

Student Project

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Amphibian biodiversity in Lower Saxonian gravel pit lakes in relation to recreational-fisheries management and habitat features

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Photo by Chente Ortiz

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Abstract

Amphibian biodiversity is declining globally, marked, among other processes, by changes in habitat features. Management of freshwater by recreational fisheries can alter the habitat of aquatic systems and have negative impacts on amphibian communities, i.e., fisheries stocking. However, the potential impact of recreational fisheries management is unclear, as various factors, many of which unrelated to fisheries management (e.g., agriculture), can contribute to the overall impact on amphibian biodiversity. Our project studied the ecological influence of recreational fisheries, as well as environmental factors on amphibian biodiversity and community structure in gravel pit lakes. We sampled 38 lakes in Lower Saxony, Germany, providing estimates of amphibian biodiversity and measures of habitat features. Lakes differed in management bodies (i.e. angler-managed vs. non-angler-managed). Results varied between management types: angler-managed lakes showed lower species richness than non-angler-managed ones. However, amphibian biodiversity was generally low, independent of management type. Fish biomass estimates were similar across lakes, suggesting similar predation rates between management types. Multivariate analyses identified the extent of shallow water (percentage of depth < 3 m) as a potential driver for amphibian biodiversity. We conclude that, while such lakes may support a wide variety of freshwater biodiversity, the effective conservation of amphibian biodiversity necessitates community-specific management and we suggest focusing efforts on vernal pools and other such fish-free wetlands.

Introduction

The 2004 IUCN Global Amphibian Assessment (GAA) outlines some sobering statistics describing the global status of amphibian biodiversity: 32.5% of all amphibians (roughly 1,856 species) are considered vulnerable by IUCN, among these are 427 species listed as critically endangered, and 122 species are to date unaccounted for in the literature since 1980; expected to be at risk or already extinct (Stuart et al. 2004). Current statistics from the IUCN confirm that these conditions have not improved since the first GAA, with well over 2000 species considered vulnerable and 588 considered critically endangered (IUCN 2020). The plight of amphibian decline can be largely attributed to freshwater habitat loss. Such ecosystems are considered the most threatened on the planet (Dudgeon et al. 2006). Within Europe, Stuart et al. (2004) explain much of the documented amphibian decline to “reduced-habitat”. Often the source of largescale habitat loss is tied to human landscape development, such as in southwestern Ontario; where depletions in amphibian abundance have been attributed to widespread deforestation in the mid 1800’s (Hecnar and M’Closkey 1998; Houlahan and Findlay 2003). The resulting land use around remaining habitat can lead to further degradation of amphibian communities.

Roadway development is considered a major threat to amphibians; directly responsible for mortality and restricting migration movements. Several studies have shown a negative correlation of roadway density with amphibian biodiversity (Houlahan and Findlay

2003; Hartel et al. 2010; Boissinot et al. 2019). Likewise, agriculture is responsible for large-scale changes in amphibian communities; stemming from changes in habitat structure and freshwater availability (Boissinot et al. 2019), as well as higher than normal nutrient loads in water bodies (i.e., nitrogen, Houlihan and Findlay 2003). Further studies have shown that the relative density of forest cover and freshwater bodies around breeding ponds can influence the abundance and richness of amphibians which access those ponds (Ficetola et al. 2009; Shulse et al. 2010; Sawatzky et al. 2019). These types of threats are systematic and generally represent human use and misuse of natural resources at a landscape scale. At the lake-level scale, factors such as lake morphology, aquatic vegetation, sunlight exposure, nutrient levels, and inter- and intraspecies interactions have been identified as important drivers for amphibian abundance (Houlihan and Findlay 2003; Shulse et al. 2010; Boissinot et al. 2019). Unfortunately, these threats (at both scales) are part of complex processes, involving multiple species with different ecological requirements and stretching across different spatial scales. Such circumstances make it difficult to understand to which degree each “scale” influences the overall biodiversity of amphibian communities, and furthermore, how factors affecting amphibians can co-vary and which factor may influence a given species. Therefore, correlations with observed biodiversity and range specific environmental variables should be analyzed with a large system in mind. While breeding ponds, may not represent the system as a whole, they are often used as representations of metapopulations for studying amphibian abundance (Marsh and Trenham 2001), and relationships with biodiversity at the pond level could have a significant impact on the overall biodiversity of a species or community of species.

This study concerns itself with the amphibian biodiversity within human-made gravel pit lakes in northwest Germany. These water bodies represent a large portion of the overall available freshwater habitat in this area (Matern et al. 2019) and may offer a substitute habitat for amphibians facing habitat degradation. In Germany (particularly western Germany), these water bodies are mostly privately owned and managed by angling clubs (Daedlow et al. 2011). This dynamic puts regional-scale angling clubs at the forefront of a complex struggle to achieve social, ecological, and economic sustainability (Arlinghaus et al. 2013): Gravel pit lakes have the conservation potential to serve as possible reservoirs for amphibian biodiversity, however the design and management of these lakes, which dictates this potential, depends upon the utility of the resource to all stakeholders in the community and often managers are forced to prioritize certain services at the cost of others (Blanchette and Lund 2016). Collaboration in this area is, from a conservation standpoint, crucial for the biodiversity conservation of amphibian communities. Furthermore, the societal and ecological role of recreational fishing is becoming (and has become) much more relevant in society (Arlinghaus et al. 2013; Arlinghaus and Cooke 2009) and is therefore a key determinant in management decisions concerning freshwater bodies in Germany. Fortunately, the ecological well-being of a lake and the social/economic benefit it affords are not mutually exclusive; steps are being taken to develop ecologically sustainable fisheries management (STÖRBAGGER, Schafft and Arlinghaus 2018; BAGGERSEE, ifishman.de 2019).

Gravel pits, in their design, were not intended as refuge for amphibian species, but rather for the extraction of raw materials (Blanchette and Lund 2016). For this reason, these lakes have relatively steep shores, and therefore limited shallows zones available to breeding amphibians. The potential influence of recreational fisheries on amphibian communities may come in the form of habitat alteration, such as removal or degradation of littoral vegetation and the construction fishing jetties or gravel beds. However, fisheries management is most generally concerned with managing fish stocks, and therefore stocking practices and harvesting are more likely to play a role in influencing amphibians (Nikolaus et al. 2020). The direct predation of fish on amphibians occurs naturally in nature and the influence of recreational fisheries does not seem to impact the total fish biomass in gravel pit lakes (Matern et al. 2019). Fishery-managed and non-fishery-managed gravel pit lakes exhibit similar fish biomasses, however stocking practices by fisheries managers can change the fish community composition, often resulting in a higher abundance of large-bodied or piscivorous species (Matern et al. 2019; Zhao et al. 2016). This change in fish species composition has the potential to negatively impact amphibian communities, as piscivorous fish abundances has been inversely correlated with amphibian abundances (Hecnar and M'Closkey 1997; Hartel et al. 2007). In addition to predatory species, large-bodied carps are another common feature of stocked lakes (Zhao et al. 2016). A high abundance of these species (i.e., *Cyprinus carpio*) can significantly impact macrophyte density (Bajer et al. 2016) and has also been associated with reductions in amphibian species richness (Kloskowski 2009). Lastly, the situation most commonly associated with amphibian decline, is the introduction of fish to otherwise fish-free waters. In such cases, populations which have adapted to a relatively predator-free environment can be quickly depleted; as is the case with alpine lake amphibian communities affected by trout and minnow introductions (Miró et al. 2018), particularly concerning paedomorphic newts (Denoel et al. 2005), which spend a significant portion of their lives in freshwater.

In order to establish common ground for amphibian species conservation within the framework of recreational fisheries management, the possible influence of recreational fisheries on amphibian communities must be analyzed. This project takes place within the framework of the STÖRBAGGER project (Schafft and Arlinghaus 2018) and provides estimates for amphibian biodiversity across 38 gravel pit lakes in Lower Saxony; an area for which recreational fisheries management practices are well documented (Matern et al. 2019), but for which the influence of recreational fisheries on amphibians is relatively unknown. Our estimates are analyzed with data for fish biodiversity from Matern et al. (2019) to determine the potential influence of recreational fishing and recreational fisheries management on amphibian biodiversity in Lower Saxonian gravel pit lakes. We predicted that managed lakes would display lower biodiversity than unmanaged lakes and that biodiversity would be inversely correlated with metabolic fish biomass. Furthermore, on the landscape scale, we expected to see a negative influence of urbanization and agriculturalization on amphibian abundances. The percentage of forest cover and wetlands around lakes were expected to be positively correlated with amphibian biodiversity. On the

habitat scale we expected the extent of reeds and the extent of shallow water to increase with amphibian biodiversity.

Methods

This work builds upon a previous framework and dataset developed by Nikolaus et al. (2020) and their methods in fieldwork and analysis have been adapted for use in this report. Additional data for fish biomass has been provided by Matern et al. (2019). The lake subset, described later in this report, includes the same set of lakes described by these two reports.

Fieldwork for this project was carried out in the state of Lower Saxony, in northwest Germany. 38 lakes were chosen based on previously established criteria from Nikolaus et al. 2020. Our sample includes 25 lakes managed by recreational fisheries and 13 unmanaged lakes (i.e., managed by other governing bodies). Each lake was sampled twice for amphibians, corresponding with the beginning and end of the breeding season and once for lake-level environmental metrics. Data collected stretch across 2017 to 2019 and all fieldwork was conducted in the spring between March and May.

Amphibian sampling consisted of daytime visual surveys along shorelines, nighttime recordings around lake perimeters, and adapted funnel traps set overnight within littoral transects. Individuals were only captured opportunistically as part of daytime surveys, otherwise quantified and identified by sight. Nighttime calls were recorded with a handheld audio recording device for further identification. Nylon mesh funnel traps were set in the evenings prior to nightly recordings and emptied the following day during shoreline visual surveys. All traps were fitted with styrofoam supports to provide air pockets for adult amphibians and baited each with a single Cormoran Lightstick (ca. 4cm). Species were identified using guidelines from Thiesmeier (2014). We refer to Podloucky and Fischer (2013) for a full species database for amphibians in Lower Saxony and Bremen. However, for our regional pool (Table 2) we only included species whose ranges overlapped the area our our study sites (range estimates from Amphibiaweb.org). Egg masses were quantified using assumed values from Trochet et al. (2014). Members of the green ranids were identified opportunistically to species, but considered for analysis as one species complex (*Pelophylax* sp.), because of difficulties in identification due to hybridization (Berger 1967). All individual sightings (including egg balls and lines) and traps were marked with GPS points (Garmin GPSMAP 62).

Lake-level habitat variables were recorded once for each lake during amphibian sampling periods and included pH, Secchi Depth, Conductivity and Max Depth, and the overall depth profile. Shoreline habitat was also recorded as individual GPS tracks and subdivided into six categories for shoreline habitat structure: Wood, Open, Reed, Herbal, Artificial, and Mixed. Shoreline length and the representative percentage of each habitat type was calculated in QGIS (version 3.4.9). Lake geometry, such as perimeter and area were also estimated in QGIS (version 3.4.9), and further lake-level factors, such as average depth and the overall depth profile were calculated from echo sounding data, processed and provided by Nikolaus et al. 2020, as well as separate surveys conducted as part of the ongoing STÖRBAGGER project (Schafft and Arlinghaus 2018).

Land use variables were calculated based on a reclassification of ATKIS land use data by Nikolaus et al. 2020: Land use classifications from the AdV (Working Committee of the Surveying Authorities of the Laender of the Federal Republic of Germany; GeoBasis-DE/BKG 2013) was pooled into six different land use classes: Urbanization, Agriculture, Forest, Wetland, Mining, and Water. A 100m buffer zone was drawn around each lake; from which the percent coverage for each land use feature was taken. All calculations for percent land use coverage used in this paper's analysis were completed using QGIS (versions 3.4.1 and 3.4.9) with GRASS (versions 7.4.2 and 7.6.1).

Statistical Analysis

All models and calculations, unless otherwise noted, were performed in R (R Core Team 2019) and all plots for PCA and RDA analysis (Figures 1-4) were created in R using the ggplot2 package (Wickham 2016).

Biodiversity metrics

Species richness (S) and relative abundance (Pi; from here on "community composition") were calculated for each lake, where values for richness include all species documented at all life stages and community composition is based solely upon adults and juveniles; excluding larval stages (i.e., tadpoles), as this stage differs significantly in terms of survival and overall reproductive potential. Shannon's diversity index (H) was also calculated based on the community composition. As our lakes were mostly dominated by one species (*Bufo bufo*), we chose to use Shannon's diversity index (Shannon 1948), as it gives equal weight to both rare and abundant species (Morris et al. 2014). All biodiversity measures were calculated using Microsoft Excel (Version 1908).

Fish metrics

For a subset of lakes, fish biomass was also included as an explanatory variable. Matern et al. (2019) provide electrofishing and multi-mesh-size gillnet data for 22 lakes assessed in our study (16 managed / 6 unmanaged) from 2016 through 2017. We first developed two metrics based on the metabolic biomass¹: one metric was calculated from electrofishing data by dividing the metabolic biomass (Total Length²) by the transect length (in 50m segments). From the gillnet data we calculated a metabolic biomass as a stratified mean per 100m² of net wall, in which pelagic nets (in which generally few or no fish were captured) were not included in calculating mean biomass. Furthermore, for both indices, a biomass subset was calculated for piscivorous fish species. Some studies have shown that a community piscivorous fish, introduced via stocking practices, can have deleterious effects on amphibian populations (Hartel et al. 2007; Hecnar and M'Closkey 1997) and we developed our subset based on the following species of fish, typically introduced as part of stocking procedure: Wels catfish (*Silurus glanis*), Northern pike (*Esox Lucius*) European perch (*Perca fluviatilis*)², Pike perch (*Sander lucioperca*), European eel (*Anguilla anguilla*), and one

¹ Metabolic biomass was used in place of other measures to account for exponential weight gain in fishes; giving more influence to heavier fish, despite only a minor increase in total length.

² Perch were included as known piscivorous predators, despite their wide abundance in both managed and unmanaged lakes (Matern et al. 2019).

ictalurid catfish (*Ictalurus sp.*). To select for a single biomass variable for use in further models, all four variables were tested in an RDA to predict amphibian community structure using an ANOVA-like permutation test (Legendre and Legendre 1998). Results indicated that our piscivorous subset from electrofishing data explained more variance compared to other metrics, and was therefore selected as our single fish variable (label = Pred_L2_50m).

Environmental variation

First and foremost, the overall variance in land use and lake-level variables between lake types was tested for significance using a simple redundancy analysis (RDA) and ANOVA-like permutation test (Legendre and Legendre 1998). Lake-level and land use variables were assessed separately using a principal component analyses (PCA: Mardia et al. 1979), in order to reduce the number of single variables and describe the variation in the dataset using a set of principal components. For a list of all environmental variables used in the lake-level and land use PCAs, see Tables 2 and 3, with descriptions in Table 5. To prevent effects of multicollinearity (i.e. the high positive or negative correlation between independent variables, which may cause uncertainty in final model predictions), we tested for correlations using bivariate plots (ggpairs from ggplot2; Wickham 2016). Factor scores from PCA analysis were used in RDAs to predict amphibian community structure and raw variables were used in GLMs to predict general amphibian biodiversity (i.e. Richness and Shannon's index).

Amphibian community composition

To explain the variation in lake-level abundance scores and test for species specific relationships with environmental variables, we ran a redundancy analysis (RDA: Legendre and Legendre 2012), testing the influence of selected variables (for both the full set and subset of lakes) on species community composition. Permutation tests were used to evaluate the significance of individual variables in the models.

Lake-level amphibian biodiversity

Generalized linear models (GLM) were used to test for the effect of selected variables on amphibian richness and Shannon's diversity index. Both land use and lake-level environmental variables were considered as explanatory variables. Alongside the non-fish environmental variables, fish biomass metrics were also considered for the lake subset.

Variable and model selection

RDA analysis: we used factor scores from the principal components (derived from our lake-level PCA and land use PCA) as explanatory variables in our RDA analysis to predict amphibian community structure. Principal components 1 and 2 were chosen from both PCAs and management type was included as a supplementary binomial variable. A second RDA was then run using a subset of lakes for which fish data were available and for this we used the piscivorous biomass metric described above as an extra explanatory variable. Table 1 below gives an overview of the principal components used in RDA analysis and their nicknames used in the model.

Table 1. Environmental variables used in RDAs (shown in Tables 3 and 4). These variables are themselves PC scores, derived from individual environmental variables in PCA analysis. This table shows which components are represented by which variables (i.e. which nicknames). “X” represents a singular variable, not PC scores.

Variable Nicknames	Principal Component	
	Lake-level	Land Use
management	X	X
wetland		PC2
open_pH	PC2	
Pred_L2_50m	X	X
depth	PC1	
development		PC1

GLMs: to predict lake-level richness, we ran two GLMs (using the function glm in R), one with a selection of variables from the set of non-fish variables (38 lakes) and one for the subset of lakes with fish data (22 lakes). Raw data, rather than PC scores from the PCA, was used for explanatory variables. The selected variables were mostly chosen based on our predictions from related studies. Urbanization and agricultural landscape were chosen as raw variables (percentage out of total buffer area) based on their high loadings in PCA. Additionally, reed habitat was chosen as an explanatory variable, as we found that many individual amphibians were found within this habitat feature.

We included relationships between certain variables (e.g., Management:Reed, Management:Urban). Below are the “full” models (i.e., with all variables and relationships) for each set of lakes respectively (see Table 5 for full list of abbreviations):

Full model -All lakes

glm (Richness ~Management*Reed + Management*Water + Management*Urban + Management*Agra + Management + Reed + Water + Forest + Percent_0.3m + Urban + Agra)

Dependent variable: *Richness*

Single predictor variables: *Management, Reed, Water, Forest, Percent_0.3m, Urban, Agra*

Interactions: *Management:Reed, Management:Water, Management:Urban, Management:Agra*

Full model -Lakes subset

glm (Richness Management*Reed + Management*Water + Management*Urban + Management*Agra + Management*Pred_L2_50m + Management + Reed + Water + Forest + Percent_0.3m + Urban + Agra + All_L2_50m + Pred_L2_50m)

Dependent variable: *Richness*

Single predictor variables: *Management, Reed, Water, Forest, Percent_0.3m, Urban, Agra, Pred_L2_50m, All_L2_50m*

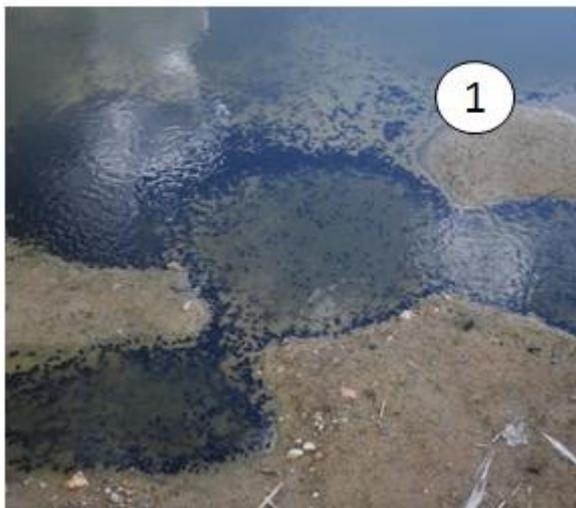
Interactions: *Management:Reed, Management:Water, Management:Urban, Management:Agra, Management:Pred_L2_50m*

To select our final models, we removed one variable (or interaction term) at a time and ran a separate GLM for each new iteration. The order in which each variable was removed was assumed to be irrelevant. For each new iteration we used Anova Tables to compare the present model to the previous model (in which each set of compared models differ in only one variable). Models were finally assessed and sorted by AIC scores, where the model with the lowest AIC score was selected.

Results

Table 6 describes the general appearance of our sample lakes, with the following key characteristics: our sample represents small, shallow lakes, mostly dominated by a woody or mixed shoreline habitat (mean relative coverage >30% and >29% respectively) and surrounded mainly by urban or agricultural landscapes (mean relative coverage >34% and >23% respectively). Lakes ranged from roughly 1-20 hectares in water surface area (mean = ca. 8 ha) with an average maximum depth of ca. 10 m (ranging from 1-23 m).

Amphibian biodiversity was generally low across all lakes (mean richness = 2, range 1-3). We identified 4 amphibian species found to inhabit gravel pit lakes, representing roughly 30% of our regional pool (Table 2). The common toad, *Bufo bufo*, was by far the most abundant species (frequency of occurrence > 89%), with a mean relative abundance (Pi) over 50%. The green ranids (*Pelophylax sp. complex*) were the second most abundant species (frequency of occurrence ca. 60%), with a mean relative abundance of ca. 33%. Common frogs (*Rana temporaria*) were only seen in five lakes and Smooth newts (*Lissotriton vulgaris*) only in two. Species profile images described below:



- 1) Dense aggregation of bufonid tadpoles (location: study site).
- 2) subadult Common frog, *Rana temporaria* (location: study site).
- 3) Adult Common Toad, *Bufo Bufo* (location: study site).
- 4) Green ranids, *Pelophylax sp.* (location: study site).
- 5) Juvenile Smooth newt, *Lissotriton vulgaris* (location: private garden pond, Berlin).



Photos by Chente Ortiz

Table 2. Regional pool of amphibian species likely to occur within our study area (n = 12), listed alphabetically (study sites cross referenced with species range estimates from Amphibiaweb.org). Notably, range estimates for the Agile frog did not cover the entire study area, while range estimates for the Yellow-bellied toad only overlapped with a single lake site near the northern border of North Rhine-Westphalia. Both species were nonetheless included in the list. In contrast, Fire salamanders, while occurring in the region, were not expected to be present in lowland lake habitat and were therefore excluded. Bold species were seen in our study.

Latin	German (common)	English (common)
<i>Bombina variegata</i>	Gelbbauchunke	Yellow-bellied toad
<i>Bufo bufo</i>	Erdkröte	Common toad
<i>Epidalea calamita</i>	Kreuzkröte	Natterjack toad
<i>Hyla arborea</i>	Europäischer Laubfrosch	Common tree frog
<i>Lissotriton helveticus</i>	Fadenmolch	Palmate newt
<i>Lissotriton vulgaris</i>	Teichmolch	Smooth newt
<i>Pelobates fuscus</i>	Knoblauchkröte	Common spadefoot
<i>Pelophylax sp.</i>	Wasserfrösche	Green ranids
<i>Rana arvalis</i>	Moorfrosch	Moor frog

<i>Rana dalmatina</i>	Springfrosch	Agile frog
<i>Rana temporaria</i>	Grasfrosch	Common frog
<i>Triturus cristatus</i>	Kammolch	Great crested newt

Environmental variation

PCA analysis was carried out for 15 lake-level and six land use variables (Figures 1, 2; raw loading scores shown in Table 3 and 4). In general, lake-level variables tested did not differ significantly between lake types (permutation tests revealed: F-statistic = 1.35, $P > 0.05$ for lake.level.variables ~ Management). PCA analysis on lake-level variables (Figure 1) shows significant overlap between ellipses and an uncorrelated distribution of variable loadings. In contrast, Figure 2 (land use PCA) exhibits a greater divide between management types, with more defined patterns of variable loading: managed lakes are surrounded by a more urbanized landscape, whereas unmanaged lakes seem to be situated within a more agricultural landscape (Figure 2). Both water bodies (excluding wetlands) and pit mines also seemed to be at higher concentration around unmanaged lakes. In contrast, forest cover did not seem to be significantly different among lake types, but rather mildly abundant around all lakes (mean of 14%, with a median around 8%; Table 6) Wetlands were only present around 6 of our lakes and their presence did not appear to be influenced by lake type.

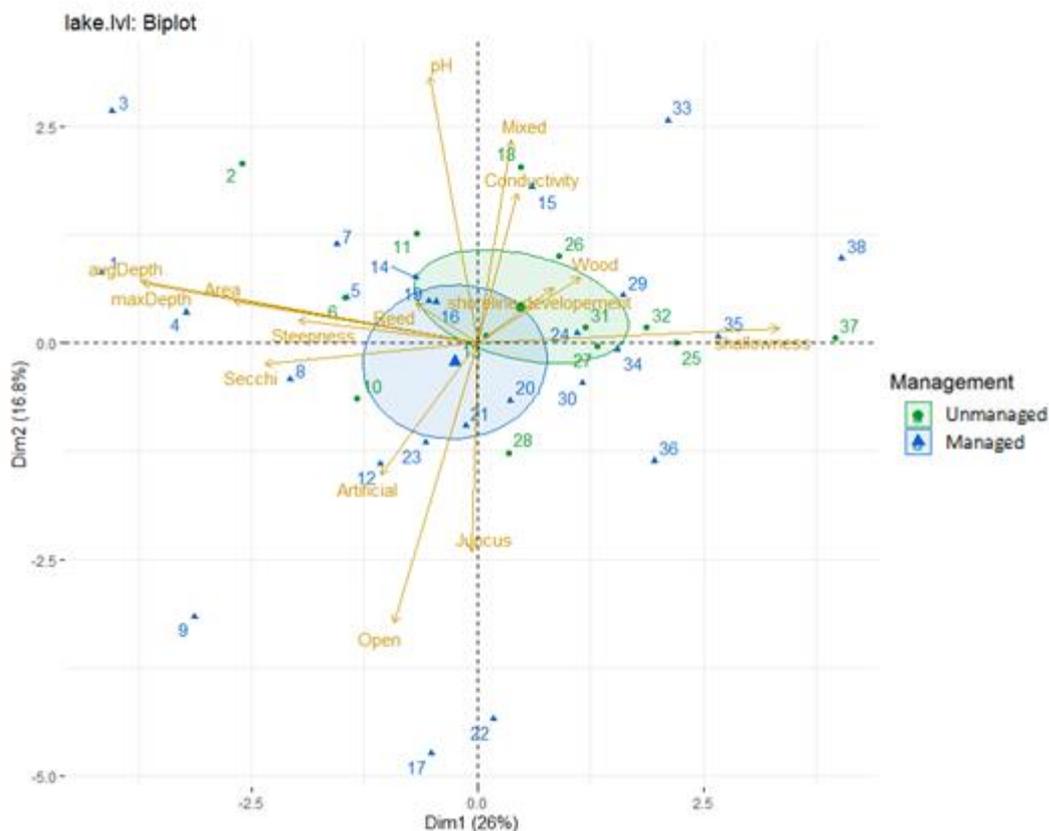


Figure 1. Principal component analysis (PCA) of lake-level environmental variables (gold, abbreviations shown in table 5). The x-axis is the first dimension of the PCA, explaining 26% of the total variance. The y-axis is the second dimension, explaining 16.8% of the variance. Numbers

represent the lake ID and management type (color coded) is considered as a supplementary variable. The ellipses around lake-type centers represent 95% confidence intervals. Differences among lake types are considered more significant if there is no overlap.

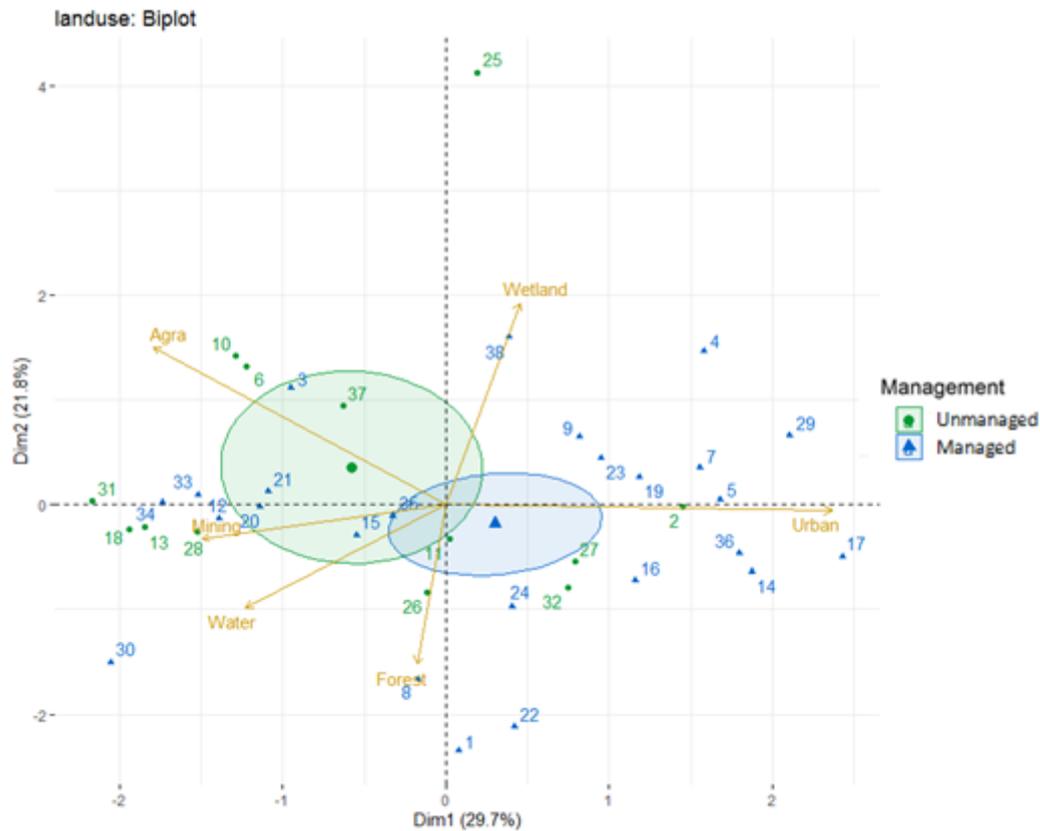


Figure 2. Principal component analysis (PCA) of land use variables (gold, abbreviations shown in table 5). The x-axis is the first dimension of the PCA, explaining 29.7% of the total variance. The y-axis is the second dimension, explaining 21.8% of the variance. Numbers represent the lake ID and management type (color coded) is considered as a supplementary variable. The ellipses around lake-type centers represent 95% confidence intervals. Differences among lake types are considered highly significant if there is no overlap.

Table 3. The most influential principal components (PCs) shown (eigenvalue > 1), representing > 78% of the total variance in lake-level variables. Eigenvalues show the amount of variance in the data represented by each corresponding component. Loading scores for each variable explain the influence and direction of the variable on each component. High loading scores (absolute value) correspond to a high influence on the axis. Abbreviations are described in Table 5.

Lake-level	PC1	PC2	PC3	PC4	PC5	PC6
Eigenvalue	3.906	2.518	1.765	1.439	1.180	1.016
Proportion of variance explained	0.260	0.168	0.118	0.096	0.079	0.068
Wood	0.146	0.122	-0.409	0.470	-0.342	-0.323
Open	-0.119	-0.517	0.160	0.121	0.037	0.132
Reed	-0.088	0.073	0.205	-0.669	-0.391	0.075
Binsen	-0.008	-0.385	0.315	0.210	0.047	0.075

Artificial	-0.138	-0.244	-0.241	-0.241	0.338	-0.475
Mixed	0.048	0.376	0.046	0.005	0.692	0.175
Area	-0.347	0.077	0.415	0.254	0.048	-0.212
SDF	0.107	0.101	0.350	-0.036	-0.056	-0.572
Conductivity	0.056	0.277	0.107	-0.200	-0.106	-0.059
Secchi	-0.302	-0.039	-0.289	-0.224	0.159	-0.372
pH	-0.068	0.492	0.122	0.189	-0.091	0.012
avgDepth	-0.480	0.114	0.043	0.053	0.031	0.066
maxDepth	-0.475	0.109	0.063	0.096	0.025	0.007
shallowness	0.429	0.027	0.059	-0.091	0.282	-0.029
Steepness	-0.254	0.042	-0.436	-0.052	-0.040	0.301

Table 4. The most influential principal components (PCs) shown (eigenvalue > 1), representing > 71% of the total variance in land use variables. Eigenvalues show the amount of variance in the data represented by each corresponding component. Loading scores for each variable explain the influence and direction of the variable on each component. High loading scores (absolute value) correspond to a high influence on the axis. Abbreviations are described in Table 5.

Land use	PC1	PC2	PC3
Eigenvalue	1.784	1.306	1.200
Proportion of variance explained	0.297	0.218	0.200
Forest	0.049	0.498	-0.728
Agra	0.501	-0.491	-0.088
Mining	0.418	0.105	0.358
Urban	-0.662	0.018	0.353
Wetland	-0.127	-0.630	-0.238
Water	0.343	0.321	0.390

Table 5. List and definition of all environmental variables used in PCA analysis.

<p>Habitat Variables (Wood, Open, Reed, Juncus, Artificial, and Mixed) represent the proportional shoreline habitat structure, excluding submerged habitat and limited to structures directly opposite or directly above the water's edge. Mixed habitat is defined as a combination of non-continuous habitat structures which are comprised of at least two different habitat variables. "Juncus" refers to plants of the genus <i>Juncus</i>, but for this study, may include various other herbaceous plants, included as one habitat variable.</p>
<p>Land Use Variables are percentages out of the total 100m buffer-zone area, pooled into six land use classes and calculated in QGIS version 3.4.9 (Forest, Agra, Mining, Urban, Wetland, and Water). Raw data comes from ATKIS land use data (GeoBasis-DE/BKG 2013; AdV, 2006).</p>
<p>Area was calculated in hectares using QGIS (version 3.4.9).</p>
<p>Shoreline Development Factor (SDF) is the ratio of a lake's perimeter to the circumference of a circle of area equal to that of the given lake area. In which, SDF or $D_L = \frac{L}{2\sqrt{(\pi A)}}$</p>
<p>Conductivity and pH were both measured using a YSI multiprobe.</p>

Secchi Depth was measured using a standard Secchi disc and without the aid of a viewer box.
maxDepth the maximum depth in meters of each lakes, measured during bathymetry surveys on the same sample of lakes for the STÖRBAGGER project (unpublished dataset).
avgDepth and shallowness were calculated based on previously configured depth contour maps for each lake. “shallowness” refers to the percentage of the lake area with a depth of less than three meters.
Steepness refers to the Relative Depth Ratio (RDR), which is a ratio of the maximum depth and total lake area, calculated simply as $RDR = \frac{(maxDepth)(\sqrt{\pi})}{2\sqrt{A}}$

Table 6. Description of mean, median and range values across entire sample for all environmental variables used in PCA analysis. Per management type is also listed under managed and unmanaged mean.

Variable	Mean	Median	Range	Unmanaged Mean	Managed Mean
Forest	14.06	8.21	0-72.6	5.19	18.67
Agra	34.35	35.06	2.42-85.7	40.41	31.20
Mining	5.83	0.00	0-39.0	11.18	3.05
Urban	23.89	9.56	0-87.5	14.06	29.00
Wetland	2.75	0.00	0-45.1	3.93	2.13
Water	8.81	5.69	0.508-50.4	9.87	8.26
Woody	30.14	25.85	1.17-84.7	37.11	26.51
Open	19.56	15.97	0-86.7	11.54	23.73
Reed	18.83	14.41	0-72.3	15.29	20.67
Herbal	1.48	0.00	0-23.8	0.77	1.84
Artificial	0.50	0.00	0-7.22	0.10	0.71
Mixed	29.49	24.90	0-72.3	35.18	26.53
Area_ha	8.25	5.83	0.85-21.6	6.04	9.40
Perimeter_m	1390	1161	417-3002	1214	1482
Shoreline Development Factor	1.47	1.38	1.10-2.24	1.55	1.44
Relative Depth Ratio	0.036	0.037	0.005-0.071	0.04	0.03
Conductivity	0.46	0.39	0.136-1.54	0.42	0.48
Secchi Depth	2.25	1.80	0.3-5.6	1.95	2.40
pH	8.03	7.90	6.7-9.2	7.98	8.06
avgDepth	5.00	4.15	0.6-12.5	4.62	5.20
maxDepth	10.45	9.20	1.1-23.5	9.94	10.72
Percent Depth <3m	37.59	31.55	8.6-100	43.21	34.67

Amphibian community composition

Management types showed a significant difference in amphibian community composition in both lake sets, perhaps best explained by a higher density of *Bufo bufo* in managed lakes vs. a higher density of *Pelophylax sp.* in unmanaged lakes (Dim1, Figures 3 and 4). The second dimension of our RDA (Dim2, Figures 3 and 4) was divided, much like the first dimension, between two species: the Smooth newt (*Lissotriton vulgaris*) and the Common frog (*Rana temporaria*), however these two species were only found in a very small subset of our total lake sample (*R. temporaria* at 5 sites and *L. vulgaris* at only 2). Wetland coverage (i.e., PC score “Wetland”) was minorly significant in predicting community composition ($P < 0.1$), showing a positive relationship with *Lissotriton* (Figures 3 and 4), but this species was only recorded at two lakes. All other variables appeared to be insignificant in predicting amphibian community composition (i.e. Open_pH, Depth and Development). For our subset with fish data, RDA showed that the variables Wetland and Open_pH were both minorly significant in predicting amphibian community composition, however the influence of fish biomass ($p > 0.1$; Table 8) was not statistically significant. In the end, the only significant predictor was management as a binomial variable ($P < 0.05$; Table 8).

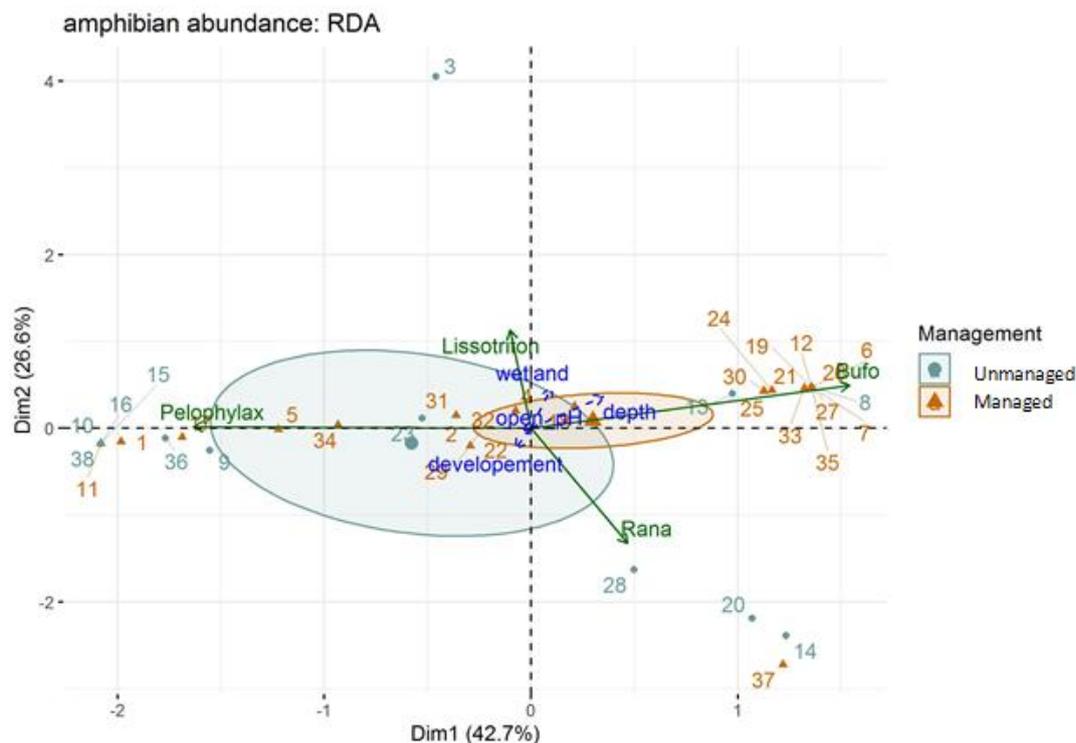


Figure 3. Redundancy Analysis (RDA) visualizing the variation in amphibian community composition (species = green) and the influence of environmental variables (blue) on this variation. The percentages on each axis refer to the proportional variance explained for that axis (i.e., dimension). Numbers represent the lake ID ($n = 38$) and management type (color coded) is considered as a supplementary variable. The ellipses around lake-type centers represent 95% confidence intervals. Differences among lake types are considered highly significant if there is no overlap.

Table 7. Output from the RDA analysis for the full set of lakes. variables (excluding “management”) are PC scores derived from lake-level and land use principal components (see Table 8). Each variable

was evaluated separately using permutation tests to select the most fitting variables for use in the RDA. As such, the Variance Explained is not in proportion to that of other variables, but rather a stand-alone measure. P-values < 0.05 are bolded and values < 0.1 are italicized.

Variable	Variance Exp.	P-value
management	0.344	0.015
wetland	0.241	<i>0.087</i>
depth	0.189	0.137
reed_wood	0.137	0.278
mining_water	0.127	0.321
steepness	0.104	0.429
development	0.094	0.491
forest	0.048	0.755
open_pH	0.011	0.970

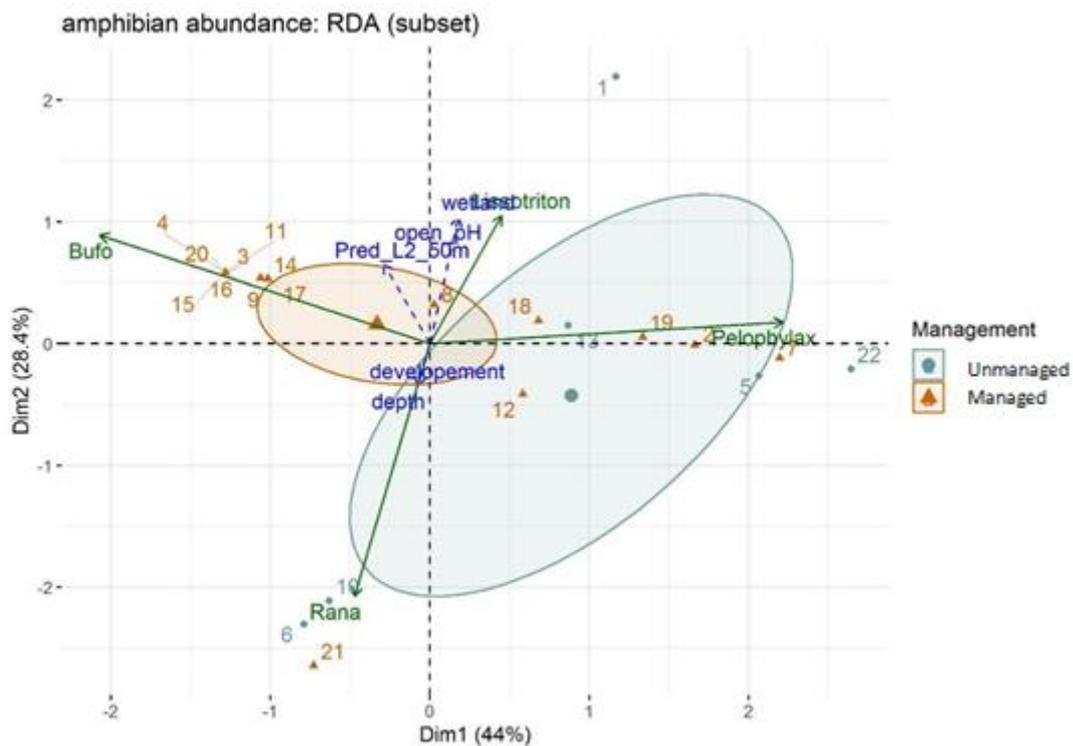


Figure 4. Redundancy Analysis (RDA) visualizing the variation in amphibian community composition (species = green) and the influence of environmental variables (blue) on this variation in the subset of lakes with fish data. The percentages on each axis refer to the proportional variance explained for that axis (i.e., dimension). Numbers represent the lake ID ($n = 22$) and management type (color coded) is considered as a supplementary variable. The ellipses around lake-type centers represent 95% confidence intervals. Differences among lake types are considered highly significant if there is no overlap.

Table 8. Output from the RDA analysis for the subset of lakes. variables, excluding fish biomass and management type are PC scores derived from lake-level and land use principal components (see Table 8). Each variable was evaluated separately using permutation tests to select the most fitting

variables for use in the RDA. As such, the variance explained (Variance Exp.) is not in proportion to that of other variables, but rather a stand-alone measure. P-values < 0.05 are bolded and values < 0.1 are italicized. “Pred_L2_50m” represents the piscivorous fish biomass metric calculated from electrofishing data (see Methods, Fish metrics).

Variable	Variance Exp.	P-value
management	0.553	0.023
wetland	0.440	<i>0.089</i>
open_pH	0.440	<i>0.085</i>
Pred_L2_50m	0.239	0.270
depth	0.092	0.766
development	0.092	0.757
steepness	0.066	0.814
forest	0.066	0.812
reed_wood	0.065	0.806
mining_water	0.065	0.808

Lake-level amphibian biodiversity

Shannon’s diversity index was not significantly different across lake types (Mann-Whitney U test: $w = 203.5$ $p = 0.172$) Species richness was significantly different and was higher in unmanaged lakes than in managed lakes (Mann-Whitney U test: $W = 250$, $P = 0.004$, Figure 5b).

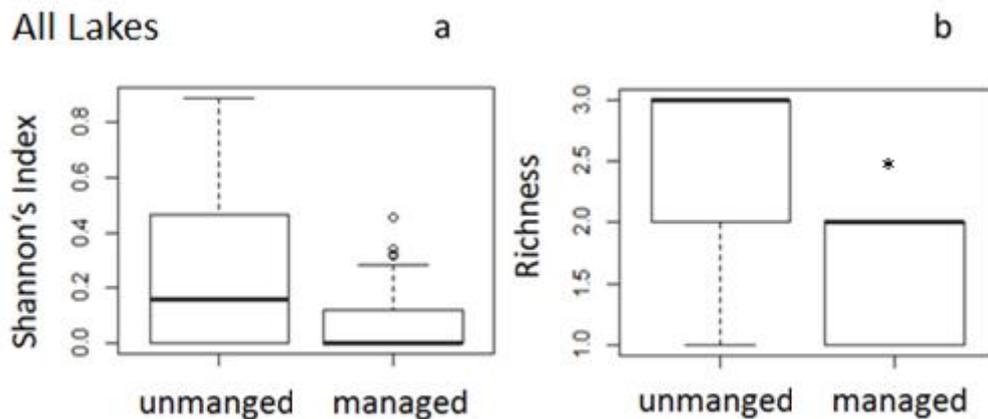


Figure 5. Box plots show biodiversity metrics (Shannon’s index and richness) over management type. T-bars with dotted lines show the maximum and minimum values, representing 25% of the dataset. Thick horizontal bands represent the median of biodiversity values. Significant test relationships ($P < 0.05$) are indicated with an asterisk.

Table 8. GLM output for all lakes and lake subset (see Methods, Variable and model selection for overview of full models)

All Lakes: <i>glm(Richness~Management)</i>				
Coefficients	Estimate	Std. Error	T-value	P-value
Intercept	2.38	0.179	13.3	< 0.001

Management	-0.785	0.221	-3.55	0.001
Null deviance	20.3 on 37 degrees of freedom			
Residual deviance	15.1 on 36 degrees of freedom			
AIC	78.7			
Lake Subset: <i>glm(Richness~Management)</i>				
Coefficients	Estimate	Std. Error	T-value	P-value
Intercept	2.50	0.249	10.0	< 0.001
Management	-0.937	0.292	-3.21	0.004
Null deviance	11.3 on 21 degrees of freedom			
Residual deviance	7.44 on 20 degrees of freedom			
AIC	44.6			

In both cases, management was the only significant predictor variable ($P < 0.005$, Table 8). However, when using Shannon's index (rather than Richness) as the dependent variable (and repeating this same selection process), we found that "Percent_0-3m" was the only significant predictor ($P < 0.05$). This relationship was also true for both lake sets. Since Shannon's index was partially explained by shallowness of the lake, we tested this variable (including management as a covariate) in a final GLM (see Table 9 and plot in Figure 6).

All Lakes

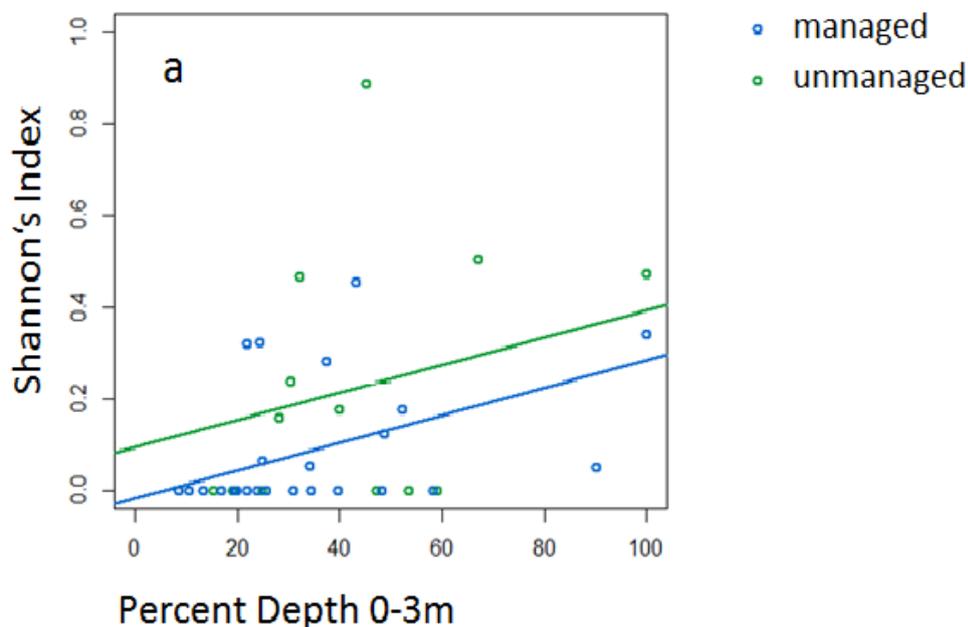


Figure 6. Raw values for Shannon's index graphed over raw values for percentage of depth < 3 m for managed and unmanaged lakes. Management was not a significant predictor (see Table 9).

Table 9. GLM output for final model (plotted above in Figure 6).

Final Model: <i>glm(Shannon~Management+Percent_0.3m)</i>				
Coefficients	Estimate	Std. Error	T-value	P-value
Intercept	0.093	0.080	1.17	0.251
Management	-0.110	0.066	-1.67	0.103
Percent_0.3m	0.003	0.001	2.17	0.037
Null deviance	1.57 on 37 degrees of freedom			
Residual deviance	1.25 on 35 degrees of freedom			
AIC	-13.9			

Shannon’s index was significantly higher in lakes with more shallow areas ($P < 0.05$), however the influence of management was not significant ($P > 0.1$).

Fish biomass was not a significant predictor of richness or Shannon’s index in our models.

Discussion

Our work identifies a total of 4 amphibian species found to inhabit gravel pit lakes, supporting the preliminary assessment from Nikolaus et al. (2020). Unlike Nikolaus et al. (2020), we found that species richness was significantly higher in unmanaged lakes, as opposed to managed lakes; unmanaged lakes hosted, on average, one more species than managed lakes. Likely our larger sample size was able to make this distinction clear ($n = 38$ vs $n = 22$ in Nikolaus et al. 2020), after all Nikolaus et al. (2020) do report a higher “conservation value” for amphibians in unmanaged lakes, than in managed lakes (which is of course related to species richness). Nonetheless, we could not explain a specific mechanism for this difference. Our results suggest that the percentage of shallow areas in lakes is positively correlated with amphibian abundance, however this relationship was only found in predicting Shannon’s index, and not richness. Furthermore, unmanaged lakes did not appear to have greater percentages of shallow areas than did managed lakes (Table 6), suggesting that while shallow areas may be important for amphibian biodiversity, it does not explain the difference in richness between management types. Fish biomass was also not a strong predictor of amphibian biodiversity; though fish represent a significant predation pressure on amphibian communities, fish biomass was present at similar abundances across all lakes in our subset; managed by recreational fisheries or otherwise. Likewise, the abundance of piscivorous fish species, while higher amongst managed lakes (Matern et al. 2019), did not predict higher or lower abundance of amphibians in our models.

The question remains, which environmental factor causing the difference in amphibian biodiversity observed across lake types? It is possible that this difference is explained by some unmeasured variable. For instance, recreational use was not quantified in this study (neither in type, nor intensity), a variable that likely plays a large role in patterns of biodiversity in gravel pit lakes (Schafft and Arlinghaus 2018). Another possibility could be that in our measurement of many environmental variables, there exists error or low level of detail which has resulted in unreliable results. One example regarding habitat variables: The shoreline structure “Mixed” represents a large fraction of total shoreline habitat structure

(see Table 6), however this “variable” is itself subdivided into different forms of habitat (be it Wood, Open, Reed, etc) and perhaps measuring shoreline habitat at a finer level could have provided more insight into this question. The following sections serve to discuss certain details regarding our methods and results and provide an idea bank for further research into this topic.

Environmental variation

Generally, lake types appeared to differ more at the landscape scale, than at the lake-level scale. This might suggest that fisheries management may have less of an influence on amphibian biodiversity, as the management itself is mostly concerned with lake-level characteristics (e.g., stocking, shoreline development, etc.). However, it is important to note that we only used a 100m buffer to investigate our landscape variables, meaning that most of this buffer is likely well within the influence (i.e., property boundaries) of recreational fisheries management. Furthermore, had we used a larger buffer (for example, 500m to 1km) we might have found more variation in land use variables. Notably wetlands were only recorded within 6 lake buffers, which makes its potential influence on management type unclear. Similar studies have used wider buffer zones: Shulse et al. used 300m buffers (2010), Holtmann et al. used 500m (2017), while others have used a variation of widths in their analyses (Hartel et al. 2010; Houlihan and Findlay 2003). Likely extending our buffer would have increased the variation in our PCA analysis and perhaps this would have helped predict amphibian biodiversity more accurately. Within our lake-level variables, we do not include information on in-water vegetation, such as macrophyte coverage and diversity; habitat factors which are likely important for amphibians (Holtmann et al. 2017; Boissinot et al. 2019). This data was left out due to time constraints, however Nikolaus et al. report a higher abundance and diversity of macrophytes at managed lake sites vs. unmanaged (2020).

Amphibian community composition

The variation in community composition was, not surprisingly, defined by the two most common species in our study: The Common toad (more dominant in managed lakes) and the Green ranids (more dominant in unmanaged lakes). Holtmann et al. (2017) also report a high abundance of these two species among shallow urban ponds, however they report a high abundance of *Rana temporaria* and *Lissotriton vulgaris* (presence at 70% and 40% of stormwater ponds respectively; Holtmann et al. 2017). The rare abundance of the Smooth newt in our sample was not surprising. Both adult and larval newts are known to be sensitive to fish predation (Denoel et al. 2005; Ficetola and Bernardi 2004). Smooth newts are among the more common species found in ponds, likely due to their relatively short residence time during the breeding season: *L. vulgaris* does not overwinter in ponds, as *T. alpestris* and *L. helveticus* sometimes do (Denoel et al. 2005), however we expect that the rarity of this otherwise common newt was due to abundant fish populations in both of our lake types. The Common frog (*Rana temporaria*) was also relatively scarce in our dataset compared to the occurrence of *Pelophylax* individuals. We suspect that the disproportionate occurrence of these species with our green ranids is more likely due to a non-fish variable, as these species appear to be equally threatened by fish predation (Altig and McDiarmid 2000).

Potentially, Common frogs were lacking at our sites due to a low percentage of forest cover in our lake buffers. Even though a few lakes in our study were predominantly surrounded by forest, most of our sites consisted of urban and agricultural landscapes. *Rana temporaria* is characterized as a common forest dwelling frog (AmphibiaWeb 2020), unlike the green ranids (known in German as “water frogs”). A similar forest inhabiting ranid, *Rana dalmatina* (the Agile frog) was found by Ficetola and Bernardi to be at greater abundance with increases in forest cover in lake buffer zones, while *Pelophylax* abundance was inversely related to forest cover (2009). It is possible that this same relationship exists within our sample, though we did not find that evidence to support this in our RDA (Tables 3 and 4).

The wide split between *Bufo bufo* and *Pelophylax sp.* between managed and unmanaged lakes (Figures 3 and 4) could be, in part, due to the defense mechanism of the Common toad’s larvae. Bufonid tadpoles are known to be unpalatable to most predators (including humans: Wassersug 1971), whereas ranid tadpoles (i.e., *Rana sp.*, *Pelophylax sp.*) are generally more palatable (Altig and McDiarmid 2000), and therefore more vulnerable to predators. Fishery-managed lakes are typically more populated by piscivorous species than unmanaged lakes (Matern et al. 2019), which have been reported to predate on adult amphibians (single observations by Hartel et al. 2007) and have been correlated with amphibian depletions (Hecnar and M’Closkey 1997; Hartel et al. 2007); therefore, selective predation may explain lower *Pelophylax* abundances in managed lakes. However, this is quite speculative, and our results (Table 8) show that predatory fish biomass did not significantly influence amphibian community composition in our RDA. Finally, although management types appeared to differ in terms of Urbanization and Agriculture (Figure 2, Table 5), these factors did not appear to influence the amphibian community composition in our RDA (see term “Development”, Figures 3 and 4).

Lake-level amphibian biodiversity

Our results showed inconsistent values for amphibian biodiversity: on the one hand, richness between lake types was statistically significant, even though richness was not explained by environmental variables. On the other hand, Shannon’s index was explained by lake shallowness (Percentage of depth <3m, Figure 6), but the effect of management type was insignificant. One explanation for this could be that our data for amphibian biodiversity, (i.e., Shannon’s index) was clearly zero-inflated (i.e., almost half of the data points are zeros). Shannon’s index assumes a value of zero for instances in which only one species is present (regardless of the absolute abundance), which was often the case in our study (Figure 6). It is possible that the fitted GLM (described in Table 8 and Table 9), which assumes a Gaussian distribution, does not accurately describe the dataset. In re-analyzing the data, it may be necessary to adjust our model and possibly run a Poisson or Gamma regression, whose assumptions more closely resemble our positively skewed data. In any case, the prediction that amphibian biodiversity may be positively correlated with shallowness would not be surprising. Amphibians thrive in shallow pools and, being ectothermic creatures, they rely on the relative warmth of shallow waters for breeding and larval development (Altig and McDiarmid 2000). Shulse et al. (2010) reported an increase in amphibian richness with shallowness and Holtmann et al. (2017) have shown that even urban stormwater pools

(being relatively fish-free) can support populations of threatened amphibians. Unexpectedly, fish biomass (total and piscivorous) was not a significant predictor for either richness or Shannon's index (Figure 7) in our lakes. However, this does not suggest that predation by fish was not occurring in our sample. It is possible, given our low richness overall, that the total fish biomass for individual lakes in our subset was high enough that even large differences in abundance between lakes would not have changed the overall predation pressure. In other words, even the lowest ratio of fish to amphibian biomass in our sample could already represent a maximum rate of predation. The predation pressure Matern et al. (2019) report a variety of biomass estimations (both electrofishing and gill net data) for the 22 lakes used in our subset, however the rate at which predation pressure increases with fish biomass, and what this curve might look like given our lakes' characteristics is unknown. Likely predation was occurring in all of our lakes, though our results suggest that this factor was not influenced by management type. The abundance of large-bodied carps was not included separately in our analysis, however the influence of these species, potentially through changes in macrophyte abundances (Bajer et al. 2016), on amphibian biodiversity in our sample is doubtful. These fish, while commonly stocked by recreational fisheries (Matern et al. 2019) were likely not present at such a high concentration as to harm the macrophyte community, and in fact Nikolaus et al. (2020) found that managed lakes hosted a higher abundance of macrophyte species than unmanaged lakes. Finally, we did not find a significant relationship between reed habitat coverage and amphibian biodiversity. Many of the individual encounters in the field were found within reed habitat (personal observation), however this habitat variable did not differ between management types (Figure 1) and was otherwise relatively abundant at most lakes (mean relative abundance >18% and median >14%, Table 6).

Final note on study design

The estimates and predictions we have put forth are not intended to represent larger holistic patterns of amphibian biodiversity. Marsh and Trenham warn that "pond-based studies will generally lead to pond-based explanations" about amphibian biodiversity (2001), and that one should take care to know the boundaries of one's research. To reiterate, our results do not offer much insight into the status of amphibian biodiversity in Lower Saxony, rather we provide a firsthand, non-biased account of the abundance of these species within gravel pit lakes in the region.

Conclusions

Recreational fisheries management (i.e., stocking practices) in Germany is often criticized by conservation entities for having a negative influence on amphibian biodiversity and while this may very well be the case under certain circumstances (i.e., Miró et al. 2018; Denoel et al. 2005), generalizations tend to emerge that could poorly advise management decisions. Our results suggest that amphibian biodiversity may be lower in gravel pit lakes managed by recreational fisheries (i.e. richness was found to be significantly lower). Nonetheless we observed a general scarcity of amphibian biodiversity across our whole sample, suggesting that these water bodies may be unsuitable for most amphibian species, regardless of the management regime. Our results suggest that shallowness is an important factor concerning amphibian biodiversity and management may consider expanding shallow areas to boost already present amphibian populations, though we would not expect this to improve upon the generally poor richness in these water bodies. In the interest of conserving amphibian biodiversity in Lower Saxony, conservation management should target those less-common species facing local or widescale depletions (i.e., *Bufo calamita*, *Rana arvalis*, and especially *Bombina variegata*; Podloucky and Fischer 2013). Therefore, while gravel pit lakes may provide adequate replacement breeding habitat for more common species (i.e., *Bufo bufo* or *Pelophylax* sp.), we would suggest focusing conservation efforts on the establishment of fish-free pools and the preservation of existing wetlands.

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References

- AmphibiaWeb., 2019. University of California, Berkeley, CA, USA. <http://amphibiaweb.org>, Accessed 18 Sep. 2019.
- 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.). Retrieved from <http://www.adv-online.de/AAA-Modell/Dokumente-der-GeoInfoDok/GeoInfoDok-6.0/>
- Altig, R., McDiarmid, R. W. (2000). Tadpoles - The Biology of Anuran Larvae. University of Chicago Press.
- Arlinghaus, R., Cooke, S.J., Potts, W., 2013. Towards resilient recreational fisheries on a global scale through improved understanding of fish and fisher behaviour. *Fisheries Management and Ecology*. <https://doi.org/10.1111/fme.12027>
- Arlinghaus, R., S.J., C., 2009. Recreational fisheries: socio-economic importance, conservation issues and management challenges.
- 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*. <https://doi.org/10.1111/gcb.13377>
- Berger, L., 1967. Embryonal and larval development of F1 generation of green frogs different combinations. *Acta Zoologica Cracoviensia*. 12, 123-160.
- 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*. <https://doi.org/10.1016/j.cosust.2016.11.012>
- Boissinot, A., Besnard, A., Lourdis, O., 2019. Amphibian diversity in farmlands: Combined influences of breeding-site and landscape attributes in western France. *Agriculture, Ecosystems and Environment*. <https://doi.org/10.1016/j.agee.2018.09.016>
- Daedlow, K., Beard, T.D. jr., Arlinghaus, R., 2011. A property rights-based view on management of inland recreational fisheries: contrasting common and public fishing rights regimes in Germany and the United States, in: *American Fisheries Society Symposium* 75.
- Denoel, M., Dzukic, G., Kalezic, M.L., 2005. Effects of widespread fish introductions on paedomorphic newts in Europe. *Conservation Biology*. <https://doi.org/10.1111/j.1523-1739.2005.00001.x>
- Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J., Sullivan, C.A., 2006. *Freshwater biodiversity: importance, threats, status and conservation challenges*.

- Biological reviews of the Cambridge Philosophical Society 81, 163–82.
<https://doi.org/10.1017/S1464793105006950>
- Ficetola, G.F., De Bernardi, F., 2004. Amphibians in a human-dominated landscape: The community structure is related to habitat features and isolation. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2003.11.004>
- Ficetola, G.F., Padoa-Schioppa, E., De Bernardi, F., 2009. Influence of landscape elements in riparian buffers on the conservation of semiaquatic amphibians. *Conservation Biology*. <https://doi.org/10.1111/j.1523-1739.2008.01081.x>
- Hartel, T., Nemes, S., Cogălniceanu, D., Öllerer, K., Schweiger, O., Moga, C.I., Demeter, L., 2007. The effect of fish and aquatic habitat complexity on amphibians. *Hydrobiologia*. <https://doi.org/10.1007/s10750-006-0490-8>
- Hartel, T., Schweiger, O., Öllerer, K., Cogălniceanu, D., Arntzen, J.W., 2010. Amphibian distribution in a traditionally managed rural landscape of Eastern Europe: Probing the effect of landscape composition. *Biological Conservation*. <https://doi.org/10.1016/j.biocon.2010.02.006>
- Hecnar, S.J., M'Closkey, R.T., 1998. Species richness patterns of amphibians in southwestern Ontario ponds. *Journal of Biogeography*. <https://doi.org/10.1046/j.1365-2699.1998.2540763.x>
- Hecnar, S.J., M'Closkey, R.T., 1997. The effects of predatory fish on amphibian species richness and distribution. *Biological Conservation*. [https://doi.org/10.1016/S0006-3207\(96\)00113-9](https://doi.org/10.1016/S0006-3207(96)00113-9)
- Holtmann, L., Philipp, K., Becke, C., Fartmann, T., 2017. Effects of habitat and landscape quality on amphibian assemblages of urban stormwater ponds. *Urban Ecosystems*. <https://doi.org/10.1007/s11252-017-0677-y>
- Houlahan, J.E., Findlay, C.S., 2003. The effects of adjacent land use on wetland amphibian species richness and community composition. *Canadian Journal of Fisheries and Aquatic Sciences* 26, 79–96. <https://doi.org/10.1139/F03-095>
- ifishman.de, 2019. *Baggersee Kick-Off-Meeting vom 03.11.2016*. <https://www.ifishman.de/news/artikel/200-baggersee-kick-off-meeting-vom-03112016>. Accessed 3 Dec. 2019
- IUCN, 2020. The IUCN Red List of Threatened Species. Version 2019-3. <https://www.iucnredlist.org>. Accessed 24.01.2020
- Kloskowski, J., 2009. Size-structured effects of common carp on reproduction of pond-breeding amphibians. *Hydrobiologia*. <https://doi.org/10.1007/s10750-009-9912-8>
- Mardia, K. V., Kent, J. T., Bibby, J. M., 1979. *Multivariate Analysis*. London: Academic Press.
- Marsh, D.M., Trenham, P.C., 2001. *Metapopulation dynamics and amphibian conservation*.

Conservation Biology. <https://doi.org/10.1046/j.1523-1739.2001.00129.x>

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*.
<https://doi.org/10.1111/jfb.13989>

Miró, A., Sabás, I., Ventura, M., 2018. Large negative effect of non-native trout and minnows on Pyrenean lake amphibians. *Biological Conservation*.
<https://doi.org/10.1016/j.biocon.2017.12.030>

Morris, E.K., Caruso, T., Buscot, F., Fischer, M., Hancock, C., Maier, T.S., Meiners, T., Müller, C., Obermaier, E., Prati, D., Socher, S.A., Sonnemann, I., Wäschke, N., Wubet, T., Wurst, S., Rillig, M.C., 2014. Choosing and using diversity indices: Insights for ecological applications from the German Biodiversity Exploratories. *Ecology and Evolution*.
<https://doi.org/10.1002/ece3.1155>

Nikolaus, R., Schafft, M., Andreas, M., Christian, W., Arlinghaus, R., 2020. Status of aquatic and riparian biodiversity in artificial lake ecosystems with and without management for recreational fisheries: Implications for conservation. not "peer reviewed."

Podloucky, R., Fischer, C., 2013. Rote Listen und Gesamtartenlisten der Amphibien und Reptilien in Niedersachsen und Bremen. Informationsdienst Naturschutz Niedersachsen.

QGIS Development Team., 2019. QGIS Geographic Information System. Open Source Geospatial Foundation.

R Core Team., 2019. R: A language and environment for statistical computing. Vienna, Austria.

Sawatzky, M.E., Martin, A.E., Fahrig, L., 2019. Landscape context is more important than wetland buffers for farmland amphibians. *Agriculture, Ecosystems and Environment*.

Schafft, M., Arlinghaus, R., 2018. StörBagger - ein neues Forschungsprojekt zur Wirkung von Anglern auf Natur und Umwelt. <https://www.ifishman.de/en/publications/single-view/1190-stoerbagger-ein-neues-forschungsprojekt-zur-wirkung-von-anglern-auf-natur-und-umwelt/>

Shannon, C., 1948. A mathematical theory of communication. *Bell System Technical Journal*.
<https://doi.org/10.1002/j.1538-7305.1948.tb01338.x>

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*. <https://doi.org/10.1007/s13157-010-0069-z>

Stuart, S.N., Chanson, J.S., Cox, N.A., Young, B.E., Rodrigues, A.S.L., Fischman, D.L., Waller, R.W., 2004. Status and trends of amphibian declines and extinctions worldwide.

Science.

Thiesmeier, B., 2014. Amphibien bestimmen – am Land und im Wasser. - Bielefeld (Laurenti).

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

Wassersug, R., 1971. On the comparative palatability of some dry-season tadpoles from Costa Rica. American Midland Naturalist. <https://doi.org/10.2307/2423690>

Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York. <https://doi.org/10.1007/978-0-387-98141-3>

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. <https://doi.org/10.1111/eff.12222>