1.00
Odonata	5 (3)	8 (4)	0.76
songbirds	9 (7)	6 (5)	0.74
waterfowl	10 (3)	6 (3)	0.69

Table 6: Univariate comparison of species richness, Simpson index, threatened species and taxon-specific conservation values in managed and unmanaged gravel pit lakes. **Statistical differences of Sidak-corrected p-values < 0.05 are bolded.**

Diversity measure	Taxa	mean ± standard deviation (minimum - maximum) managed (N = 16)	unmanaged (N = 10)	Test *	Comparison statistic	p-value
species richness	submerged macrophytes	6.4 ± 3.2 (2-14)	5.2 ± 3.6 (0-11)	t-test	t = 0.91	0.962
	riparian vascular plants	42.3 ± 12.6 (15-57)	49.2 ± 11.2 (30-64)	t-test	t = -1.43	0.718
	riparian trees	12.8 ± 2.1 (9-17)	12.6 ± 5.3 (3-24)	F-test	t = 0.08	1.000
	amphibians	1.6 ± 0.5 (1-2)	2.2 ± 0.8 (1-3)	U-test	W = 43	0.220
	- reproducing	1 ± 0.6 (0-2)	1.4 ± 0.8 (0-3)	U-test	W = 59	0.832
	Odonata	7.9 ± 2.8 (2-12)	9 ± 4.3 (4-18)	t-test	t = -0.77	0.985
	- damselflies	4.3 ± 1.3 (2-6)	4.4 ± 1.3 (3-7)	t-test	t = -0.29	1.000
	- dragonflies	3.7 ± 2.1 (0-7)	4.6 ± 3.4 (1-12)	t-test	t = -0.84	0.975
	songbirds	9.2 ± 2.8 (5-14)	11.3 ± 3 (7-17)	t-test	t = -1.81	0.452
	waterfowl	9.5 ± 2.8 (3-13)	9.1 ± 3.5 (2-13)	t-test	t = 0.32	1.000
Simpson index	submerged macrophytes	0.6 ± 0.2 (0.1-0.9)	0.5 ± 0.3 (0-1)	t-test	t = 0.32	1.000
	riparian vascular plants	0.9 ± 0.1 (0.8-0.9)	0.9 ± 0.0 (0.8-0.9)	U-test	W = 79	1.000
	riparian trees	0.7 ± 0.2 (0.2-0.8)	0.7 ± 0.1 (0.4-0.9)	U-test	W = 94	0.990
	amphibians	0.1 ± 0.1 (0-0.4)	0.3 ± 0.2 (0-0.6)	U-test	W = 46	0.382
	- reproduction	0.2 ± 0.4 (0-1)	0.2 ± 0.3 (0-1)	U-test	W = 69.5	0.997
	damselflies	0.5 ± 0.1 (0.3-0.6)	0.6 ± 0.1 (0.4-0.8)	t-test	t = -2	0.339
	dragonflies	0.6 ± 0.3 (0-1)	0.5 ± 0.4 (0-0.9)	U-test	W = 87	1.000
	songbirds	0.8 ± 0.0 (0.7-0.9)	0.9 ± 0.0 (0.8-0.9)	t-test	t = -2.41	0.156
	waterfowl	0.7 ± 0.1 (0.3-0.8)	0.7 ± 0.1 (0.4-0.9)	U-test	W = 81	1.000
threatened species	submerged macrophytes	1.3 ± 1.1 (0-4)	0.6 ± 1.0 (0-3)	U-test	W = 109	0.565
	riparian vascular plants	0.6 ± 0.7 (0-2)	0.5 ± 0.7 (0-2)	U-test	W = 88	0.999
	riparian trees	0 ± 0 (0-0)	0.1 ± 0.3 (0-1)	U-test	W = 72	0.848
	amphibians	0.4 ± 0.5 (0-1)	0.8 ± 0.4 (0-1)	U-test	W = 51	0.440
	Odonata	0.8 ± 0.8 (0-2)	1.3 ± 1.2 (0-4)	U-test	W = 61.5	0.929
	- damselflies	0.3 ± 0.4 (0-1)	0.1 ± 0.3 (0-1)	U-test	W = 92	0.963
	- dragonflies	0.6 ± 0.7 (0-2)	1.2 ± 1.2 (0-4)	U-test	W = 54.5	0.692
	songbirds	0.3 ± 0.5 (0-1)	0.5 ± 0.5 (0-1)	U-test	W = 65	0.958
	waterfowl	1.1 ± 1.0 (0-3)	0.9 ± 1.0 (0-3)	U-test	W = 88	1.000
conservation value	submerged macrophytes	2.0 ± 0.8 (1-3.5)	1.6 ± 1.2 (1-4.6)	U-test	W = 99.5	0.571
	riparian vascular plants	1.1 ± 0.2 (1-1.9)	1.0 ± 0.0 (1-1.1)	U-test	W = 94.5	0.978
	riparian trees	1.0 ± 0.0 (1-1)	1.0 ± 0.1 (1-1.3)	U-test	W = 72	0.848
	amphibians	1.7 ± 0.8 (1-2.5)	2.2 ±0.9 (1-4)	U-test	W = 61.5	0.924
	Odonata	1.5 ± 0.5 (1-2.3)	1.7 ±0.5 (1-2.6)	U-test	W = 59.5	0.901
	- damselflies	1.2 ± 0.4 (1-2.2)	1.1 ±0.2 (1-1.8)	U-test	W = 91.5	0.972
	- dragonflies	1.7 ± 0.9 (1-3.3)	2.2 ±1.0 (1-3.8)	U-test	W = 55.5	0.889
	songbirds	1.2 ± 0.2 (1-1.6)	1.2 ±0.1 (1-1.3)	U-test	W = 72.5	1.000
	waterfowl	2.2 ± 1.0 (1.1-3.9)	2.2 ±0.8 (1.2-3.2)	U-test	W = 79	1.000

* t-test = Student's t-test / F-Test = Welch-F-test / U-test = Mann-Whitney-U-test

Table 7: ANOVA results of forward selection of RDA models explaining species richness across taxa. Variables are ordered by their R^2_{adj}-value. **Significant variables (p < 0.05) are bolded**.

Modelling step	Variable	Variance explained	R^2_{adj} *	F-statistic	p-value
full model	**woody_habitat**	**14.7%**	**0.112**	**6.63**	**< 0.001**
	watershed	**14.8%**	**0.080**	**2.22**	**0.028**
	acidity	**6.9%**	**0.063**	**3.12**	**0.019**
	agricultural_extent	**9.1%**	**0.060**	**4.10**	**0.003**
	age	3.6%	0.050	1.63	0.165
	lake_steepness	**6.9%**	**0.043**	**3.12**	**0.015**
	management	2.3%	0.028	1.04	0.424
	vegetated_habitat	3.8%	0.024	1.73	0.145
	conductivity	4.2%	0.015	1.91	0.116
	nitrogen	3.7%	0.012	1.65	0.160
	wetland	2.3%	0.009	1.04	0.406
	lake_shallowness	3.5%	0.009	1.57	0.186
	inaccessibility	2.2%	0.000	1.00	0.431
	trophic_state	1.1%	-0.004	0.48	0.825
	rural	3.7%	-0.007	1.66	0.163
	general_recreational_use_intensity	2.1%	-0.011	0.96	0.466
	forest_extent	2.4%	-0.015	1.10	0.379
	distance_to_next_river	1.4%	-0.016	0.65	0.694
best model	**woody_habitat**	**14.7%**	**0.271**	**5.05**	**< 0.001**
	agricultural_extent	**11.8%**		**4.06**	**0.001**
	lake_steepness	**9.3%**		**3.20**	**0.004**

* R^2_{adj}-values are shown for single-variable-models and the best model. The full model has an R^2_{adj}-value of 0.445.

Figure legends

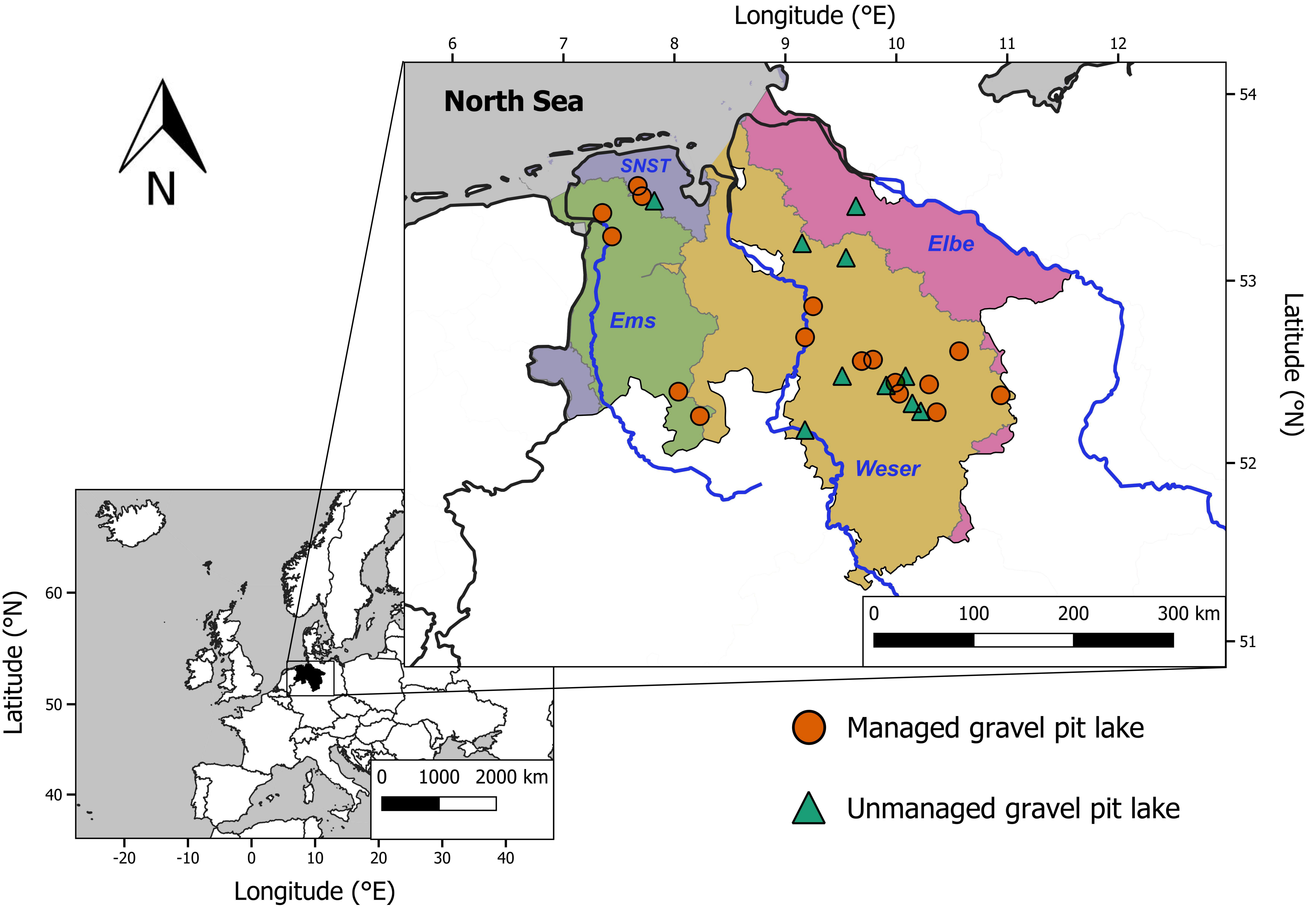
Figure 1: Map of study sites in Lower Saxony (Germany) together with the watersheds (green = Ems, orange = Weser, magenta = Elbe, blue = SNST/small North Sea tributaries) and main rivers (dark blue; Ems, Weser, Elbe).

Figure 2: Principal component analysis (PCA) by category of environmental variables visualized for habitat structure (SDW_Vol = volume-% of simple dead wood, CWS_Vol = volume-% of coarse woody structure, Rip_Trees = mean riparian tree coverage, Herb = mean riparian vascular plants coverage, Reed = mean litoral reed coverage, MP_Cov = submerged macrophyte coverage in the littoral zone; Table 3). Percentages in brackets show the proportional variance explained by each axis respectively. Numbers reflect the different lakes (Table 1). The centroids of management types are plotted as supplementary variables that did not influence the ordination. The 95% confidence-level around centroids are plotted to visualize differences between lake types.

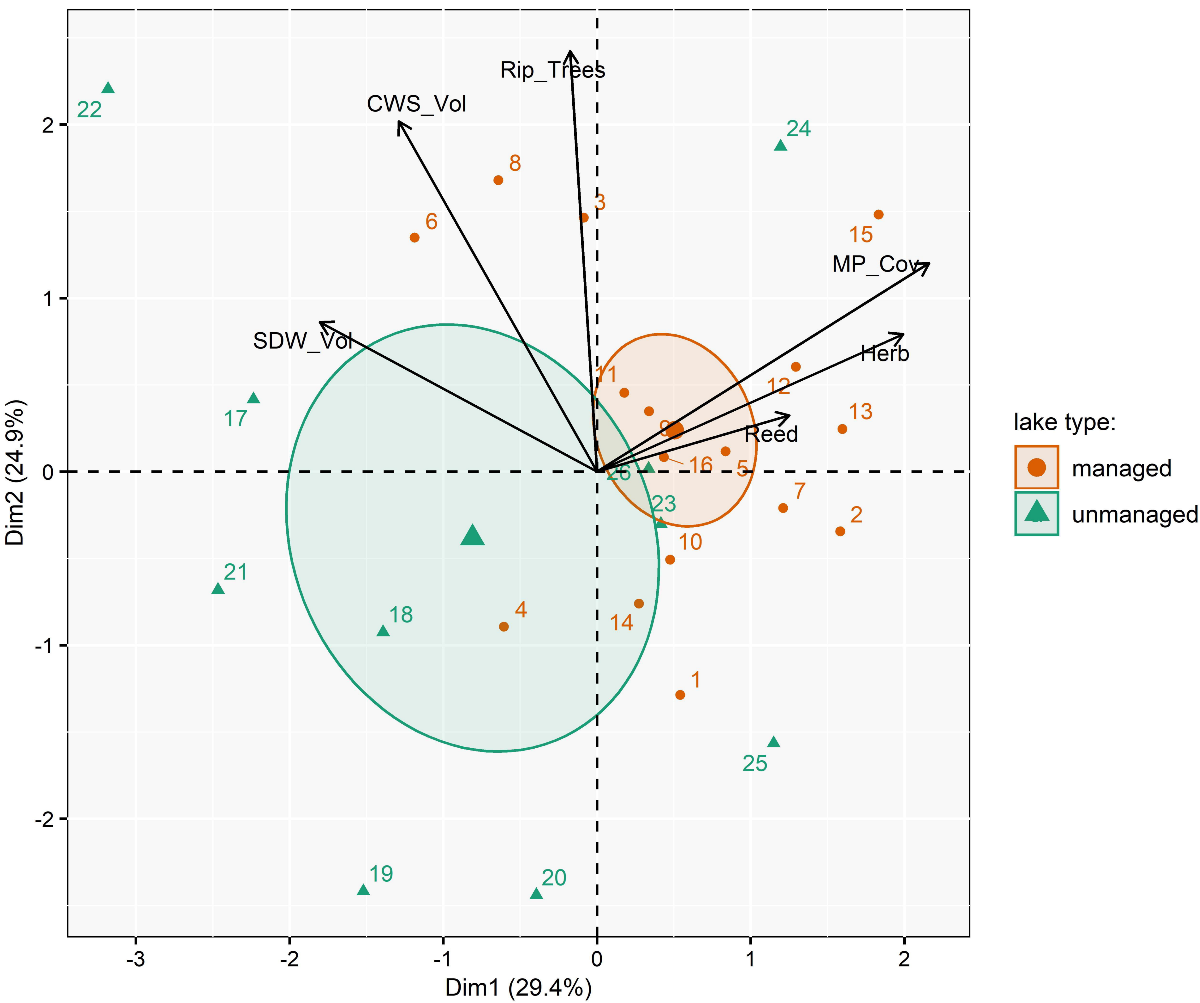
Figure 3: Principal component analysis (PCA) by category of environmental variables visualized for recreational use intensity (A_Lit = litter related to angling, NonA_Lit = litter unrelated to angling, open_sites = angling-sites and open spaces, Trails = trails and paths per shoreline, Anglers = angling people per visit, Dogs = dog walkers per visit, Swimmers = swimming people per visit, other_people = other recreationists per visit; Table 4). Percentages in brackets show the proportional variance explained by each axis respectively. Numbers reflect the different lakes (Table 1). The centroids of management types are plotted as supplementary variables that did not influence the ordination. The 95% confidence-level around centroids are plotted to visualize differences between lake types.

Figure 4: Principal component analysis (PCA) of species richness plotted for the first two axes (only relevant, i.e. highly contributing, variables are shown). Percentages in brackets show the proportional variance explained by each axis respectively. Numbers reflect the different lakes (Table 1). The centroids of management types and the explanatory variables from redundancy analysis (RDA, slashed purple lines, only the important ones for Dim1 and Dim2 are shown) are plotted as supplementary variables to not influence the ordination. The 95% confidence-level around centroids are plotted to visualize differences between lake types.

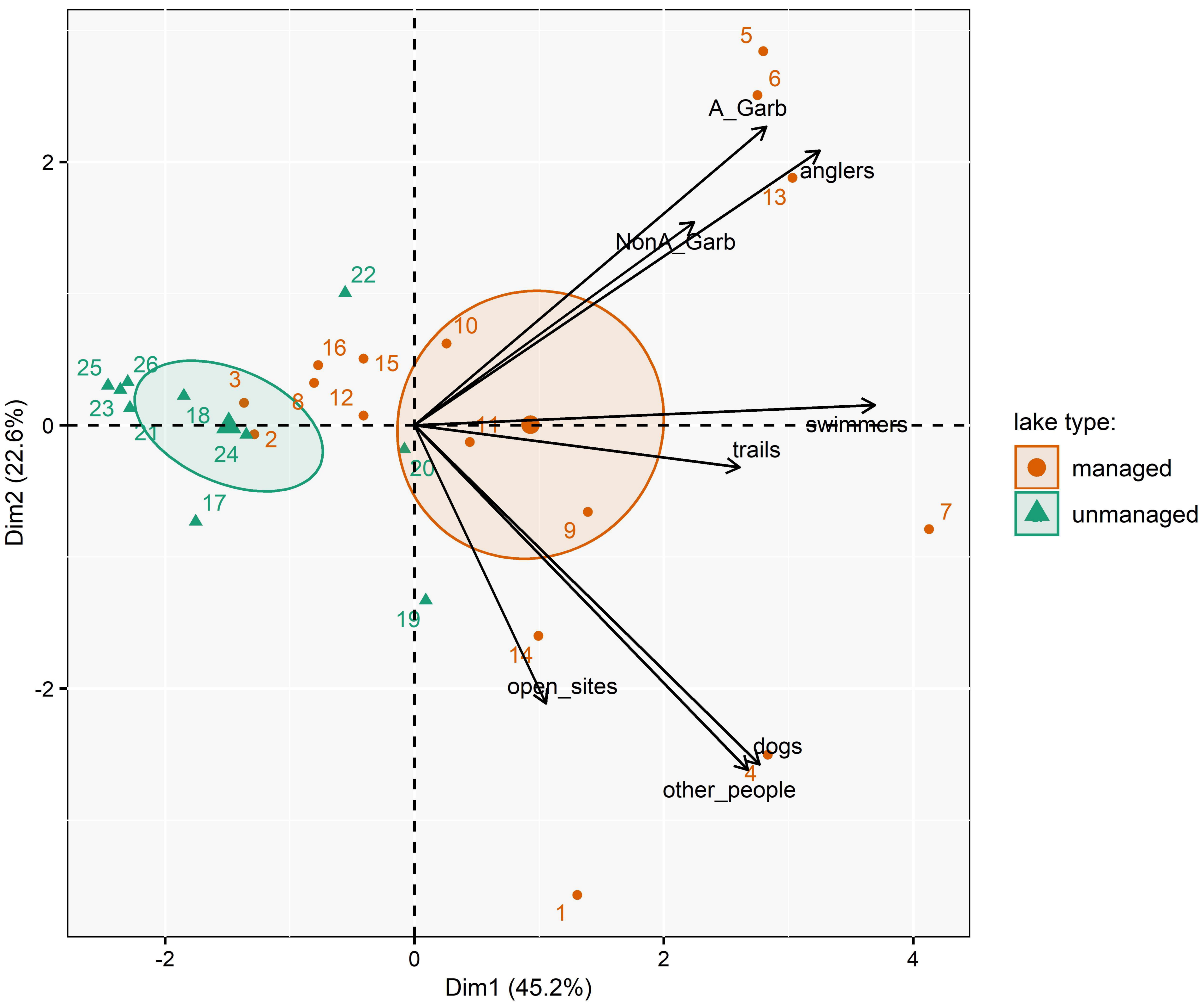
Figure 5: Principal component analysis (PCA) of species richness plotted for the third and fourth axis (only relevant, i.e. highly contributing, variables are shown). Percentages in brackets show the proportional variance explained by each axis respectively. Numbers reflect the different lakes (Table 1). The centroids of management types and the explanatory variables from redundancy analysis (RDA, slashed purple lines, only the important ones for Dim3 and Dim4 are shown) are plotted as supplementary variables to not influence the ordination. The 95% confidence-level around centroids are plotted to visualize differences between lake types.



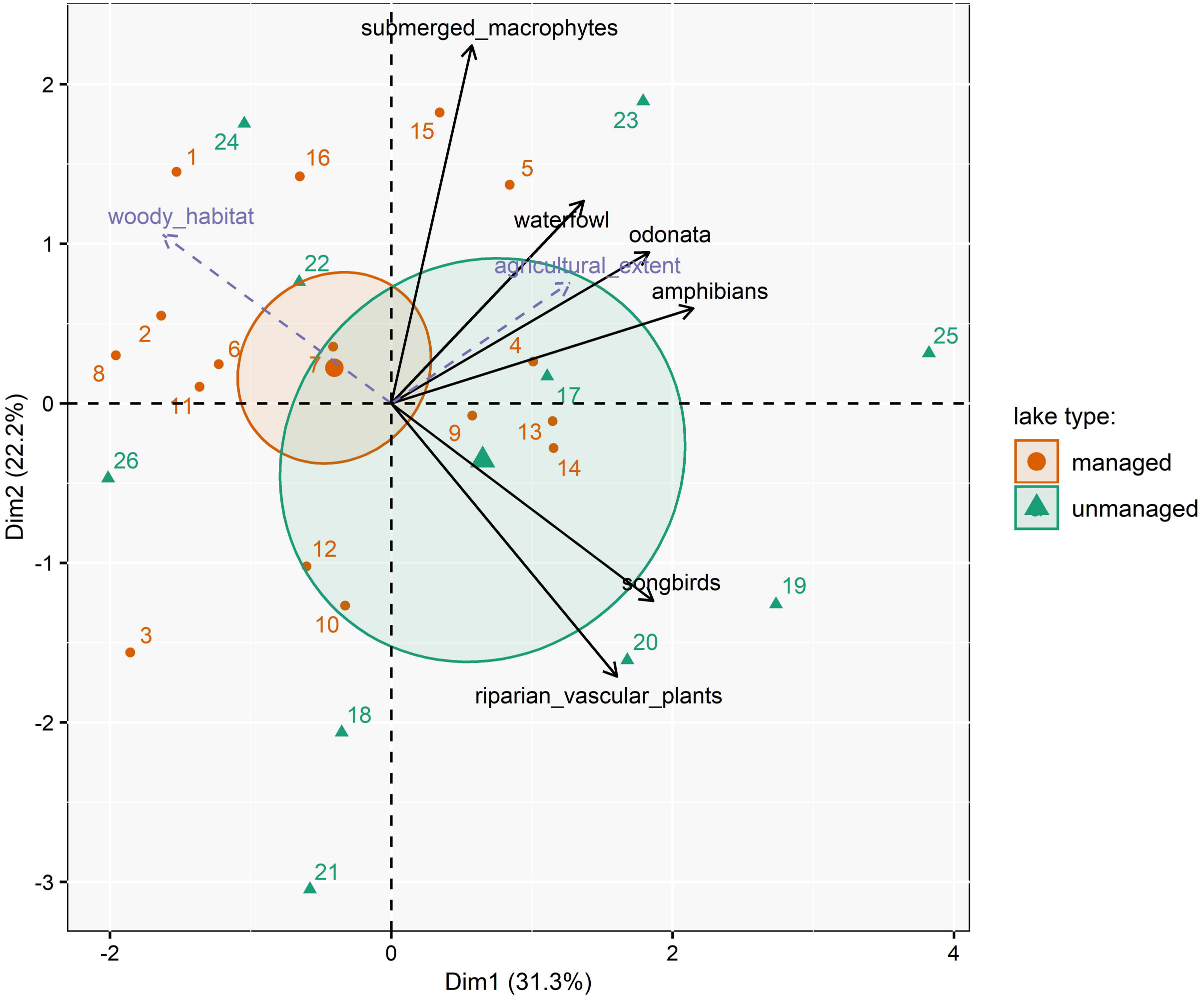
Longitude (°E)
6
7
8
9
10
11
12
54
53
52
51
Latitude (°N)
North Sea
N
SNST
Elbe
Ems
Weser
0
100
200
300 km
60
50
40
-20
-10
0
10
20
30
40
0
1000
2000 km
Managed gravel pit lake
Unmanaged gravel pit lake



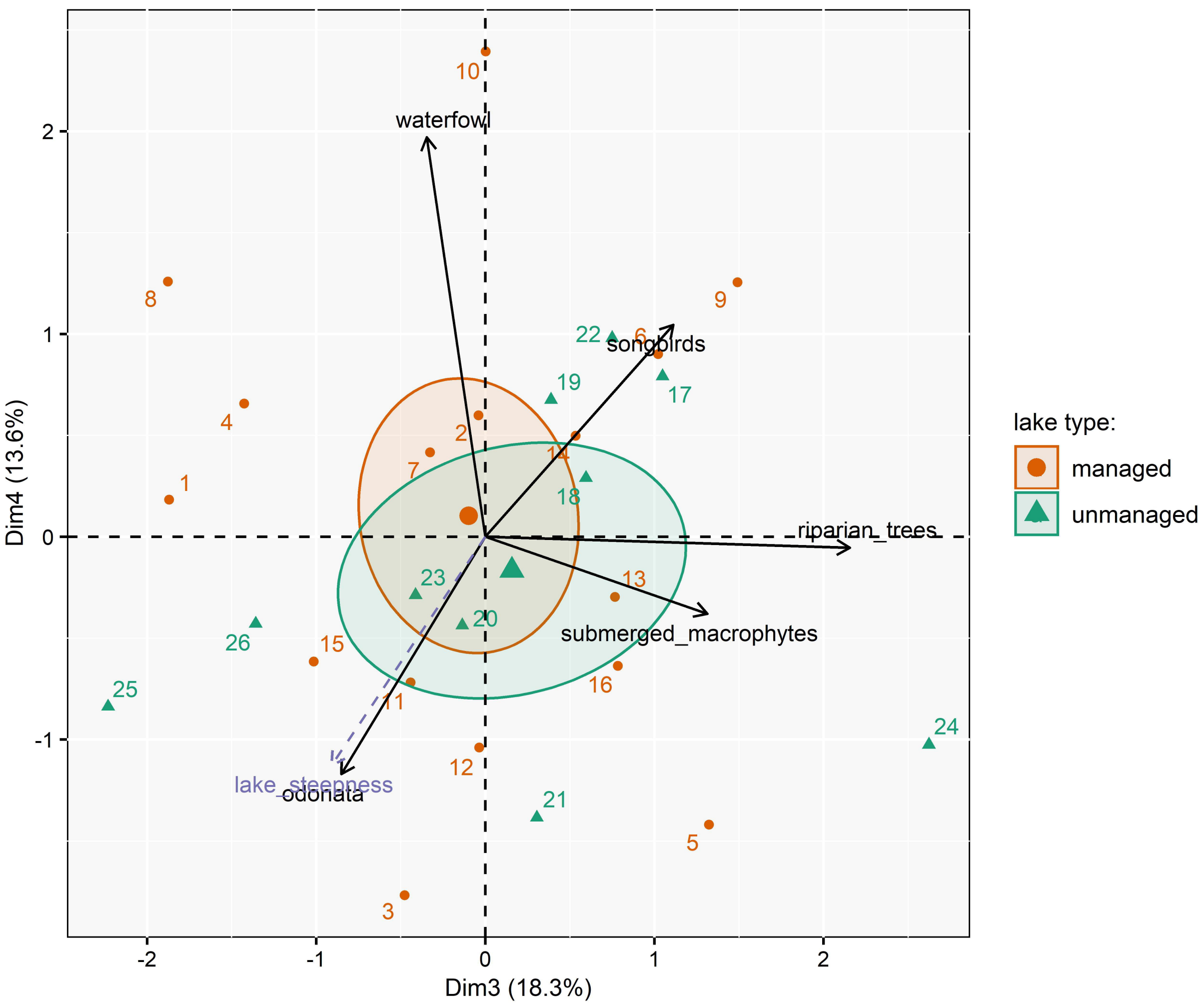
Rip_Trees
CWS_Vol
SDW_Vol
MP_Cov
Herb
Reed
Dim1 (29.4%)
Dim2 (24.9%)
lake type:
managed
unmanaged



A_Garb
anglers
NonA_Garb
swimmers
trails
open_sites
dogs
other_people
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
Dim1 (45.2%)
Dim2 (22.6%)
-2
0
2
4
lake type:
managed
unmanaged



submerged_macrophytes
waterfowl
odonata
agricultural_extent
amphibians
woody_habitat
songbirds
riparian_vascular_plants
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
22
23
24
25
26
Dim1 (31.3%)
Dim2 (22.2%)
-2
0
2
4
-3
-2
-1
0
1
2
lake type:
managed
unmanaged



10
waterfowl
2
8
9
1
22
6
songbirds
19
17
4
2
14
7
1
18
0
riparian_trees
23
13
20
submerged_macrophytes
26
15
25
16
11
24
-1
12
lake_steepness
odonata
21
5
3
-2
-1
0
1
2
Dim3 (18.3%)
Dim4 (13.6%)
lake type:
managed
unmanaged