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Elemental composition and microbial decomposition of different angling baits

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Abstract

Recreational fishing, through groundbaiting, provides a potentially important trophic subsidy to freshwater ecosystems that could promote eutrophication. To date, our understanding of the role of bait properties on their fate when they are not consumed remains limited. The present study aimed to determine if the C:N:P composition of some of the most commonly used angling baits modulated microbial decomposition. The C:N:P composition of 28 models of four commercially available angling baits was assessed and a microcosm experiment was used to quantify microbial decomposition. Elemental composition strongly varied among angling baits, with N content varying by a factor of three and P content varying by a factor of five. Microbial decomposition of angling baits was highly variable and faster for more industrialised baits. Microbial decomposition was five times faster for angling baits rather than natural subsidies (leaves) and was not significantly correlated with high N or P content. Compared to natural subsidies, angling baits were a favourable substrate for bacterial growth. Angling baits represent a source of highly available nutrients and the most-enriched angling baits might affect ecological processes and water quality in ecosystems with high angling pressure, especially when their use is not regulated.

KEYWORDS

artificial subsidies, fisheries, groundbaiting, natural subsidies, recreational fishing, stoichiometry

1 | INTRODUCTION

Fluxes of energy and materials between terrestrial and aquatic ecosystems are ubiquitous and allochthonous subsidies have important implications on community structure and the functioning of recipient ecosystems (Polis et al., 1997). Human activities generate a significant, yet underappreciated, source of subsidies to ecosystems. Specifically, humans provide important food subsidies to animal species through agriculture, livestock, hunting, fishing and commercial trade (Oro et al., 2013). Human-generated subsidies particularly affect aquatic ecosystems by modifying nutrient cycling that alters community structure and interactions among species (Oro et al., 2013). Recreational fishing, through baiting (i.e., the introduction of organic matter to enhance fish harvest), generates potentially important artificial subsidies to freshwater ecosystems (Arlinghaus & Mehner, 2003; Cryer & Edwards, 1987). Recreational fishing is a widespread activity in industrialised countries, practiced by more than 10% of the global population (Arlinghaus, 2004; Cooke & Cowx, 2004). Many angling techniques require the use of baiting that can represent, on average, 7.3 kg of bait per angler per year and can reach, in some cases, more than 200 kg (Arlinghaus, 2004; Arlinghaus & Mehner, 2003). Angling baits are primarily used in European waters as they are associated with the capture of Cyprinids (Linfield, 1980; Wedekind et al., 2001) and, more rarely, in other areas such where common carp (*Cyprinus carpio* L.) are voluntary introduced into recreational fisheries (FAO, 2018). In addition to the impacts on fish growth (Mehner et al., 2019), the use of nutrientrich angling baits can affect water quality (Lewin et al., 2006; Wolos et al., 1992) by promoting eutrophication (Amaral et al., 2013). The most commonly used angling baits can be categorised into four main types varying along a gradient of processing: seeds (mainly cereals), groundbaits (mixed of plant and animal flours), boilies (boiled animal or vegetable flours, compacted into balls of about 20-30 mm diameter) and pellets (mixture of compacted animal flours and oils, often fish oil, compacted to obtain a dense and solid bait). All types usually contain different additives to increase their attractiveness. These four types of baits could strongly differ in nutrient digestibility and carbohydrate, crude fibre and phosphorus contents (Arlinghaus & Niesar, 2005), which could modulate their potential ecological impact. Understanding the link between the elemental composition of angling baits and their potential impacts on recipient ecosystems is therefore needed.

Once introduced in freshwater ecosystems, angling baits can integrate food webs through direct consumption by targeted fish (and untargeted invertebrates) and microbial decomposition. When directly consumed by fish, angling baits can represent more than 80% of their diet when fishing pressure is high (Bašić et al., 2015). P content modulates the digestibility and nutrient retention in fish, with low digestibility positively correlated with angling baits P content (Arlinghaus & Niesar, 2005). This linkage between digestibility and phosphorus concentration is extremely important because fish excretion of consumed baits can affect nutrient bioavailability balance and create hotspots of nutrient cycling depending on spatial fish distribution (McIntyre et al., 2008). When inputs are high or fish consumption is limited, some angling baits can enter a microbial decomposition. This may affect water quality through deoxygenation and an increase in ammonium and hydrogen sulphide production (Koel et al., 2019; Wolos et al., 1992). The temporal dynamic of organic matter decomposition is enhanced by high N and P content and low C:N ratio (Zhang et al., 2008), and the elemental composition of angling baits likely modulates their dynamic in ecosystems.

Our study aimed to quantify how elemental composition modulates microbial decomposition of some commonly used angling bait types. Our first objective was to determine if the elemental composition (C:N:P) of the main types of commercially available angling baits strongly differed due to differences in ingredient composition. Our second objective was to determine if the microbial decomposition of different angling bait types increased with increasing N and P concentrations (i.e., lower N:P and C:N ratios).

2 | MATERIALS AND METHODS

2.1 | Elemental composition of angling baits

A total of 28 models were analysed among the most popular, commercially available, angling bait types used in France belonging to the four aforementioned types (seeds, groundbaits, boilies and pellets, see Figure S1 in Supplementary materials). They were purchased eries Management

in one popular shop and online from a popular angling website. Six models of seeds, nine models of groundbaits, eight models of boilies and five models of pellets were analysed. Each model was triplicated (84 samples in total). A sample with a mass between 5 and 7 g was oven-dried at 60°C for 72 h before grinding (Retsch MM200) into a homogeneous powder. Samples were analysed for C and N contents using a CHN analyser (Flash 200, Thermo Fisher Scientific, Waltham, MA, USA; method ISO 10694:13878) and P content was determined following the molybdate-blue method using a spectrophotometer (Uvi Light XT5, Secomam, Alès, France; absorbance measured at 880 nm) after acid digestion in the autoclave. Bait C, N and P contents were compared with natural subsidies represented by poplar leaves (*Populus nigra*) (9 samples collected in 2016 in 3 gravel pit lakes in Haute-Garonne, France) (Zhao et al., 2016).

2.2 | Microbial decomposition

2.2.1 | Experimental design

Microcosm experiments were used to quantify microbial decomposition by monitoring dry mass loss of bait and poplar leaves during 25 days of exposure in lake water. Three models per bait type were selected to represent the range of observed elemental composition (n = 12 treatments). In addition, poplar leaves were used as natural subsidies (Alp et al., 2016). Each microcosm was a 710-ml cylindrical vessel (117-mm diameter and 107-mm height) made of transparent PET, closed with a lid and filled with 500 ml of lake water. Following previous work using persistence of angling baits in situ (Koel et al., 2019), exposure durations were 2, 4, 8, 16 and 25 days. Every bait model and leaves were triplicated for each exposure time, for a total of 39 microcosms per exposure time and an overall total of 195 microcosms. Two microcosms filled with lake water only were added to be sampled on day-25 for bacterial analyses. All microcosms corresponding to the same exposure time were randomly placed on one of the five shelves of the same shelving unit. The experiment was conducted in the dark and at 18°C room temperature.

2.2.2 | Mass loss monitoring

Microcosms were inoculated with 500 ml of water from a nearby gravel pit lake (Four de Louge, N43°26′24″, E1°18′9″) (Zhao et al., 2016). Then, a known fresh mass of bait or leaves ($5.7 \pm 0.6 \text{ g}$ for baits and $1.0 \pm 0.3 \text{ g}$ for leaves) was added to each microcosm. Baits were introduced as used by anglers, so seeds were boiled and groundbaits were mixed with water to form compact balls before introduction to microcosms. To quantify dry mass introduced at the beginning of each experiment, the dry-mass ratio of each type of bait and leaves was computed from the mass before and after ovendrying at 60°C for 48 h. The dry mass ratio was then applied to each type of bait as the mean of 10 mass-loss measures (n = 130 measures for 13 baits models and leaves).

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After each exposure time, water from each microcosm was filtered through a coffee filter to collect the remaining material. The material was then oven-dried at 60°C for 48 h and weighed to the nearest 0.001 g to quantify the dry mass loss. Mass loss dynamics were first monitored through leaching as the physical dissolution of bait or leaves from water mechanical action. Leaching was calculated as a proportion between initial dry mass and dry mass after 2 days of exposure time. Mass loss then measured from microbial decomposition as a daily microbial decomposition rate (k) (Alp et al., 2016):

$$k = -\ln(X)/d$$

where X is the proportion of remaining mass calculated as the ratio of final (25 days) to initial (2 days) oven-dried mass and d the exposure time (23 days).

2.2.3 | Bacterial density

At the end of the experiment, and before water filtration, 1 ml of water in each remaining microcosm was collected to quantify bacterial density. Samples were fixed with formaldehyde (4% final concentration) and stored at 4°C before analysis. Baits and leaves microcosms corresponding to day 25 (n = 39) and microcosm with lake water only (n = 2) were analysed, with three measures in each microcosm with lake water only (n = 45 samples analysed in total). Total bacteria were counted using a Guava flow cytometer (Luminex) equipped with a blue laser providing 50 mW at 488 nm. Prior to flow cytometer analysis, samples were pre-filtered on 25 µm, diluted in 0.22 um filtered TE buffer (0.1 mM Tris-HCL and 1 mM EDTA, pH 8) and incubated with SYBR Green I (Thermo Fisher Scientific) at a final 10^{-4} dilution of the commercial stock solution, for 10 min at 75°C. Analysis was thus made on stained samples in which a suspension of 1-µm beads was added (Polysciences). Flow cytometer files obtained were analysed using the Guava EasyCyte 3.3 software with the InCyte assay, combining the green fluorescence (log) with the side scatter parameter (log).

2.3 | Statistical analyses

Generalised linear mixed-effects models were used to test differences in C, N and P proportions between bait types with bait model as a random effect and a Beta distribution as family. Stoichiometric ratios were log-transformed (Isles, 2020) and differences between bait types were tested using linear mixed-effects models with each bait model as a random effect. The validity of Imer model was tested by checking the normality of the residual distribution. Coefficients of variation were calculated (CV) as the ratio of the standard deviation to the mean, to quantify the range of C:N:P content and stoichiometric ratios within each angling bait type. Generalised linear mixedeffects models were used to compare leaching rate and microbial decomposition rate among bait types with bait model as a random factor and Gamma distributions as families. The Poisson family was used for testing differences in bacterial density among bait types, with the bait model as a random effect. Finally, correlations between microbial decomposition rates, the bacterial density of each angling bait, and the C:N and N:P ratios were tested using spearman correlations. All statistical analyses were performed using R v.4.0.3 (R Development Core Team, 2020). Generalised linear mixed-effects models were performed using the glmmTMB function in the package 'glmmTMB' for elemental composition and the glmer function in the package 'Ime4' for mass loss monitoring. Linear mixed-effects models were performed using the lmer function from package 'Ime4'. The significance of each variable was estimated using Anova from the 'car' package. Post-hoc pairwise comparisons among baits were performed with the emmeans function from 'emmeans' package.

3 | RESULTS

3.1 | Elemental composition of angling baits

C, N and P contents were highly variable among and within angling bait types (Figure 1, Table S1 in Supplementary materials). The average C content of angling baits ranged from 33.5% to 60.5% (Figures 1 and 2a) and significantly differed among bait types (glmer, $X^2 = 17.51$, p < 0.001; Figure 2a). Groundbaits C content (40.4 \pm 3.5%) was significantly lower than seeds (47.4 \pm 7.0%, post-hoc test, p = 0.002) and pellets (46.7 \pm 2.2%, post-hoc test, p = 0.012). Groundbaits and boilies did not differ significantly in C content (44.5 \pm 2.7%, post-hoc test, p = 0.092). Variability in C content was greater among models of seeds (CV = 15%) than groundbaits (CV = 9%), boilies (CV = 6%) and pellets (CV = 5%) (Figure 2a). Leaves C content (43.3 \pm 1.4%) was within the range of angling bait types.

Average N content of angling baits ranged from 0.6% to 7.3% (Figures 1 and 2b), and differed significantly among bait types (glmer, $X^2 = 19.879$, p < 0.001; Figure 2b). Pellets N content (5.5 \pm 1.5%) was significantly higher than all baits models: groundbaits (1.8 \pm 0.3%, post-hoc test, p < 0.001), seeds (2.5 \pm 1.9%, post-hoc test, p = 0.002) and boilies (2.9 \pm 1.6%, post-hoc test, p = 0.045). Variability of N content was greater among models of seeds (CV =78%) than boilies (CV =55%), pellets (CV =28%) and groundbaits (CV = 17%). Leaves N content (1.0 \pm 0.4%) was similar to the value of seeds and lower than the other bait types (Figure 2b).

Average P content ranged from 0.1% to 1.8% (Figures 1 and 2c) and differed significantly among bait types (glmer, $X^2 = 60.76$, p < 0.001). Seeds were lowest (0.25 \pm 0.14%) and pellets were highest (1.24 \pm 0.46%) in P content. P content differed significantly among all bait types, except between groundbaits (0.36 \pm 0.15%) and seeds (0.25 \pm 0.14%, post-hoc test, p = 0.1756). Boilies P content (0.57 \pm 0.17%) was lower than pellets but higher than seeds and groundbaits. Variability of P content was greater in seeds (CV = 57%) than boilies (CV = 31%). Variation within pellets (CV = 36%) was lower than within seeds and groundbaits (CV = 41%), but one model of pellets strongly differed from the others. Leaves P content

FIGURE 1 C, N and P contents (%) of four angling bait types: seeds (n = 6), groundbaits (n = 9), boilies (n = 8) and pellets (n = 5). Each bait type was replicated 3 times. Poplar leaves originate from 3 gravel pit lakes (n = 9)



• Seeds • Groundbaits • Boilies • Pellets • Leaves

 $(0.07 \pm 0.02\%)$ was lower than groundbaits, boilies and pellets, but was comparable to seeds (Figure 2c).

The C:N ratio ranged from 6.3 to 88.4 and differ significantly among bait types (Imer, X² =14.062, p = 0.002), with pellets (9.7 ± 4.4) significantly lower than seeds (34.7 ± 26.8, post-hoc test, p = 0.012) and groundbaits (23.1 ± 2.9, post-hoc test, p = 0.016) (Figure 2d). Boilies C:N ratio was 19.0 ± 7.2. Other pairwise differences were not significant. Variability in the C:N ratio was greater in seeds (CV = 77%), with one significantly higher than others. Groundbaits were less variable (CV = 13%) than boilies (CV = 38%) and pellets (CV = 46%). Leaves C:N ratio (48.6 ± 15.6) was similar to seeds but 2 to 4 times higher than other bait types (Figure 2d).

The N:P ratio ranged from 1.5 to 42.0 among all baits, and from 4.8 (\pm 1.1) for pellets to 11.6 (\pm 10.9) for seeds (Figure 2e), but did not differ significantly among bait types (lmer, $X^2 = 6.1843$, p = 0.103). The N:P ratio of groundbaits and boilies were 5.5 (\pm 1.8) and 5.3 (\pm 2.3), respectively. Heterogeneity of seeds was high (CV = 94%), with some models having a ratio ten times higher than others. Other bait types were more homogeneous, with a lower CV for pellets and boilies (both CV = 23%), and intermediate CV for groundbaits (CV = 32%). Leaves N:P ratio was 14.9 (\pm 4.4) and higher than all bait types.

3.2 | Microbial decomposition of angling baits

At the end of the experiment (day 25), remaining dry mass averaged only 0.2 for groundbaits, but ranged between 0.6 and 0.9 for seeds and poplar leaves (Figure 3). The proportion of dry mass loss by leaching ranged from 0 to 52.5% and significantly differed among bait types (glmer, $X^2 = 337.10$, p < 0.001; Figures 3 and 4a). Leaching of groundbaits was significantly higher than for other bait types (49.8 \pm 2.0%, post-hoc test, p < 0.001): seeds $(10.2 \pm 7.0\%)$, boilies $(13.7 \pm 4.8\%)$ and pellets $(21.2 \pm 1.9\%)$. Dry mass leached in leaf treatments was 9.2 ± 1.0%. Microbial decomposition rate of baits ranged from 0.0001 to 0.06 day⁻¹. Microbial decomposition rate differed significantly among bait types (glmer, $X^2 = 12.248$, p = 0.006, Figure 4b). Specifically, the decomposition rate of seeds (0.012 \pm 0.009 day⁻¹) was significantly lower than for groundbaits (0.043 \pm 0.014 day⁻¹, post hoc-test, p = 0.003), boilies (0.029 \pm 0.009 day⁻¹, post hoc-test, p = 0.034) and pellets $(0.030 \pm 0.006 \text{ day}^{-1}, \text{ post-hoc test}, p = 0.030)$. Leaves microbial decomposition was 0.006 \pm 0.001 day⁻¹. Stoichiometric nutrient ratios were not significantly correlated to microbial decomposition rates of angling baits (Spearman correlations, $\rho = -0.10$, p = 0.75 and $\rho = -0.21$, p = 0.51, for C:N and N:P ratios, respectively).

Bacterial density ranged from 0.4×10^7 to 2.9×10^7 cells ml⁻¹. Bacterial density did not differ significantly among angling bait types (glmer, $X^2 = 2.114$, p = 0.549; Figure 4c), but was three times higher in microcosms with baits instead of leaves ((6.0 ± 4.9).10⁶ cells ml⁻¹), and ten times higher instead of no substrate ((1.3 ± 0.4) 10⁶ cells ml⁻¹). Stoichiometric ratios were not significantly correlated to bacterial density (Spearman correlations, $\rho = -0.27$, p = 0.39 and $\rho = 0.25$, p = 0.43, for C:N and N:P ratios, respectively).



FIGURE 2 Elemental composition of each type of angling bait, expressed in (a) C content, (b) N content, (c) P content, (d) C: N and (e) N: P ratios. Dotted lines represent poplar leaves. Big coloured dots are mean and standard deviation. Small size points represent individual data points. The grey area represents data distribution



FIGURE 3 Temporal dynamic of mass loss (proportion of remaining dry mass) of each type of angling baits (n = 3 models replicated three times): seeds, groundbaits, boilies and pellets. Poplar leaves are displayed with grey points



FIGURE 4 The proportion of dry mass loss by (a) leaching, (b) microbial decomposition rate and (c) bacterial concentration after 25 days of exposure time in lake water for each bait type. The dotted lines are poplar leaves. The dashed line in (c) represents the bacterial density in microcosms with lake water only. Different letters indicate significant differences between types. Big coloured points correspond to mean values and standard deviations. Small size points are individual data

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4 | DISCUSSION

Angling baits could represent an important artificial subsidy for freshwater ecosystems where angling activity is high and potential effects of angling bait on ecosystems are likely modulated by variability in bait elemental composition. The P composition of the large panel of baits analysed in this study was in the same order of magnitude as the values measured on 4 bait models used to feed juvenile carps (Niesar et al., 2004), suggesting that baits used in this study were qualitatively similar to those found elsewhere. The absence of correlation between microbial decomposition and elemental composition suggests that other factors such as differences in physical characteristics of baits may explain why seeds were less easily decomposable than groundbaits and why groundbaits were more easily leached than seeds. In addition, the potential presence of artificial preservatives in angling baits might inhibit microbial decomposition in the most enriched baits, and thus mitigate the causal relationship between elemental composition and decomposition kinetics (Rapp et al., 2008).

Angling bait composition and microbial decomposition were compared to natural subsidies using poplar leaves, a very ubiquitous subsidy in many ecosystems (Alp et al., 2016). Importantly, the microbial decomposition rate of poplar leaves measured in the present study was comparable to values measured directly in lakes (Alp et al., 2016), which suggests that the experimental conditions of our study were realistic. Bait microbial decomposition rates revealed higher decomposability and lower persistence of angling baits than natural subsidies. Except for seeds, angling baits microbial decomposition rates were more than 5 times faster than poplar leaves, and exceeded decomposition rates of dead fish biomass in a boreal lake (<0.02 days⁻¹; Chidami & Amyot, 2008), fish carcasses in some eutrophic European lakes (<0.048 days⁻¹; Premke et al., 2010) and aquatic plants (<0.01 days⁻¹; Reddy & DeBusk, 1991). Compared with tree leaves, the predominant form of natural subsidies in freshwater ecosystems (Bartels et al., 2012), angling baits represent a potential source of highly available nutrients for recipient ecosystems, with P composition 3-10 times higher, and with an N:P ratio of pellets more than 2.5 lower than leaves. Then, understanding how it enters and potentially affects freshwater food webs is crucial.

Angling baits not directly consumed by fish could modulate nutrient bioavailability balance as P-rich inputs in freshwater ecosystems where productivity is often limited by phosphorus (Hecky & Kilham, 1988). The large difference observed between leaves and angling baits bacterial density, 3 times higher than in microcosms with leaves and 10 times higher than in microcosms without substrate (only lake water), suggests that angling subsidies represent an allochthonous substrate that might favour bacterial growth. Such P-rich subsidies could favour the growth of denitrifying bacteria (Zhang et al., 2018) and promote cyanobacterial development (Isles et al., 2017). Therefore, the ecosystem impacts of angling baits on eutrophication will largely depend on the type of bait that anglers use. For example, a specialised carp angler can use up to 200 kg of WILEY- Fisheries Manage and Ecology

angling baits per year (Arlinghaus & Mehner, 2003), which would represent a potential yearly input ranging from 0.5 to 2.5 kg of phosphorus, depending on whether pellets or seeds are used. The elemental composition of poplar leaves measured in the present study was within the range of N and P contents observed for 32 other tree species (0.8% to 2.61% and 0.07% to 0.37%, respectively) in northern Spain (Sardans et al., 2011). Assuming that riparian vegetation loses 500 g of leaves per meter of the riparian zone (Gasith & Hosier, 1976; Staehr et al., 2010), this artificial input of phosphorus would be similar to the annual subsidies provided by a 100–500 m of the riparian zone. Artificial subsidies of angling baits can occur all year, unlike more seasonal subsidies from leaf litter and invertebrates (Nakano & Murakami, 2001), although seasonal variability of angling technique and target species could similarly affect the temporal dynamics of food webs (McMeans et al., 2015).

Angling bait-derived nutrients could be counterbalanced by the removal of nutrients stored in caught fish. A carp angler would need to remove annually more than 200 kg of fish for counterbalancing a mean annual phosphorus input of 1 kg (Niesar et al., 2004). However, this compensation is unlikely to occur because, in most recreational fisheries where baits are used, catch-and-release angling is extremely common, especially in carp fishing (Arlinghaus & Mehner, 2003). Thus, the angling bait matter is not removed from the recipient ecosystem, irrespective of whether they are consumed or not by fish. Another effect of angling baits subsidies on the ecosystem enrichment is the increase in fish growth rate and thus biomass due to the energetic advantages of angling baits (Mehner et al., 2019). Effects of nutrient subsidies on nutrient bioavailability and primary production also depend on the way they are released, with differences between continuous, pulsed and disrupted subsidies (Weber & Brown, 2013). In addition to the way baits are introduced, the characteristics of recipient ecosystems are important for assessing their potential impacts. Although this remains to be quantified, angling baits could induce larger impacts on water quality in small and eutrophic lakes than in larger and/or oligotrophic lakes.

Fishery management recommendations that regulate the number of baits should therefore consider the type of baits and fishing pressure and practices and explicitly account for the ecological status of the ecosystem (e.g., productivity and size). This might include limited use of most-enriched and faster decomposable baits and the limitation of quantity uses by anglers in periods with high productivity levels. Nevertheless, the consequences of these inputs on ecosystem functioning, which will largely depend on the type of baits and the seasonality of baiting, remain to be quantified in ecosystems with variable levels of eutrophication.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest for this article.

DATA AVAILABILITY STATEMENT

Data available on request from the authors.

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REFERENCES

- Alp, M., Cucherousset, J., Buoro, M. & Lecerf, A. (2016) Phenological response of a key ecosystem function to biological invasion. *Ecology Letters*, 19(5), 519–527. https://doi.org/10.1111/ele.12585
- Amaral, S.D., Brito, D., Ferreira, M.T., Neves, R. & Franco, A. (2013) Modeling water quality in reservoirs used for angling competition: Can groundbait contribute to eutrophication? *Lake and Reservoir Management*, 29(4), 257–269. https://doi.org/10.1080/10402381.2013.845804
- Arlinghaus, R. (2004). 'Recreational fisheries in Germany a social and economic analysis', Berichte des IGB, Heft 18. Berlin: Leibniz-Institut für Gewässerökologie und Binnenfischerei (IGB) im Forschungsverbund Berlin e.V., p. 166.
- Arlinghaus, R. & Mehner, T. (2003) Socio-economic characterisation of specialised common carp (*Cyprinus carpio* L.) anglers in Germany, and implications for inland fisheries management and eutrophication control. *Fisheries Research*, 61(1-3), 19–33. https://doi. org/10.1016/S0165-7836(02)00243-6
- Arlinghaus, R. & Niesar, M. (2005) Nutrient digestibility of angling baits for carp, *Cyprinus carpio*, with implications for groundbait formulation and eutrophication control. *Fisheries Management and Ecology*, 12(2), 91–97. https://doi.org/10.1111/j.1365-2400.2004.00425.x
- Bartels, P., Cucherousset, J., Steger, K., Eklöv, P., Tranvik, L.J., Hillebrand, H., & Bartels, P. (2012) Reciprocal subsidies between freshwater and terrestrial ecosystems structure consumer resource dynamics. *Ecology*, 93(5), 1173–1182. https://doi.org/10.1890/11-1210.1
- Bašić, T., Britton, J.R., Jackson, M.C., Reading, P. & Grey, J. (2015) Angling baits and invasive crayfish as important trophic subsidies for a large cyprinid fish. *Aquatic Sciences*, 77(1), 153–160. https://doi. org/10.1007/s00027-014-0370-7
- Chidami, S. & Amyot, M. (2008) Fish decomposition in boreal lakes and biogeochemical implications. *Limnology and Oceanography*, *53*(5), 1988–1996. https://doi.org/10.4319/lo.2008.53.5.1988
- Cooke, S.J. & Cowx, I.G. (2004) The role of recreational fishing in global fish crises. *BioScience*, 54(9), 857.
- Cryer, M. & Edwards, R.W. (1987) The impact of angler groundbait on benthic invertebrates and sediment respiration in a shallow eutrophic reservoir. *Environmental Pollution*, 46(2), 137–150. https://doi. org/10.1016/0269-7491(87)90199-0
- FAO. (2018) 'Review of the state of the world fishery resources : inland fisheries', p. 397.
- Gasith, A. & Hosier, A.D. (1976) Airborne litterfall as a source of organic matter in lakes. *Limnology and Oceanography*, 21(2), 253–258. https://doi.org/10.4319/lo.1976.21.2.0253
- Hecky, R.E. & Kilham, P. (1988) Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment1: Nutrient enrichment. *Limnology* and Oceanography, 33(4part2), 796–822. https://doi.org/10.4319/ lo.1988.33.4part2.0796
- Isles, P.D.F. (2020) The misuse of ratios in ecological stoichiometry. Ecology, 101(11), e03153. https://doi.org/10.1002/ecy.3153
- Isles, P.D.F., Xu, Y., Stockwell, J.D. & Schroth, A.W. (2017) Climate-driven changes in energy and mass inputs systematically alter nutrient concentration and stoichiometry in deep and shallow regions of Lake Champlain. *Biogeochemistry*, 133(2), 201–217. https://doi. org/10.1007/s10533-017-0327-8

- Koel, T.M., Thomas, N.A., Guy, C.S., Doepke, P.D., MacDonald, D.J., Poole, A.S. et al. (2019) Organic pellet decomposition induces mortality of lake trout embryos in yellowstone lake. *Transactions of the American Fisheries Society*, 149(1), 57–70. https://doi.org/10.1002/ tafs.10208
- Lewin, W.-C., Arlinghaus, R. & Mehner, T. (2006) Documented and potential biological impacts of recreational fishing: Insights for management and conservation. *Reviews in Fisheries Science*, 14(4), 305– 367. https://doi.org/10.1080/10641260600886455
- Linfield, R.S.J. (1980) Catchability and stock density of common Carp, *Cyprinus carpio* L. in a Lake Fishery. *Aquaculture Research*, 11(1), 11– 22. https://doi.org/10.1111/j.1365-2109.1980.tb00277.x
- McIntyre, P.B., Flecker, A.S., Vanni, M.J., Hood, J.M., Taylor, B.W. & Thomas, S.A. (2008) Fish distributions and nutrient cycling in streams: can fish create biogeochemical hotspots ? *Ecology*, 89(8), 2335-2346. https://doi.org/10.1890/07-1552.1
- McMeans, B.C., McCann, K.S., Humphries, M., Rooney, N. & Fisk, A.T. (2015) Food web structure in temporally-forced ecosystems. *Trends* in Ecology & Evolution, 30(11), 662–672. https://doi.org/10.1016/j. tree.2015.09.001
- Mehner, T., Rapp, T., Monk, C.T., Beck, M.E., Trudeau, A., Kiljunen, M. et al. (2019) Feeding aquatic ecosystems: Whole-lake experimental addition of angler's ground bait strongly affects omnivorous fish despite low contribution to lake carbon budget. *Ecosystems*, 22(2), 346–362. https://doi.org/10.1007/s10021-018-0273-x
- Nakano, S. & Murakami, M. (2001) Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. *Proceedings of the National Academy of Sciences*, 98(1), 166–170. https://doi.org/10.1073/pnas.98.1.166
- 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
- Oro, D., Genovart, M., Tavecchia, G., Fowler, M.S. & Martínez-Abraín, A. (2013) 'Ecological and evolutionary implications of food subsidies from humans', *Ecology Letters*. 16(12), 1501–1514. https://doi. org/10.1111/ele.12187
- Polis, G.A., Anderson, W.B. & Holt, R.D. (1997) Toward an integration of landscape and food web ecology: the dynamics of spatially subsidized food webs. *Annual Review of Ecology and Systematics*, 28(1), 289–316. https://doi.org/10.1146/annurev.ecolsys.28.1.289
- Premke, K., Fischer, P., Hempel, M. & Rothhaupt, K.-O. (2010) Ecological studies on the decomposition rate of fish carcasses by benthic organisms in the littoral zone of Lake Constance, Germany. Annales de Limnologie - International Journal of Limnology, 46(3), 157–168. https://doi.org/10.1051/limn/2010017
- R Development Core Team. (2020). R: a language and environment for statistical computing. Vienna, Austria. Available at: http://www.rproject.org/
- Rapp, T., Meinelt, T., Krüger, A. & Arlinghaus, R. (2008) Acute toxicity of preservative chemicals in organic baits used in carp, Cyprinus

carpio, recreational fishing. *Fisheries Management and Ecology*, 15(2), 163–166. https://doi.org/10.1111/j.1365-2400.2008.00598.x

- Reddy, K.R. & DeBusk, W.F. (1991) Decomposition of water hyacinth detritus in eutrophic lake water. *Hydrobiologia*, 211(2), 101–109. https://doi.org/10.1007/BF00037366
- Sardans, J., Rivas-Ubach, A. & Peñuelas, J. (2011) Factors affecting nutrient concentration and stoichiometry of forest trees in Catalonia (NE Spain). Forest Ecology and Management, 262(11), 2024–2034. https://doi.org/10.1016/j.foreco.2011.08.019
- Staehr, P.A., Sand-Jensen, K., Raun, A.L., Nilsson, B. & Kidmose, J. (2010) Drivers of metabolism and net heterotrophy in contrasting lakes. *Limnology and Oceanography*, 55(2), 817-830. https://doi. org/10.4319/lo.2010.55.2.0817
- Weber, M.J. & Brown, M.L. (2013) Continuous, pulsed and disrupted nutrient subsidy effects on ecosystem productivity, stability, and energy flow. *Ecosphere*, 4(2), art27. https://doi.org/10.1890/ ES12-00354.1
- Wedekind, H., Hilge, V. & Steffens, W. (2001) Present status, and social and economic significance of inland fisheries in Germany, *Fisheries Management and Ecology*, 8(4-5), 405–414.
- Wolos, A., Teodorowicz, M. & Grabowska, K. (1992) Effect of groundbaiting on anglers'catches and nutrient budget of water bodies as exemplified by Polish lakes. Aquaculture Research, 23(4), 499–509. https://doi.org/10.1111/j.1365-2109.1992.tb00793.x
- Zhang, D., Hui, D., Luo, Y. & Zhou, G. (2008) Rates of litter decomposition in terrestrial ecosystems: global patterns and controlling factors. *Journal of Plant Ecology*, 1(2), 85–93. https://doi.org/10.1093/jpe/ rtn002
- Zhang, Y., Song, C., Ji, L., Liu, Y., Xiao, J., Cao, X., & Zhou, Y. (2018) Cause and effect of N/P ratio decline with eutrophication aggravation in shallow lakes. *Science of the Total Environment*, *6*27, 1294–1302. https://doi.org/10.1016/j.scitotenv.2018.01.327
- 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

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