

Peatland fish of Sebangau, Borneo: diversity, monitoring and conservation

S.A. Thornton^{1,2}, Dudin^{2,3}, S.E. Page^{1,2}, C. Upton¹ and M.E. Harrison^{1,2}

¹School of Geography, Geology and the Environment, University of Leicester, UK

²Borneo Nature Foundation, Palangka Raya, Central Kalimantan, Indonesia

³UPT LLG CIMTROP, University of Palangka Raya, Central Kalimantan, Indonesia

SUMMARY

Tropical peat swamp forests provide important ecosystem services, ranging from carbon storage and fire prevention to fish provision. In the Sebangau catchment of Central Kalimantan, Indonesia, we completed the first detailed spatial and temporal assessments of local fish biodiversity in peat swamp forest and blackwater river habitats. Monthly environmental and fish data were collected over a 15-month period in both riverine and forest habitats. This resulted in a species list of 55 species from 16 different families. Species richness in the river was almost 1.5 times higher than in the forest, probably due to the sampling methods and trap selectivity. Average monthly river fish catches were negatively correlated with average monthly river depth. River fish surveys were conducted pre- and post- fire in 2015, with results showing increased river acidity and reduced fish catches post-fire. The fish and environmental data presented form a baseline for future monitoring projects and highlight a previously overlooked potential impact of fire on local biodiversity in Indonesia, namely that fire is likely to have negative impacts on the sizes of fish populations and catches. There are direct implications for human communities that depend on fishing for their livelihoods. Because peatlands and their rivers face continued human disturbance and degradation, assessments of fish biodiversity and water quality are of high priority.

KEY WORDS: blackwater fish, fire, Kalimantan, livelihoods, peat swamp forest

INTRODUCTION

Due to the characteristics of the blackwater aquatic habitats associated with tropical peat swamp forest (PSF), *i.e.* high acidity, high content of dissolved organic matter and low nutrient content, Indonesia's peatland rivers provide habitat for unique assemblages of fish species that often exhibit high endemism (Ng *et al.* 1994, Noor *et al.* 2005). In peatland areas such as those in lowland Central Kalimantan, fishing is usually one of the main sources of livelihood for communities living beside the blackwater rivers (Lyons 2003, Chokkalingam *et al.* 2007). However, with ever-increasing human populations, alarming deforestation rates *e.g.* in Indonesia, and ongoing aquatic pollution and habitat degradation, the prognosis for freshwater aquatic habitats throughout Asia is a matter of growing concern (Dudgeon 2000, Giam *et al.* 2012). There have been few studies of fish in SE Asia's peatlands, but the one by Giam *et al.* (2012) found that 77 % of fish species are likely to become extinct in Sundaland if deforestation of PSF continues, with Central Kalimantan being most heavily impacted. There are severe implications for these wetland ecosystems, as well as for the local communities that depend on PSF

and its associated aquatic habitats for their livelihoods. Therefore, it is vital to better understand these wetland habitats and their fish populations alongside their importance for community livelihoods and cultures, with the aim of finding ways to conserve and promote biodiversity, and specifically fish biodiversity, in conjunction with community development.

The results presented in this article arise from a broader interdisciplinary project that investigated values related to fishing and the importance of fish for local human communities in the Sebangau river catchment (Thornton 2017). Here we focus on fish species diversity, a comparison of fish populations in forest and riverine habitats, and how the species composition of PSF fish populations relates to habitat type and abiotic conditions. We aim to provide important baseline data to support future peatland fish monitoring in the Sebangau and elsewhere.

METHODS

Study site

The Sebangau Forest (Figure 1) is centred on the Sebangau River and is bordered by the Katingan River

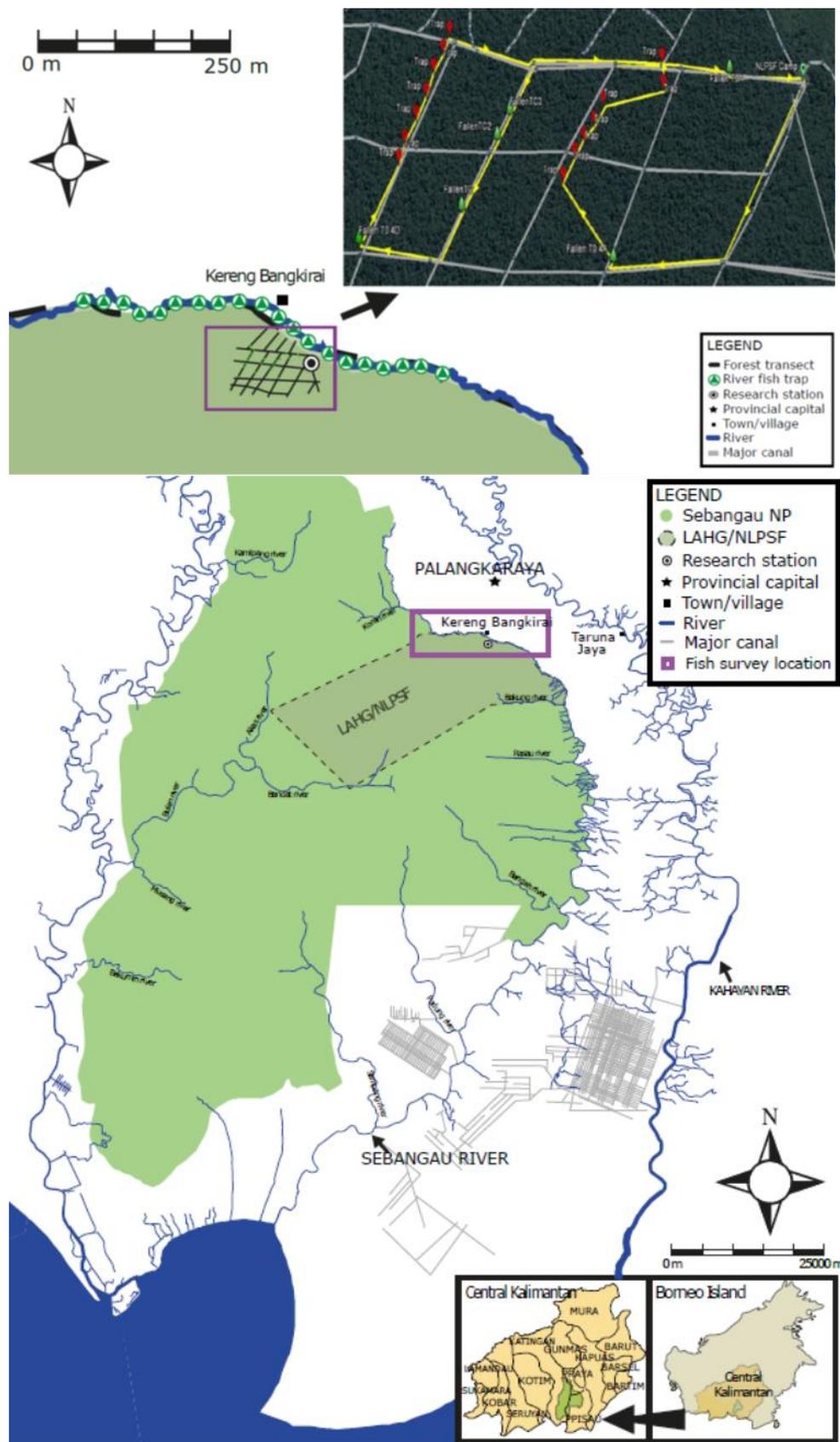


Figure 1. Below: map of the study area, showing Sebangau National Park and the Sebangau River. Above: detailed map of river and forest fish trap locations. Forest trap placements in canals are shown with red markers, and placements beside fallen trees with (green) tree markers. An example route is indicated in yellow. Maps from and edited in Google Earth, Image CNES/Astrium, 2016.

to the west and the Kahayan River to the east. The Sebangau River is a mid-sized blackwater river that arises from the swamp (in contrast to all the main rivers in the area, which rise in the hills) and runs through the Sebangau catchment for about 150 km to its mouth on the Java Sea coast (Tachibana *et al.* 2006) (Figure 1). Blackwater rivers like the Sebangau typically have low quantities of suspended matter, high amounts of humic acids (giving the water a brownish-reddish colour that can look black in certain light conditions) and a pH ranging from 4 to 5 (Ríos-Villamizar *et al.* 2014). The Sebangau Forest is characterised by a dome-shaped ombrogenous peatland with thick peat and low topographic elevation (Page *et al.* 1999). The forest experiences flooding for some months of the year during the wet season, with resulting standing water pools. There are also small canals in the forest which were previously dug and used during the logging years (approximately 1997–2004) to transport timber out of the forest. The Sebangau peat formation is the oldest known in SE Asia - approximately 26,000 years old (Page *et al.* 2004) - and has a maximum thickness of 13 m (Page *et al.* 1999, Weiss *et al.* 2002). The area of the Sebangau Forest selected for the fish community surveys was the Natural Laboratory of Peat Swamp Forest (NLPSF), which is managed by the Center for International Cooperation in Sustainable Management of Tropical Peatland (UPT LLG-CIMTROP) at the University of Palangkaraya. This study site was chosen due to the long history of PSF research arising from collaboration between the Borneo Nature Foundation (BNF) and UPT LLG-CIMTROP.

Fish sampling

Sampling in the Sebangau River was conducted from September 2014 to December 2015, and in the Sebangau Forest from February 2015 to July 2015. The sampling period was shorter in the forest due to insufficient above-surface water depth to set traps during the dry-season months (a minimum of 5 cm water depth was needed). Many types of fishing gear have been developed by fishing communities around the world, but relatively few have been adopted for the purposes of research and management (Portt *et al.* 2006). We employed local fish trapping methods (Dayak names in italics), using *tampirai* traps baited with a mixture of *tempeh* (fermented soya bean) and *terasi* (fermented shrimp paste). *Tampirai* traps are rectangular wire mesh traps with two (inner and outer) tapering mouths, which allow fish to enter the trap but not to escape (Figure 2). These traps are in regular use by local communities in the Sebangau area because of their reported effectiveness. The bait



Figure 2. Tampirai trap used in this study.

mixture used for the fish surveys is also commonly used by local fishermen and was recommended by them for use in this project. Following an initial trial of various locally available traps (bottle trap, bamboo trap and two different sizes of wire traps; Figure 3), a *tampirai* trap with mesh size 0.6 cm and dimensions 38 × 89 cm was chosen because it could trap a wider variety of fish species and fish sizes than other locally available trap types. This was also found by Worthington (2016), who compared the traps used in this study with other locally available traps not covered by the original trial survey. Worthington tested the larger *tampirai* used in this survey along with the *pangilar* (similar shape to *tampirai*, with dimensions 100 × 80 × 80 cm and mesh size 2 cm), the smaller rattan *pangilar* (28 × 33 × 33 cm, mesh size 2.5 cm) and a *buwu* (cylindrical trap 140 cm long, circumference 60 cm, with a double set of conical mouths) made of wire with mesh size 1 cm). The *tampirai* gear not only caught the most varied fish species assemblage, but also had the lowest species selectivity (where selectivity is the inverse of diversity). The *tampirai* traps (with mesh size 1 cm) are, however, selective against smaller fish and will not effectively sample fish guilds that include other smaller species. Therefore, this study focused only on the larger fish species that could be trapped using these methods.

When setting the traps, the bait was rolled into a small ball and put into a wire holder that was attached to the inside of the trap. On every sampling day the bait was (removed and) replaced, whether or not it had been eaten, in order to minimise negative impacts on captures resulting from bait predation and loss.

Twenty traps were set in the river and 20 in the forest (see the detailed map in Figure 1). Sampling took place every month, resulting in 13 months' data for the river and six months' data for the forest. Each sampling campaign involved five days of trapping



Figure 3. Fish trap types tested in our initial trial. Top left: a bottle trap similar to the one tested (image from Tarka Challenge 2012); bottom left: a bamboo *buwu* trap; and right: wire *tampirai* traps in two different sizes.

(traps were set, then retrieved the following day and set again following data collection) in each location *per* month. The sampling strategy for each location was chosen to balance sampling frequency and feasibility. For the river surveys, 20 traps were deployed on alternate sides of the river at 400 m intervals over a 7 km stretch (Figure 1). Trap 1 was located at latitude 2° 17' S, longitude 113° 52' E, and Trap 20 was downstream at 2° 18' S, 113° 56' E. The forest surveys involved sampling of two different open water habitats. Fourteen traps were set on the sides of two canals (maximum width 2.5 m; Figure 4), with the traps located 50 m apart from each other. Six traps were placed in tip-up pools (formed when shallow-rooted trees are uprooted; Dommain *et al.* 2015, Figure 4) spread across the study area. These traps were set towards the edges of the pools with the entrance facing the middle of the pool, as recommended by local fishermen.

The forest sampling locations were chosen to maximise the area over which sampling took place while keeping it practically possible to check all of the traps in one day. Due to the physical difficulty of walking in PSF, especially during the wet season, the trapping area covered in the forest was smaller than that covered along the river. Surveying the traps was facilitated by designing an efficient 4 km walking route through the forest using an existing permanent transect system (Figure 1). In the river, the traps were set at approximately mid-depth in the water column. We recognise that this depth will vary between locations, and future research could test whether trap depth (its vertical location in the water column) has an impact on the numbers and species of fish trapped. Our chosen approach followed advice from local fishermen on the placement most likely to trap the widest variety of species, and we avoided placing traps either at the very bottom of the water



Figure 4. Traps in the forest were set in previous logging canals (above) and tip-up pools (below).

column or at the top. The traps were aligned with the opening facing upstream to further discourage fish from escaping.

In both river and forest habitats, traps were set the day before the first sampling day. Throughout each monthly sampling campaign, each trap was then checked and emptied daily (usually between ~ 08.00 and 14.00 hours, although this could vary depending on the number of fish trapped). On emptying the trap, all fish were identified and their standard length (SL, measured from the most anterior extremity, mouth closed, to the base of the median tail fin rays) recorded to the nearest mm. If more than 100 fish

were caught in a trap, a sample of 20 individuals of each species was measured and the rest counted. Fish identification was performed visually using Kottelat & Whitten (1993), subsequently checked using online resources (e.g. FishBase; www.fishbase.org), and then verified following consultation with taxonomic experts. In some cases it was only possible to identify fish to genus level in the field, e.g. *Clarias* spp. (walking catfish). Care was taken to avoid stressing the fish by placing them in buckets of water after collection, keeping the buckets covered to reduce the risk of overheating, and returning the fish to the water immediately after measurement. Any

mortality (*i.e.* whether the fish was alive or dead when counted) was noted.

There was occasional opportunistic sampling of fish in that, when a fish species which had not been trapped was observed in the river or forest, it was collected with a hand net to add to the species list. Opportunistic sampling led to four species being added to the list, namely *Rasbora kalbarensis* and *Kottelatlimia pristis* along with two *Mystus* species.

Water sampling

Environmental measurements were taken either during each trapping site visit or once a month. Daily environmental measurements were water temperature, pH, dissolved oxygen (DO) and Secchi disk depth as a proxy for water turbidity (river only) (Table 1). Monthly measurements were water depth, width, flow rate (forest only) and nutrient levels (Table 1). On the final day of sampling of each month, surface water samples were collected from the sampling locations and on the same day were taken to a refrigerator and kept at 4 °C if storage prior to analysis was needed. Chemical analysis of the water samples was performed at the University of Palangka Raya laboratory using Atomic Absorption

Spectrophotometry (AAS; Spectra 30, Nordson, Duluth, USA) following standardised procedures. Analysis for P followed the ascorbic acid method of Eisenreich *et al.* (1975) after Murphy & Riley (1962) (see Sulistiyanto 2005 for further details), NO₂ analysis was carried out using the Griess test (a standard procedure for testing nitrite in water, see Sulistiyanto 2005), and NO₃ analysis followed Yang *et al.* (1998). Secchi disk depth data were collected only from February 2015 onwards, after the digital turbidity meter that was used previously failed.

There were some differences between river and forest habitats in the types of environmental data that could be gathered. Secchi disk depth data could not be obtained in the forest because the pools and canals were too shallow. Flow measurements were not taken in the river, as they were impossible to perform accurately from a boat, but surface water flow rate in canals in the forest was measured using a stopwatch and a floating ping-pong ball. A previous study by Tachibana *et al.* (2006) reported that the water discharge rate of the Sebangau River varied greatly between dry and wet seasons, ranging from a maximum of 50 m³ s⁻¹ during wet season months to a minimum of 5 m³ s⁻¹ in dry season months.

Table 1. Summary of the environmental variables, methods of measurement and frequency.

Water variable	Method	Frequency
Depth	Measuring tape with weight attached	Monthly
Width	Measuring tape or GPS	Monthly
Temperature	pH meter or the temperature function on the ProODO YSI Digital meter ¹	Daily
Flow rate	Ping-Pong ball and measuring stick ²	Monthly (Forest only)
pH	Stick meters (Hanna HI-98127 or equivalent) ³	Daily
Dissolved oxygen	ProODO YSI Digital meter ¹	Daily
Turbidity	Secchi disk ⁴	Daily
Nutrient content (P, NO ₂ , NO ₃)	Laboratory analysis; see text for details of methods.	Monthly

¹ This is a standard method, as used by *e.g.* Bodamer & Bridgeman (2014), Hedström *et al.* (2017), Geeraert *et al.* (2017).

² A known technique, see *e.g.* Petr (1970), Ikomi *et al.* (2005).

³ Standard method using portable pH meters, see *e.g.* Li & Li (2009), Aziz *et al.* (2012), Dodemaide *et al.* (2018).

⁴ A known technique, see *e.g.* Preisendorfer (1986), Sandén & Håkansson (1996).

Post-fire data collection

Indonesia was hit by extensive forest and peatland fires in 2015, when a strong El Niño-related drought combined with forest disturbance and widespread peatland drainage made 2015 the worst fire season since 1997 (Chisholm *et al.* 2016). In Kalimantan the 2015 fire season began in August (Field *et al.* 2016), with conditions worsening until the fires were extinguished at the onset of the wet season in November. Fish and water surveys were discontinued in October due to health and safety concerns, then restarted in November and December when the start of the wet season led to improved air quality and made fieldwork conditions safe once more. Owing to resource restrictions, the post-fire surveys were conducted over three days (in comparison to the usual five days of sampling before the fires). Data were collected on the *in-situ* environmental variables (*i.e.* DO, pH, water temperature, depth *etc.*) but no nutrient analyses were undertaken.

Data analysis

To standardise captures for data analysis and comparisons, fish catch *per unit effort* (CPUE) was calculated using the following formula (Merilä 2015):

$$CPUE = \frac{N_{catch}}{(N_{traps} \times N_{nights})} \quad [1]$$

where N_{catch} = number of fish trapped, N_{traps} = number of traps set, N_{nights} = number of nights for which traps were set. This allows the catch data to be compared between times when the number of trapping days varied (*e.g.* pre- and post-fire, and if a trap went missing).

Due to the selective nature of any trapping gear, and the heterogeneity of the river and the forest habitats, it is possible that the ‘catchability’ and/or species composition of the fish populations in the two habitats differs. Knowing whether this is the case is very challenging in practice. To understand the potential number of species that may have been missed by our surveys, estimated species richness was calculated. EstimateS computes non-parametric asymptotic species richness estimators: Chao-1 and ACE using abundance data, ICE using incidence data (presence data) and Chao-2 using replicated incidence data (as samples were replicated over several days) (Gotelli & Colwell 2010). As recommended by Colwell (2013), the classic instead of the bias-corrected option was used for these calculations, as Chao’s estimated coefficient of variation for Abundance distribution and CI for Incidence distribution was high (>0.5). Therefore,

the larger Chao-1 Classic and ACE are reported as the better estimates of abundance-based richness, and the larger Chao-2 and ICE as better estimates of incidence-based richness (see Colwell 2013 for detailed descriptions of these estimators and procedures).

Percentage fish mortality was calculated as the percentage of fish that were dead when the traps were emptied. To explore species turnover over time, Jaccard’s species similarity between the months was calculated for both habitats and between habitats.

As data collection in the forest ran for only six months (February–July 2015), correlation or regression analyses were not carried out between variables due to low sample size. Correlation analyses are presented for the river data. We acknowledge that the spatial and temporal sampling of the traps are not statistically independent of each other. Achieving completely independent samples was not practically feasible. Therefore, we recommend continued long-term data collection which will allow more in-depth and complex statistical analyses to be completed in the future.

RESULTS

Over the course of 1,300 river survey trap nights (20 traps × 5 days *per month* × 13 months), a total of 55,147 fish of 38 species were trapped and counted, with 22,917 fish measured. In the forest, a total of 3,938 fish of 27 species were trapped and counted over 600 trap nights (20 traps × 5 days *per month* × 6 months), with 3,905 fish measured. Four other species, namely: *Rasbora kalbarensis*, *Kottelatlimia pristes* and two *Mystus* species, were trapped opportunistically at the mouth of a canal by the river. From published literature, some other fish species are also known to be present in the Sebangau area: *Betta hendra*, *Silurichthys ligneolus*, *Hemirhamphodon tengah* (Page *et al.*, 1997, Ng & Tan 2011, Schindler & Linke 2013). *Hemirhamphodon chrysopunctatus* was seen in the river but not trapped. These species are examples of those missed by our trapping method due to their small body size and their feeding behaviours (*Betta* spp. are small, slow swimming air breathers while *Hemirhamphodon* spp. are surface feeders and thus not attracted to the bait in fishing traps). A further species, *Anabas testudineus*, was trapped during a pilot survey in the forest in September 2014, whilst *Wallago leeri* is reported to be present in the Sebangau River by local fishermen. These examples illustrate that other species are likely to be present in the area but were not collected by our sampling methods. With these additions, our final

species list comes to 29 species in the forest and 41 in the river, producing a total of 55 species from 16 families in the Sebangau study area (Table A1 in Appendix). Abundances of each species trapped for each month and habitat are presented in Table A2. The estimated species richness in forest and river indicated that the forest would have fewer species than the river (Figure 5), and this was supported by the survey data. On this basis we detected 75–94 % of the estimated species richness of the river (average of 47 species estimated), while the trapping in the forest captured more species than were estimated to be present across all estimators (29 species were recorded, with an average estimation of 26 species between ACE, ICE and Chao estimates). In both the river and the forest, ACE estimators were the most conservative, with the Chao-1 estimator always giving the highest estimated species richness. For the forest estimates there is good agreement between all richness estimators (minimum = ACE with 25.69, maximum = Chao-1 with 25.99) while the river estimators show higher variance (minimum = ACE with 43.57, maximum = Chao-1 with 54.99).

The river fish assemblage was dominated by *Osteochilus spilurus* (32 % of river catch), *Sphaerichthys acrostoma* (20 % of river catch), *Desmopuntius foerschi* (11 % of river catch), *Mystus olyroides* (9 % of river catch) and *Rasbora cephalotaenia* (8 % of river catch), while the forest was dominated by *Rasbora kalochroma* (52 % of forest catch), *Betta anabatooides* (13 % of forest catch), *Encheloclarias tapeinopterus* (8 % of forest catch), *Channa gachua* (5 % of forest catch) and *Belontia hasselti* (5 % of forest catch). The total number of species trapped in both the river and the forest was 17, which constitutes 45 % of the total species count for the river and 63 % for the forest. In the forest the remaining 37 % included species that are usually found in forest streams and standing water habitats (e.g. *Betta anabantoides*, *Rasbora kalochroma*, *Clarias meladerma*, *Anabas testudineus*).

With regard to species turnover over time, the species similarity in the river decreased over the transition between the dry and wet season (October–November 2014), and then had a general trend of increasing throughout the wet season (Figure 6A). The greatest turnover of species in the river occurred in November–December 2014 when there was a species similarity of 42 %. This corresponds with a decrease in species richness during these months (Figure 7) and, therefore, the increase in species turnover can be explained by fewer species being caught at the onset of the wet season. For the forest, there was a relatively high species similarity of 70 %

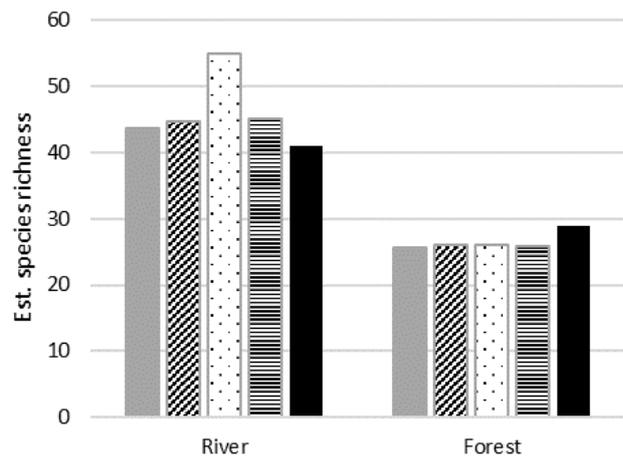


Figure 5. Estimated species richness in the Sebangau Forest and River using ACE [grey], ICE [diagonal line], Chao1 [dots] and Chao2 [horizontal line], compared to final species list numbers [black column].

in February–March 2015, with an increase in species similarity to 79 % in March–April (thereby lower species turnover). After March, there was a consistent decrease in species similarity (*i.e.* increasing turnover of species) until the end of the forest sampling (in July). The greatest species turnover in the forest, with a similarity of 53 %, was between June and July. This corresponds to a decrease in species richness from 15 species trapped in June to 8 species in July (Figure 7). The calculated Jaccard's similarity values between months for the river were not significantly correlated to any of the environmental variables (Table 2). Species richness showed no statistically significant correlation with any of the environmental variables. There was a positive correlation with river water temperature, although this is likely to be an artefact of the small sample size (Table A3). Correlation analyses were not run for the forest dataset due to the small sample size.

Comparing the species similarity values between the two habitats in February–July 2015 (Figure 6b), there was consistently low similarity, with similarity never increasing above the maximum similarity value of 21 % found in May 2015. The lowest similarity between the two habitats occurred in July with a value of only 7 %. Statistical analysis comparing species similarity to environmental trends was not done due to small sample sizes.

A consideration of changes in fish body size (SL in mm) over time demonstrated large variations in mean body size (M) between sample months (Figure 8). Overall, there was no significant difference in the average body sizes of fish trapped in the river (M = 63.1 mm, SD = 9.3) and forest

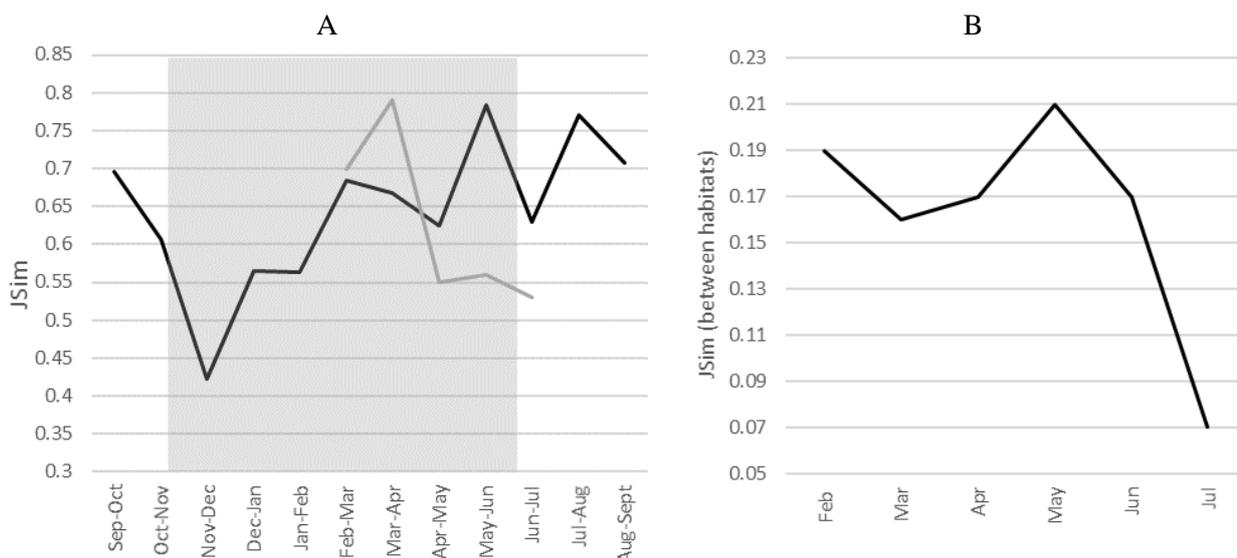


Figure 6. A: Species turnover in the river (black) and forest (grey) over time, calculated as Jaccard's similarity coefficient between each month. The grey box indicates the approximate duration of the wet season. B: Species similarity between the forest and river surveys from February to July 2015, calculated as Jaccard's similarity coefficient.

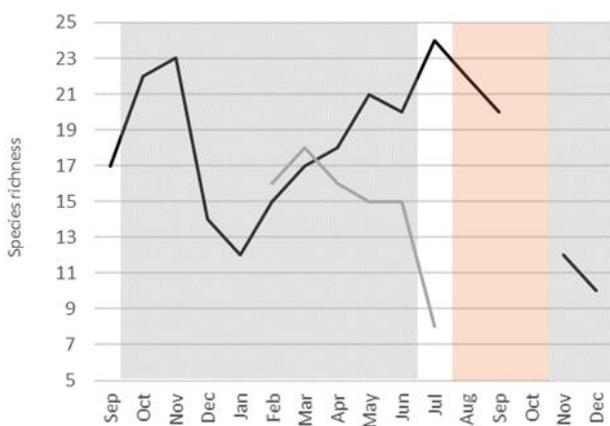


Figure 7. Species richness over time in the river (black) and forest (grey). The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

Table 2. Statistical analysis (Spearman rho, r_s) or Pearson's product correlation (PPC) if indicated, between Jaccard's species similarity (JS) in the river over time compared to the changes in environmental variables; $n = 12$, $p > 0.05$ in all cases.

Change in DO	-0.446 (PPC)
Change in pH	-0.112 (PPC)
Change in water depth	-0.544 (PPC)
Change in temperature	0.256 (PPC)
Change in rainfall	-0.455 (r_s)
Change in P	0.035 (r_s)
Change in NO ₂	-0.476 (r_s)
Change in NO ₃	-0.350 (r_s)

($M = 64.6$ mm, $SD = 5.4$); $t(15) = -0.47$, $p = 0.644$. At the change from the dry season to the wet season (October to November 2014), the average body size of fish trapped in the river increased from approximately 62 mm in October to 86 mm in November. It then decreased back to 61 mm in December with fluctuation around a generally decreasing trend until September 2015. In the forest there were similar fluctuations, with the greatest decrease in body size occurring with the onset of the dry season between June (average body size 72 mm) and July (average body size 56 mm).

The river had an average CPUE of 42.2 (from September 2014 to September 2015), compared to 6.1 in the forest (from February 2015 to July 2015; over the same time period, the river had an average CPUE of 45.93) (Mann-Whitney $U = 0.003$, $n = 19$, $p = 0.003$) (Figure 9 plots the CPUE on a \log_{10} scale). In the forest, the CPUE decreased between June and July with the onset of the dry season. In the river, there was a large increase in CPUE between June (CPUE = 2.03) and July (CPUE = 41.2) (Figures 9 and 10). The CPUE in the river was negatively correlated with average river depth ($r_s = -0.571$,

$n = 13$, $p = 0.041$) (Figure 10, Table A3). While there was no statistically significant correlation between monthly CPUE and turbidity (Table A3) this could be due to a small sample size (Secchi disk measurements were only collected from February onwards). Considering daily rather than monthly CPUE data (in order to increase the sample size) did result in a modest negative correlation between CPUE and average daily Secchi disk depth ($r_s = -0.479$, $p = 0.002$, $n = 38$), suggesting that fewer fish were trapped in the river with increasing water clarity. More long-term data collection is recommended to better explain the relationship between fish catch and water turbidity.

The river was deeper and wider than the water bodies in the forest, with an average water depth of 5.4 m in the river and 0.4 m in the forest (Table 3). The average water body width in the river was 30 m and in the forest 2.4 m. Both water depth and width varied a lot in both habitats, particularly for the river (Table 3). Over the time of the surveys, there was an increase in water depth in the river following the onset of the wet season in October–November 2014

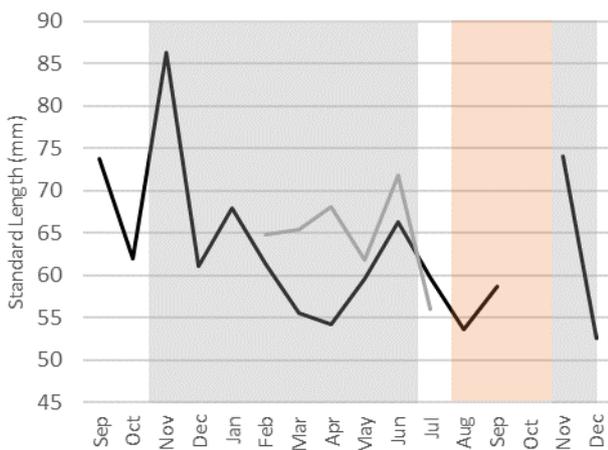


Figure 8. Variation in average body size over time, measured as Standard Length (mm) of fish trapped in the river (black) and forest (grey). The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

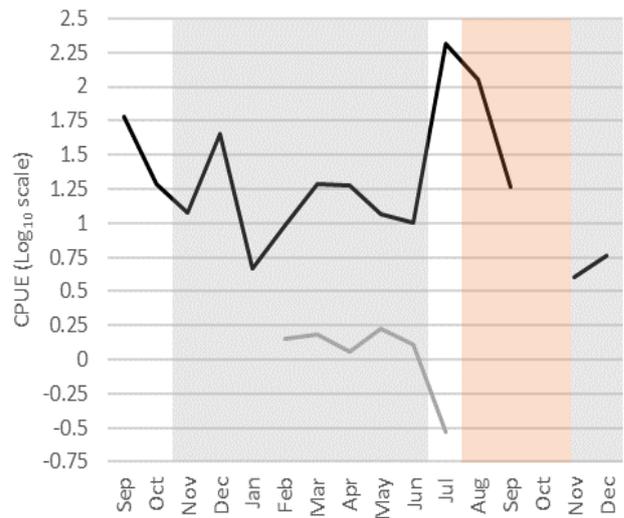


Figure 9. Variation in monthly Catch per Unit Effort (CPUE) over time, plotted on a \log_{10} scale, in the Sebangau River (black) and Forest (grey). The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

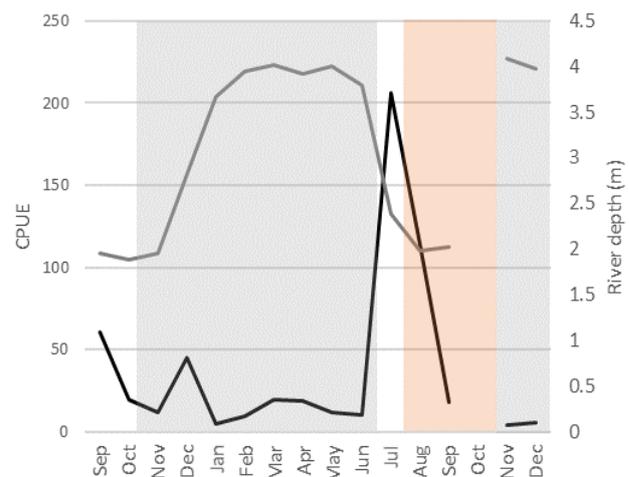


Figure 10. Variation in river CPUE (black) and water depth (m) (grey) over time. The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

Table 3. Average water depth (m) and water body (WB) width (m) in the river and the forest, with minimum (Min.) and maximum (Max.) measured. Standard deviation indicated in brackets.

Location	Average water depth	Min.	Max.	Average WB width	Min.	Max.
River	5.4 (± 1.48)	1.5	8.7	30.0 (± 18.73)	3.3	130.0
Forest	0.4 (± 0.17)	0.1	0.9	2.4 (± 1.68)	0.3	12.5

(Figure 11, water depth graph). Water depth then remained stable between February and June before decreasing from June through until September 2015 with the onset of the dry season. Water depth in the forest decreased constantly from February through to

July 2015. In the river the average monthly water depth was positively correlated with average DO levels ($r_s = 0.718$; Table A3).

There were greater fluctuations in DO levels in the river compared to the forest (Figure 11). Forest

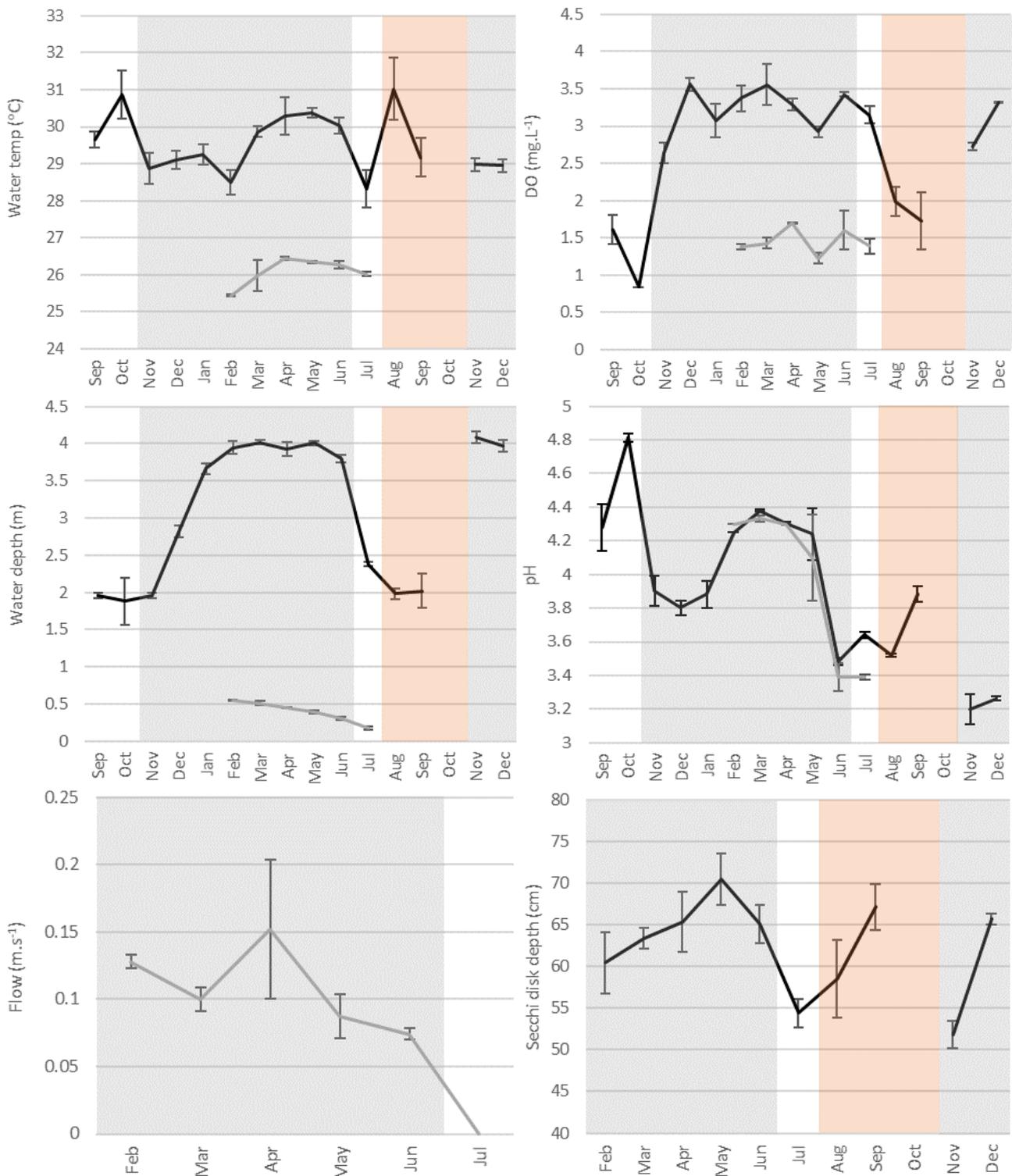


Figure 11. Environmental variables in the river (black) and forest (grey). Error bars indicate standard deviation where appropriate. The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

values fluctuated between monthly averages of 1.2 and 1.7 mg L⁻¹ (February to July), whilst river values fluctuated between 2.9 and 3.6 mg L⁻¹ over the same months. Across all months, the river had the lowest average DO value in October 2014 (0.8 mg L⁻¹) and the highest average values between December 2014 and June 2015 (maximum of 3.6 mg L⁻¹ in December). These higher values correspond with the wet season, which ran from January to April/May, with a positive correlation between amount of rainfall (and hence higher river flow rates, Tachibana *et al.* 2003) and river DO ($r_s=0.718$, $n=13$, $p=0.006$). There was a negative correlation between average monthly river water temperature and DO levels (PPC $r=-0.589$, $n=13$, $p=0.034$). Also, while the forest exhibited lower water temperatures in general compared to the river, there was not a corresponding higher DO level in the forest, with DO levels

consistently lower in the forest compared to the river between February and July 2015 (Figure 11).

Following the fires in 2015, there was an almost five-fold increase in acidity of the water in the Sebangau River, pH dropping from a pre-fire mean of 3.88 ($n=100$) in September to 3.20 ($n=60$) in November at the end of the fire season. The latter measurement was also the lowest pH value obtained during the whole river sampling period (Figure 11). This drop in pH corresponded with a decline in fish CPUE in the river from 18.21 in September to 4.02 in November.

Nutrient levels also showed temporal variations. At the onset of the wet season there was a spike in P levels in the river, which rose from 0.03 mg L⁻¹ in October to 0.09 mg L⁻¹ in November, and then a decreasing trend throughout the rest of the year (Figure 12). NO₃ levels showed a similar spike but in

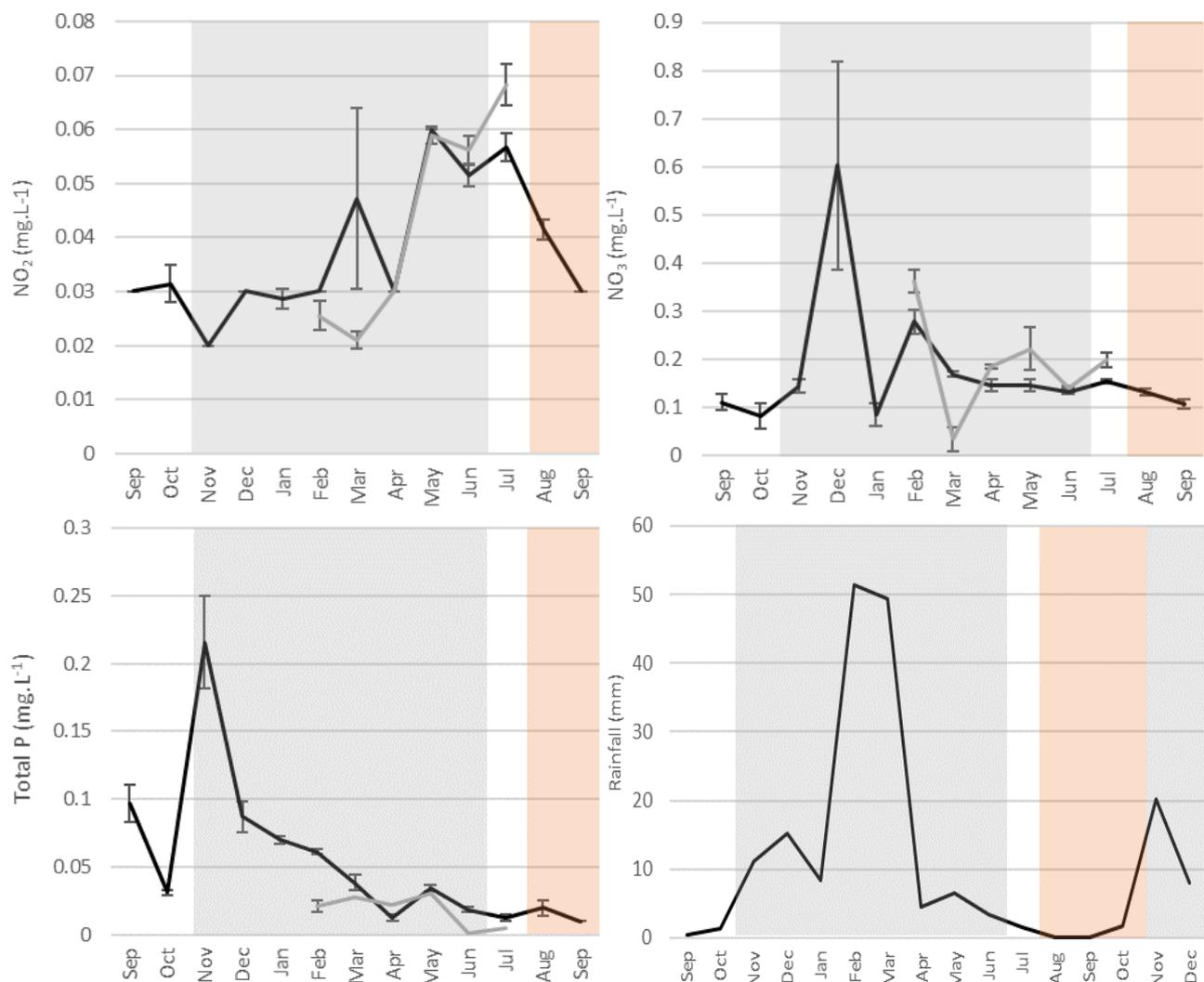


Figure 12. Environmental variables (NO₂, NO₃, total P and rainfall) in the river (black) and forest (grey). Error bars indicate Standard Deviation where appropriate. The grey boxes indicate the approximate duration of the wet season, and the orange box the approximate fire season (August to end of October).

this case over the months of November and December, when levels increased from 0.14 mg L⁻¹ to 0.60 mg L⁻¹. NO₂ levels were variable with a general increase over time during the wet season (from 0.02 mg L⁻¹ in November to 0.05 mg L⁻¹ in June) and a subsequent decrease during the dry season (from 0.04 mg L⁻¹ in July to 0.03 mg L⁻¹ in September).

For the river, percentage fish mortality was calculated for each month of trapping (Figure 13). In October 2014, a maximum level of 50 % fish mortality occurred. This dropped to less than 10 % for much of the wet season when DO levels were also higher, but there was an increasing trend of mortality after the onset of the dry season in July, which corresponded with decreasing DO levels and was confirmed by a strong negative correlation between average DO (mg L⁻¹) and mortality rate (PPC $r = -0.754$, $n = 13$, $p = 0.003$). Mortality rate showed no correlation with any other environmental variables that were measured.

DISCUSSION

Our results provide an initial picture of the composition of this Bornean peatland fish community and how it changed over the year-long sampling period in conjunction with various environmental variables. These data establish a valuable baseline for future monitoring of forest and river fish in the Sebangau peatland and a comparison for studies conducted in other peatlands throughout the region. The authors recognise that the sampling methods will be selective against certain species and that our methods sample a specific fish guild that is large enough to be trapped and is attracted to the bait used. However, methods used to exhaustively sample fish can be intensive and destructive to fish habitats and also require the collection and preservation of many specimens. This can be argued to be counter to conservation best practice, particularly as the population sizes of species and their distributions in PSF habitats are poorly understood. Therefore, we propose that the *tampirai* trapping method should be used for future monitoring.

Whilst the survey methods used in this study may not have been exhaustive, the resulting list of 55 species is longer than those presented by Page *et al.* (1997) who reported 34 fish species in the Sebangau River and Forest, and by Haryono (2012) who reported only eleven species in the Sebangau River. One of the eleven species reported by Haryono (2012) (*Hemibagrus nemurus*) was not recorded in our survey and is likely to have been a mis-

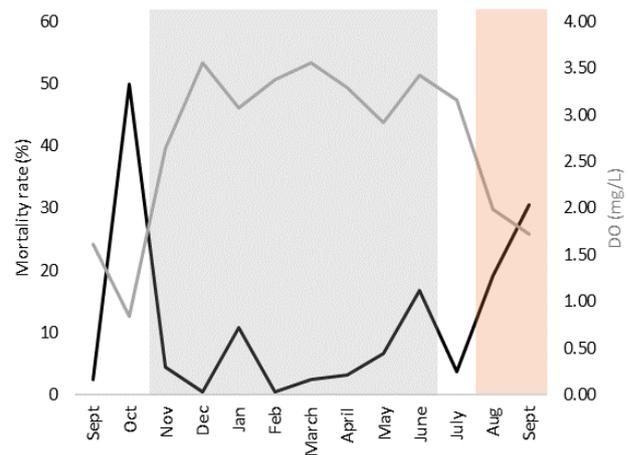


Figure 13. Monthly mortality rate (%; black) and dissolved oxygen levels (mg L⁻¹; grey) in the Sebangau River from September 2014 to September 2015. The grey box indicates the approximate duration of the wet season and the orange box the approximate fire season (August to end of October).

identification of another Bagridae species (Ng 2012). Haryono (2012) also reports the presence of *Pristolepis fasciata*, which is difficult to differentiate from the species *Pristolepis grootii* (Froese & Pauly 2017) which was recorded in our survey. Page *et al.* (1997) report the presence of both of these species in the Sebangau; furthermore, they report the presence of *Parosphromenus parvulus*, *Sphaerichthys selatanensis* (which is difficult to distinguish from *S. osphromenoides* as identified in our survey) and *S. vaillanti*, as well as *Chendol keelini*. This indicates that the species number reported here is likely to increase with further surveys, and that additional taxonomic work is needed, for example on species from the genera *Pristolepis* and *Sphaerichthys*. Sule *et al.* (2016) recently compiled lists of fish species recorded in Malaysian PSF. They list 114 species from North Selangor, 49 from Paya Beriah, 13 from multiple sites in Johor, 58 from multiple sites in Pahang, and nine from Pahang and Terengganu. In Malaysian Borneo, 31 species from 12 families and 40 species belonging to 13 families have been recorded from peatlands in Sabah and Sarawak, respectively. Concurring with Sule *et al.* (2016), most of the species we captured were from the Cyprinidae family, followed by Osphronemidae, with equal numbers from Bagridae and Siluridae. While direct comparisons are difficult owing to variations in sampling effort and environmental conditions, the Sebangau does appear to be confirmed as a notable area for fish diversity amongst peat swamp forests since it has the highest fish species richness recorded

in a Bornean peatland ecosystem to date (55 species). Furthermore, this is a greater number of species than has been noted from at least three of the five sites in Peninsular Malaysia reported by Sule *et al.* (2016).

With regard to temporal trends of species turnover it was observed that species similarity in the river decreased during the transition between dry and wet seasons and that this corresponded to a decrease in species richness for these months. Increased rainfall and river water depth at the onset of the wet season is likely to lead to decreased ‘catchability’ of the fish during these months, but as water levels decrease following the onset of the dry season, there is a corresponding increase in ‘catchability’ along with species richness and turnover.

The higher fish species richness in the Sebangau River, compared to the forest, could be a consequence of the larger area encompassed by the surveys in the river compared to those in the forest. Water body dimensions are likely to play a role in fish ‘catchability’ in the two locations, and will also affect catch sizes and species richness. Predicted species richness was higher for the river than for the forest and, while future surveys using a variety of methods would help to elucidate whether the difference is real or an artefact of this study, there are various reasons why we might expect to find more species in the river compared to the forest. First, with greater volumes of water, a greater surface area of land below and to the sides of the water body and a much wider variation in water flow rates, the river potentially has a greater number of niches than the forest. In the forest, water depth imposed a limitation on the fish surveys, which ceased during the dry season owing to very low water levels in both pools and canals. In temperate rivers, Grenouillet *et al.* (2004) found that increased stream size was associated with increased fish species richness (Gorman & Karr 1978, Taylor & Warren 2001). It is also well established that water depth influences fish assemblages in streams (Harvey & Stewart 1991, Matthews 1998, Carvalho & Tejerina-Garro 2014, Marion *et al.* 2015), as deep water is related to environmental stability (*e.g.* due to damping of temperature fluctuations) and allows greater vertical separation of fish microhabitats (*e.g.* Baker & Ross 1981, Gorman 1988a, 1988b; D.A. Jackson *et al.* 2001). Increased habitat stability favours higher species richness and abundance (Schlosser 1982, Winemiller *et al.* 2000, Grenouillet *et al.* 2004, Jardine *et al.* 2015); thus, water depth can play a significant role in determining habitat diversity and consequently fish assemblage structure and species diversity (Sheldon 1968, Evans & Noble 1979, Schlosser 1982, R.B. Jackson *et al.* 2001).

In addition, the forest pools and canals had consistently lower water DO levels compared to the river. This is probably due to the inherent nature of the aquatic habitat in peat swamp forests, where DO levels are kept low due to the high amount of tannins in the water (from the high organic matter content of the peat), with the accumulation of decaying organic matter depleting DO levels. Additionally, there is low or no water flow (especially in the pools) which further ensure low levels of DO regardless of the lower surface temperatures of forest water bodies (Yule & Gomez 2009). Low concentrations of DO can make water uninhabitable for certain fish species (Kramer 1987, Goodman & Campbell 2007, Zhang *et al.* 2009, Essington & Paulsen 2010), therefore the forest is likely to be a more challenging environment for fish survival compared to the river. This is also supported by Beamish *et al.* (2003) who found that Malaysian peat fish assemblages which were relatively rich in species and numerical abundance were associated with habitats offering comparatively high levels of DO. Additional data collection using methods to analyse differences in water quality in the forest and river (*e.g.* tannin quantities) could further elucidate these correlations. DO levels will also depend on mixing of the water caused by turbulence and water flow, but we were unable to collect data on these factors for the river. Therefore, further sampling of these environmental variables is highly recommended for future research to allow a more complete evaluation of the differences between the forest and river environments.

Across all species richness estimators, there was a clear under-estimation for the forest species, as more species were trapped than predicted. This could relate to the aforementioned ‘catchability’ of forest species which was lower than in the river, *i.e.* making it harder to catch new species, but this does not mean that they were not there. Using a variety of other sampling methods, such as nets with a smaller mesh size, could provide further insights into any potential bias introduced by using traps in the forest compared to the river. Furthermore, the estimators themselves are also subject to bias because they all tend to underestimate true diversity (O’Hara 2005). The Chao-1 estimator was originally derived as a ‘minimum asymptotic estimator’ (Chao 1984), but Gotelli & Colwell (2010) posit that all other estimators should be treated as estimating the lower bound on species richness. Nevertheless, both the survey results and all the estimated total species richness results indicate lower fish species diversity in the forest compared to the river.

Fish mortality was significantly correlated to DO levels (Figure 13) and to none of the other measured

environmental variables. pH in the river increased in October when there was a spike in mortality rate, however there was no statistically significant correlation overall between pH and mortality. Longer term data collection with larger sample sizes would allow future detailed analyses of interactions between environmental variables, CPUE and mortality rates. Worthington (2016) also concluded that low DO concentrations during the study period were the main determinant of high rates of fish mortality. Based on the correlation between surface water DO levels and fish mortality, any future fish surveys should take this into consideration by carrying out preliminary measurements of surface water DO levels to determine whether conditions are suitable for setting traps. Based on our data, the best months for reducing mortality in trap-based fish surveys were between November and July (mortality rate 0–17%), during which time DO levels were above 1.98 mg L⁻¹. For future fish surveys, care should be taken when DO levels fall below this value, and if surveys result in high rates of fish mortality they should be discontinued.

Increased water turbidity can have significant impacts on aquatic ecology, for example by impairing underwater visibility and, thereby, the ability of fish that forage by sight to feed (Utne-Palm 2002). It can also cause harm to the respiratory systems of fish (Kennedy *et al.* 2004), while short-term increases in turbidity have been found to lead to immediate behavioural changes in fish populations (*e.g.* Gray *et al.* 2011 found a significant shift from fish displaying territorial and courting behaviours to foraging behaviours in their experiments with Lake Malawi cichlids). As the difficulty of finding food increases, fish are likely either to move away from the impacted area or to prioritise processes other than feeding to survive. It could therefore be expected that increasing turbidity, as indicated by decreasing Secchi disk depths, would lead to lower CPUE, but the results (when considering daily rather than monthly measurements) indicate the complete opposite. More long-term data collection is recommended to clarify this relationship between fish catch and turbidity, but we provide some suggestions as to why this relationship may occur. Turbidity changes are dependent on both suspended sediments and organic materials, as well as algae in the water; thus, the higher fish catches could be due to there being higher levels of small food items for the fish in the water at times of greater turbidity. Additionally, the correlation between daily turbidity and CPUE may be influenced by the type of fish species being caught. The fish traps used were selective against bigger fish, which are likely to be

the bigger carnivorous species, and were more effective in capturing species of lower trophic levels. The latter tend to be omnivorous and planktivorous/algivorous species. Indeed, there was a dominance of omnivorous fish in the river, with a high proportion of these being *Osteochilus spilurus*. These smaller prey species are more vulnerable to predation in clearer waters, so during times of lower water turbidity they may stay close to vegetated areas and 'safer' locations, or may suffer higher predation levels from other fish. Therefore, catches of these smaller species are likely to increase during periods of higher turbidity. The effects of turbidity on fish are likely to be species-specific and further studies could usefully elucidate the complex relationships between fish behaviours and environmental conditions.

CPUE in the river was not correlated with any environmental variables apart from river depth. This is a surprising result, as CPUE might be expected to correlate with DO level, rainfall, temperature or pH. Rather than ruling out any influence of these variables, we believe instead that this illustrates the difficulty of trying to untangle the complex relationships between fish behaviour and the full spectrum of habitat factors on the basis of limited measurements (we gathered data only from the river surface, for example). Our methods were chosen for practical reasons and we recommend that, if resources allow, future monitoring based on the methods presented here should run alongside more in-depth and exhaustive studies on the relationships between fish behaviour and environmental variables.

There were notable temporal variations in nutrient levels in the river during the study period. The spikes in nutrient levels at the onset of the wet season are probably due to increased nutrient runoff from both the peat dome and the small town of Kereng Bangkirai (see Figure 1) and its adjacent agricultural land upstream. The sampling period for nutrient levels in the forest habitats was too short-term to determine any seasonal trends (running only from February to June 2015, *n* = 5). Continued long-term monitoring of nutrient levels could help clarify monthly trends and allow comparisons between habitats. Furthermore, investigations of the occurrence and growth of algae is recommended in order to further explore relationships between nutrient levels and food availability for algivorous fish species. Despite being incomplete, the initial dataset on nutrient levels collected during this study provides a baseline for future monitoring.

Measurements taken in November and December, immediately after the end of the 2015 fire season, did not reveal any statistically significant changes in DO, water temperature or water turbidity. Fires can

significantly increase sediment loads in peatland rivers following heavy rainfall, which would be expected to increase water turbidity (Maltby *et al.* 1990, Brown *et al.* 2015). Increased turbidity will have an impact on both fish and fishing. Indeed, some authors identify increased turbidity as the greatest threat to aquatic fauna (Beschta 1990, Beaty 1994, Rieman *et al.* 1997, Benda *et al.* 2003, Meyer & Pierce 2003). However, the data from this study showed no significant changes in water turbidity following the 2015 fires. This could be due to measurements being taken after there had been sufficient rainfall to clear the upper reaches of the Sebangau River of much of its sediment load. Holden *et al.* (2012) found that post-fire organic carbon loss occurs very rapidly (within a few weeks) after fire, while Moore *et al.* (unpublished data) found an immediate post-fire enhancement of dissolved organic carbon losses from tropical peatlands. Given that carbon loss is probably linked directly to sediment transport (Grieve & Gilvear 2008, Shuttleworth *et al.* 2014), it is possible that a post-fire period of enhanced sediment loss and increased turbidity had already come to an end by the time the post-fire measurements were made at the start of the wet season.

There was, however, a significant decrease in river pH in the post-fire period, which also corresponded to a drop in CPUE. This result is in agreement with reports from local fishermen that post-fire fish catches in the Sebangau river were extremely poor (Dudin, personal communication 14 Dec 2015). A decrease in river pH could be due to the fire damaging the soil structure and burning organic matter (Lyon & O'Connor 2008, Brown *et al.* 2014), leading to a release of organic acids and other low-pH substances (Page *et al.* 2002, Holden *et al.* 2012, Moore *et al.* 2013, Jauhiainen *et al.* 2016). Likewise, the pH of water in temperate streams can fall as low as 3 when organic acids are flushed out of peats after high precipitation events (Rothwell *et al.* 2005). A decrease in river pH is likely to cause changes in the behaviour and, potentially, survival of fish, as pH changes affect the ion and acid-base regulatory mechanisms in their gills as well as mucus secretion and gill structure (McDonald 1983, Laurent & Perry 1991, Kwong *et al.* 2014). Therefore, a decrease in pH could either cause local fish mortality or act as a trigger for fish to migrate to other parts of the river with more favourable pH levels. The behaviour and sensitivity to pH fluctuations of the river and forest fish is a key knowledge gap, as is the exact geochemical mechanism behind the pH decrease in the river. The evidence from this study strongly suggests that the post-fire increase in river

acidity impacted negatively on local fish catches, which in turn has negative implications for human livelihoods and wellbeing.

Continued monitoring of fish populations using the same sample locations as in this study is recommended in order to acquire longer-term datasets on fish catches and, hence, on population trends and water quality. This will facilitate ongoing evaluation of river health which is vital given the role that fish and fishing play in providing an important source of protein and income for local human communities. Continued research will not only improve our understanding of aquatic environments in peat swamp forests, but also the consequences of environmental changes for human communities. Fish constitute one of the clearest links between people, their livelihoods and their environment. In areas with high dependence on fish for livelihoods, fish research and conservation projects could provide excellent opportunities to increase the relevance of environmental research to local communities and thus, potentially, to increase local support for conservation projects.

ACKNOWLEDGEMENTS

Many thanks to the research assistants who supported this study in the field: Iwan, Ahmad, Krisyoyo, Karno and Bustani Arifin Unyil. Thanks also to the Centre for International Cooperation in Sustainable Management of Tropical Peatlands at the University of Palangka Raya for their support of the project; RISTEK for research permissions; plus all BNF staff and volunteers who assisted with logistics and fieldwork. Thanks to Hendra Tommy and Dr. Xingli Giam for their help with fish identification. Finally, thanks to The Rufford Foundation and the International Peatland Society for their vital funding.

REFERENCES

- Aziz, A., Rahmman, M. & Ahmed, A. (2012) Diversity, distribution and density of estuarine phytoplankton in the Sundarban Mangrove Forests, Bangladesh. *Bangladesh Journal of Botany*, 41(1), 87–95.
- Baker, J.A. & Ross, S.T. (1981) Spatial and temporal resource utilization by southeastern cyprinids. *Copeia*, 1981(1), 178–189.
- Beamish, F.W.H., Beamish, R.B. & Lim, S.L.H. (2003) Fish assemblages and habitat in a Malaysian blackwater peat swamp. *Environmental Biology of Fishes*, 68(1), 1–13.

- Beatty, K.G. (1994) Sediment transport in a small stream following two successive forest fires. *Canadian Journal of Fisheries and Aquatic Science*, 51(12), 2723–2733.
- Benda, L., Miller, D., Bigelow, P. & Andras, K. (2003) Effects of post-wildfire erosion on channel environments, Boise River, Idaho. *Forest Ecology and Management*, 178(1–2), 105–119.
- Beschta, R.L. (1990) Effects of fire in water quantity and quality. In: Walstad, J.D., Radosevich, S.R. & Sandberg, D.V. (eds.) *Natural and Prescribed Fire in Pacific North-West Forests*, Oregon State University Press, Corvallis, 219–231.
- Bodamer, B.L. & Bridgeman, T.B. (2014) Experimental dead zones: two designs for creating oxygen gradients in aquatic ecological studies. *Limnology and Oceanography*, 12(7), 441–454.
- Brown, L.E., Holden, J. & Palmer, S.M. (2014) *Effects of Moorland Burning on the Ecohydrology of River Basins*. Full report of key findings from the EMBER project, University of Leeds, UK, 19 pp. Online at: <https://pdfs.semanticscholar.org/a067/286ca602c4f3e3593d6eb7a86af1c65ed298.pdf>, accessed 17 Aug 2018.
- Brown, L., Holden, J., Palmer, S., Johnstone, K., Ramchunder, S. & Grayson, R. (2015) Effects of fire on the hydrology, biogeochemistry, and ecology of peatland river systems. *Freshwater Science*, 34(4), 1406–1425.
- Carvalho, R.A. & Tejerina-Garro, F.L. (2014) Environmental and spatial processes: what controls the functional structure of fish assemblages in tropical rivers and headwater streams? *Ecology of Freshwater Fish*, 24(2), 317–328.
- Chao, A. (1984) Nonparametric estimation of the number of classes in a population. *Scandinavian Journal of Statistics*, 11(4), 265–270.
- Chisholm, R.A., Wijedasa, L.S. & Swinfield, T. (2016) The need for long-term remedies for Indonesia's forest fires. *Conservation Biology*, 30(1), 5–6.
- Chokkalingam, U., Suyanto, Permana, R.P., Kurniawan, I., Mannes, J., Darmawan, A., Khususyiah, N. & Sustanto, R.H. (2007) Community fire use, resource change, and livelihood impacts: The downward spiral in the wetlands of southern Sumatra. *Mitigation and Adaptation Strategies for Global Change*, 12, 75–100.
- Colwell, R.K. (2013) *EstimateS, Version 9.1: Statistical Estimation of Species Richness and Shared Species from Samples*. Software and User's Guide. Online at: <http://viceroy.eeb.uconn.edu/estimates/index.html>, accessed 18 Aug 2018.
- Dodemaide, D.R., Matthews, T.G., Iervasi, D. & Lester, R. (2018) Anthropogenic water bodies as drought refuge for aquatic macroinvertebrates and macrophytes. *Science of The Total Environment*, 616–617, 543–553.
- Dommain, R., Cobb, A.R., Joosten, H., Glaser, P.H., Chau, A.F.L., Gandois, L., Kai, F.-M., Noren, A., Salim, K.A., Su'ut, N.S.H. & Harvey, C.F. (2015) Forest dynamics and tip-up pools drive pulses of high carbon accumulation rates in tropical peat dome in Borneo (Southeast Asia). *Journal of Geophysical Research: Biogeosciences*, 120, 617–640.
- Dudgeon, D. (2000) The ecology of tropical Asian rivers and streams in relation to biodiversity conservation. *Annual Review of Ecology and Systematics*, 31, 239–263.
- Eisenreich, S.J., Bannerman, R.T. & Armstrong, D.E. (1975) A simplified phosphorus analysis technique. *Environmental Letters*, 9(1), 43–54.
- Essington, T.E. & Paulsen, C.E. (2010) Quantifying hypoxia impacts on an estuarine demersal community using a hierarchical ensemble approach. *Ecosystems*, 13, 1035–1048.
- Evans, J.W. & Noble, R.L. (1979) The longitudinal distribution of fishes in an east Texas stream. *The American Midland Naturalist*, 101(2), 333–334.
- Field, R.D., van der Werf, G.R., Fanin, T., Fetzer, E.J., Fuller, R., Jethva, H., Levy, R., Livesey, N.J., Luo, M., Torres, O. & Worden, H.M. (2016) Indonesian fire activity and smoke pollution in 2015 show persistent nonlinear sensitivity to El Niño-induced drought. *Proceedings of the National Academy of Sciences of the United States of America*, 113(33), 9204–9209.
- Froese, R. & Pauly, D. (eds.) (2017) FishBase. In: Roskov, Y., Abucay, L., Orrell, T., Nicolson, D., Bailly, N., Kirk, P.M., Bourgoin, T., DeWalt, R.E., Decock, W., De Wever, A., van Nieuwerkerken, E., Zarucchi, J. & Penev, L. (eds.) *Catalogue of Life: 25th August 2017*. ITIS/Species 2000, Naturalis, Leiden, The Netherlands; digital resource at www.catalogueoflife.org/col
- Geeraert, N., Omengo, F.R., Borges, A.V., Govers, G. & Bouillon, S. (2017) Shifts in the carbon dynamics in a tropical lowland river system (Tana River, Kenya) during flooded and non-flooded conditions. *Biogeochemistry*, 132, 141–163.
- Giam, H., Koh, L.P., Tan, H.H., Miettinen, J., Tan, H.T.W. & Ng, P.K.L. (2012) Global extinctions of freshwater fishes follow peatland conversion in Sundaland. *Frontiers in Ecology and the Environment*, 10, 465–470.
- Goodman, R.L. & Campbell, J.G. (2007) Lethal levels of hypoxia for gulf coast estuarine animals.

- Marine Biology*, 152(2), 37–42.
- Gorman, O.T. (1988a) An experimental study of habitat use in an assemblage of Ozark minnows. *Ecology*, 69, 1239–1250.
- Gorman, O.T. (1988b) The dynamics of habitat use in a guild of Ozark minnows. *Ecological Monographs*, 58, 1–18.
- Gorman, O.T. & Karr, J.R. (1978) Habitat structure and stream fish communities. *Ecology*, 59, 507–515.
- Gotelli, N.J. & Colwell, R.K. (2010) Estimating species richness. In: Magurran, A.E. & McGill, B.J. (eds.) *Biological Diversity: Frontiers in Measurement and Assessment*, Oxford University Press, Oxford, 39–54.
- Gray, S.M., Sabbah, S. & Hawryshyn, C.W. (2011) Experimentally increased turbidity causes behavioural shifts in Lake Malawi. *Ecology of Freshwater Fish*, 20(4), 529–536.
- Grenouillet, G., Pont, D. & Hérisse, C. (2004) Within-basin fish assemblage structure: the relative influence of habitat versus stream spatial position on local species richness. *Canadian Journal of Fisheries and Aquatic Science*, 61, 93–102.
- Grieve, I. & Gilvear, D. (2008) Effects of wind farm construction on concentrations and fluxes of dissolved organic carbon and suspended sediment from peat catchments at Braes of Doune, central Scotland. *Mires and Peat*, 4(03), 1–11.
- Harvey, B.C. & Stewart, A.J. (1991) Fish size and habitat depth relationships in headwater streams. *Oecologia*, 87, 336–342.
- Haryono (2012) Fish fauna of Central Kalimantan peatland waters in rainy season. *Journal Iktiologi Indonesia*, 12(1), 83–91.
- Hedström, P., Bystedt, D., Karlsson, J., Bokma, F. & Byström, P. (2017) Brownification increases winter mortality in fish. *Oecologia*, 183, 587–595.
- Holden, J., Chapman, P., Palmer, S., Kay, P. & Grayson, R. (2012) The impacts of prescribed moorland burning on water colour and dissolved organic carbon: A critical synthesis. *Journal of Environmental Management*, 101, 92–103.
- Ikomi, R.B., Arimoro F.O. & Odihirin, O.K. (2005) Composition, distribution and abundance of macroinvertebrates of the upper reaches of River Ethiopie, Delta State, Nigeria. *Zoologist*, 3, 68–81.
- Jackson, D.A., Peres-Neto, P.R. & Olden, K.D. (2001) What controls who is where in freshwater fish communities—the roles of biotic, abiotic, and spatial factors. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 157–170.
- Jackson, R.B., Carpenter, S.R., Dahm, C.N., McKnight, D.M., Naiman, R.J., Postel, S.L. & Running, S.W. (2001) Water in a changing world. *Ecological Applications*, 11(4), 1027–1045.
- Jardine, T.D., Bond, N.R., Burdord, M.A., Ward, D.P., Bayliss, P., Davies, P.M., Douglass, M.M., Hamilton, S.K., Kennard, M.J., Melack, J.M., Naiman, R.J., Pettit, N.E., Pusey, B.J., Warfe, D.M. & Bunn, S.E. (2015) Does flood rhythm drive ecosystem responses in tropical riverscapes? *Ecology*, 96(3), 684–692.
- Jauhainen, J., Page, S.E. & Vasander, H. (2016) Greenhouse gas dynamics in degraded and restored tropical peatlands. *Mires and Peat*, 17(06), 1–12.
- Kennedy, B.W., Whitman, M.S., Burrows, R.L. & Richmond, S.A. (2004) *Assessment of Fish Habitat, Water Quality, and Selected Contaminants in Streambed Sediments in Noyes Slough, Fairbanks, Alaska, 2001–2002*. Water-Resources Investigations Report 03-4328, U.S. Geological Survey, Anchorage, Alaska, 62 pp.
- Kottelat, M. & Whitten, A.J. (1993) *Freshwater Fishes of Western Indonesia and Sulawesi*. Periplus Editions, Jakarta, 221 pp.
- Kramer, D.L. (1987) Dissolved oxygen and fish behaviour. *Environmental Biology of Fishes*, 18(2), 81–92.
- Kwong, R.W.M., Mumai, Y. & Perry, S.F. (2014) The physiology of fish at low pH: the zebrafish as a model system. *Journal of Experimental Biology*, 217, 651–662.
- Laurent, P. & Perry, S.F. (1991) Environmental effects on fish gill morphology. *Physiological Zoology*, 64(1), 4–25.
- Li, W. & Li, Z. (2009) *In situ* nutrient removal from aquaculture wastewater by aquatic vegetable *Ipomoea aquatica* on floating beds. *Water Science and Technology*, 59(10), 1937–1943.
- Lyon, J.P. & O'Connor, J.P. (2008) Smoke on the water: Can riverine fish populations recover following a catastrophic fire-related sediment slug? *Austral Ecology*, 33(6), 794–806.
- Lyons, A.L. (2003) *The Importance of Non-Timber Forest Products to Livelihoods and Conservation in the Sebangau Catchment, Central Kalimantan, Indonesia*. BSc dissertation, University of East Anglia School of Development Studies, 48 pp.
- Maltby, E., Legg, C.J. & Proctor, M.C.F. (1990) The ecology of severe fires on the North York Moors. *Journal of Ecology*, 78, 490–518.
- Marion, C.A., Scott, M.C. & Kubach, K.M. (2015) Multiscale environmental influences on fish assemblage structure of South Atlantic coastal plain streams. *Transactions of the American Fisheries Society*, 144(5), 1040–1057.
- Matthews, W.J. (1998) *Patterns in Freshwater Fish*

- Ecology*. Springer Science and Business Media, Dordrecht, 784 pp.
- McDonald, D.G. (1983) The effects of H⁺ upon the gills of freshwater fish. *Canadian Journal of Zoology*, 61(4), 691–703.
- Merilä, J. (2015) Baiting improves CPUE in nine-spined stickleback (*Pungitius pungitius*) minnow trap fishery. *Ecology and Evolution*, 5(17), 3737–1742.
- Meyer, G.A. & Pierce, J.L. (2003) Climatic controls on fire-induced sediment pulses in Yellowstone National Park and central Idaho: a long-term perspective. *Forest Ecology and Management*, 178, 89–104.
- Moore, S., Evans, C.D., Page, S.E., Garnett, M.H., Jones, T.G., Freeman, C., Hooijer, A., Wiltshire, A., Limin, S.H. & Gauci, V. (2013) Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. *Nature*, 493, 660–663.
- Murphy, J. & Riley, J.P. (1962) A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta*, 27, 31–36.
- Ng, H.H. (2012) *Hemibagrus nemurus*. The IUCN Red List of Threatened Species 2012: e.T180954A1681839. Online at: <http://www.iucnredlist.org/details/180954/0>, accessed 19 Aug 2018. <http://dx.doi.org/10.2305/IUCN.UK.2012-1.RLTS.T180954A1681839.en>
- Ng, H.H. & Tan, H.H. (2011) *Silurichthys ligneolus*, a new catfish (Teleostei: Siluridae) from southern Borneo, Indonesia. *Zootaxa*, 2793, 56–62.
- Ng, P.K.L., Tay, J.B. & Lim, K.K.P. (1994) Diversity and conservation of blackwater fishes in Peninsular Malaysia, particularly in the north Selangor peat swamp forest. *Hydrobiologia*, 285, 203–218.
- Noor, Y.R., Heyde, J., & Suryadiputra, I.N. (2005) Community-based approach to peatland adaptation and management in Central Kalimantan, Jambi, South Sumatra, Indonesia. In: Murdiyarso, D. & Herawati, H. (eds.) *Carbon Forestry: Who Will Benefit?* Proceedings of workshop on carbon sequestration and sustainable livelihoods, held in Bogor on 16–17 February 2005, Center for International Forestry Research, Bogor, 124–138.
- O’Hara, R.B. (2005) Species richness estimators: how many species can dance on the head of a pin? *Journal of Animal Ecology*, 74, 375–386.
- Page, S.E., Rieley, J.O., Doody, K., Hodgson, S., Husson, S., Jenkins, P., Morrogh-Bernard, H., Otway, S. & Wilshaw, S. (1997) Biodiversity of tropical peat swamp forest: A case study of animal diversity in the Sungai Sebangau catchment of Central Kalimantan, Indonesia. In: Rieley, J.O. & Page, S.E. (eds.) *Biodiversity and Sustainability of Tropical Peatlands*, Samara Publishing Limited, Cardigan, 231–242.
- Page, S.E., Rieley, J.O., Shoty, O.W. & Weiss, D. (1999) Interdependence of peat and vegetation in a tropical peat swamp forest. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 354(1381), 1885–1897.
- Page, S.E., Siegert, F., Rieley, J.O., Boehm, H.D.V., Jaya, A. & Limin, S. (2002) The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, 420, 61–65.
- Page, S.E., Wust, R.A.J., Weiss, D., Rieley, J.O., Shoty, W. & Limin, S.H. (2004) A record of late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): Implications for past, present and future carbon dynamics. *Journal of Quaternary Science*, 19, 625–635.
- Petr, T. (1970) The bottom fauna of rapids of the Black Volta River in Ghana. *Hydrobiologia*, 35, 399–418.
- Portt, C.B., Coker, G.A., Ming, D.L. & Randall, R.G. (2006) *A Review of Fish Sampling Methods Commonly Used in Canadian Freshwater Habitats*. Canadian Technical Report of Fisheries and Aquatic Sciences 2604, Fisheries and Oceans Canada, Burlington ON, 51 pp.
- Preisendorfer, R.W. (1986) Secchi disk science: Visual optics of natural waters. *Limnology and Oceanography*, 31(5), 909–926.
- Rieman, B., Lee, D., Chandler, G. & Myers, D. (1997) Does wildfire threaten extinction for salmonids? Responses of redband trout and bull trout following recent large fires on the Boise National Forest. In: Greenlee, J. (ed.) *Proceedings—Fire Effects on Threatened and Endangered Species and Habitats Conference, Coeur d’Alene, Idaho*, International Association of Wildland Fire (IAWF), Fairfield, Canada, 47–57.
- Ríos-Villamizar, E.A., Piedade, M.T.F., Da Costa, J.G., Adeney, J.M. & Junk, W.J. (2014) Chemistry of different Amazonian water types for river classification: a preliminary review. *WIT Transactions on Ecology and the Environment*, 178, 17–28.
- Rothwell, J.J., Robinson, S.G., Evans, M.G., Yang, J. & Allot, T.E.H. (2005) Heavy metal release by peat erosion in the Peak District, southern Pennines, UK. *Hydrological Processes*, 19(15), 2973–2989.
- Sandén, P. & Håkansson, B. (1996) Long-term trends

- in Secchi depth in the Baltic Sea. *Limnology and Oceanography*, 41(2), 346–351.
- Schindler, I. & Linke, H. (2013) *Betta hendra* – a new species of fighting fish (Teleostei: Osphronemidae) from Kalimantan Tengah (Borneo, Indonesia). *Vertebrate Zoology*, 63(1), 35–40.
- Schlosser, I.J. (1982) Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs*, 52, 395–414.
- Sheldon, A.S. (1968) Species diversity and longitudinal succession in stream fishes. *Ecology*, 49, 193–98.
- Shuttleworth, E.L., Evans, M.G., Hutchinson, S.M. & Rothwell, J.J. (2014) Peatland restoration: controls on sediment production and reductions in carbon and pollutant export. *Earth Surface Processes and Landforms*, 40, 459–472.
- Sule, H.A., Ismail, A. & Amal, M.N.A. (2016) A review of the ichthyofauna of Malaysian peat swamp forest. *Tropical Agricultural Science*, 39(4), 421–458.
- Sulistiyanto, Y. (2005) *Nutrient Dynamics in Different Sub-types of Peat Swamp Forest in Central Kalimantan, Indonesia*. PhD thesis, University of Nottingham, 388 pp.
- Tachibana, H., Iqbal, R., Akimoto, S., Kobayashi, M., Ohno, K., Mori, A. & Itakura, T. (2003) Chemical characteristics of water at the upper reaches of the Sebangau River, Central Kalimantan, Indonesia. In: *Proceedings of the International Symposium on Land Management and Biodiversity in Southeast Asia*, Hokkaido University, Sapporo, Japan, 361–365.
- Tachibana, H., Iqbal, R., Akimoto, S., Kobayashi, M., Kanie, S., Mori, A., Itakura, T., Takahashi, H., Utosawa, K., Sumawijaya, N., Dohong, S., Darung, U. & Limin, S. (2006) Chemical characteristics of water at the upper reaches of the Sebangau River, Central Kalimantan, Indonesia. *Tropics*, 15(4), 411–415.
- Tarka Challenge (2012) *Fish Trap*. Online at: <http://rudistarkachallenge.blogspot.com/2012/09/fish-trap.html>, accessed 19 Aug 2018.
- Taylor, C.M., & Warren, M.L. (2001) Dynamics in species composition of stream fish assemblages: environmental variability and nested subsets. *Ecology*, 82, 2320–2330.
- Thornton, S.A. (2017) *(Un)tangling the Net, Tackling the Scales and Learning to Fish: An Interdisciplinary Study in Indonesian Borneo*. PhD thesis, University of Leicester, UK, 352 pp.
- Utne-Palm, A. (2002) Visual feeding of fish in a turbid environment: Physical and behavioural aspects. *Marine and Freshwater Behaviour and Physiology*, 35(1–2), 111–128.
- Weiss, D., Shoty, W., Rieley, J., Page, S., Gloor, M., Reese, S. & Martinez-Cortizas, A. (2002) The geochemistry of major and selected trace elements in a forested peat bog, Kalimantan, SE Asia, and its implications for past atmospheric dust deposition. *Geochimica et Cosmochimica Acta*, 66, 2307–2323.
- Winemiller, K.O., Tarim, S., Shormann, D. & Cotner, J.B. (2000) Fish assemblage structure in relation to environmental variation among Brazos River oxbow lakes. *Transactions of the American Fisheries Society*, 129, 451–468.
- Worthington, W. (2016) *Testing Fish Traps for the Purposes of Conservation Research on the Sabangau River, Central Kalimantan*. BSc dissertation, Department of Geography, University of Leicester, 57 pp.
- Yang, J.E., Kim, J.J., Skogley, E.O. & Schaff, B.E. (1998) A simple spectrophotometric determination of nitrate in water, resin, and soil extracts. *Soil Science Society of America Journal*, 62(4), 1108–1115.
- Yule, C.M. & Gomez, L.N. (2009) Leaf litter decomposition in a tropical peat swamp forest in Peninsular Malaysia. *Wetlands Ecology and Management*, 17(3), 231–241.
- Zhang, H., Ludsin, S.A., Mason, D.M., Adamack, A.T., Brandt, S.B., Zhang, X., Kimmel, D.G., Roman, M.R. & Boicourt, W.C. (2009) Hypoxia-driven changes in the behaviour and spatial distribution of pelagic fish and mesozooplankton in the northern Gulf of Mexico. *Journal of Experimental Marine Biology and Ecology*, 381, S80–S91.

Submitted 21 Nov 2017, final revision 07 Oct 2018
 Editor: Ab Grootjans

Author for correspondence:

Dr Sara Thornton, School of Geography, Geology and the Environment, University of Leicester, UK and Borneo Nature Foundation, Jl. Bukit Raya No. 82, Palangka Raya 73112, Central Kalimantan, Indonesia.
 Tel: +447518424922; E-mail: sat32@leicester.ac.uk or s.thornton.p@gmail.com

Appendix

Table A1. List of freshwater fish (Actinopterygii) species recorded in the Sebangau River and Forest together with IUCN Red List classifications (DD = data deficient, LC = least concern, VU = vulnerable) and Borneo endemic species assignments. None of these species are on the Indonesian protected species list. Data from Ng & Tan (2011) and Thornton (2017). Additional local and Bahasa Indonesia species names are provided in Thornton (2017).

ORDER / Family	Genus	Species	English name	IUCN	Borneo endemic?	
BELONIFORMES						
Zenarchopteridae	<i>Hemirhamphodon</i>	<i>chrysopunctatus</i>				
		<i>tengah</i>				
CYPRINIFORMES						
Cobitidae	<i>Kottelatlimia</i>	<i>cf. pristis</i>				
Cyprinidae	<i>Cyclocheilichthys</i>	<i>janthochir</i>			Endemic	
	<i>Desmopuntius</i>	<i>foerschi</i>	Foersch's fire barb		Endemic	
		<i>hexazona</i>	Six-banded tiger barb			
		<i>johorensis</i>	Striped barb			
		<i>rhomboocellatus</i>	Snakeskin barb		Endemic	
	<i>Eirmotus</i>	sp. ¹	Eight-banded barb			
	<i>Osteochilus</i>	<i>melanopleura</i>	Greater bony lipped barb		LC	
		<i>spilurus</i>			LC	
	<i>Rasbora</i>	<i>cephalotaenia</i>	Porthole rasbora			
		<i>dorciocelatta</i>	Eyespot rasbora			
		<i>kalbarensis</i>	Kalbar rasbora			Endemic
		<i>kalochroma</i>	Clown rasbora			
	<i>Striuntius</i>	<i>lineatus</i>	Lined barb			
	<i>Trigonopoma</i>	<i>gracile</i>	Blackstripe rasbora			
PERCIFORMES						
Anabantidae	<i>Anabas</i>	<i>testudineus</i>	Climbing perch	DD		
Channidae	<i>Channa</i>	<i>bankanensis</i>	Bangka snakehead			
		<i>gachua</i>	Forest snakehead	LC		
		<i>melanoptera</i>	Black finned snakehead			
		<i>micropeltes</i>	Giant snakehead	LC		
		<i>pleurophthalmus</i>	Oscellated snakehead			
		<i>striata</i>	Snakehead murrel	LC		
Helostomatidae	<i>Helostoma</i>	<i>temminckii</i>	Kissing gourami	LC		

ORDER / Family	Genus	Species	English name	IUCN	Borneo endemic?
Nandidae	<i>Nandus</i>	<i>nebulosus</i>	Bornean leaffish	LC	
Osphronemidae	<i>Belontia</i>	<i>hasselti</i>	Malay combtail		
	<i>Betta</i>	<i>anabatooides</i>	Giant betta		Endemic
		<i>foerschi</i>			Endemic
		<i>hendra</i>			Endemic
	<i>Luciocephalus</i>	<i>aura</i>	Peppermint pikehead		
		<i>pulcher</i>	Giant pikehead		
	<i>Sphaerichthys</i>	<i>acrostoma</i>	Giant chocolate gourami		Endemic
		<i>osphromenoides</i>	Chocolate gourami		
<i>Trichopodus</i>	<i>pectoralis</i>	Snakeskin gourami	LC		
Pristolepidae	<i>Pristolepis</i>	<i>grootii</i>	Indonesian leaffish		
SILURIFORMES					
Bagridae	<i>Leiocassis</i>	<i>micropogon</i>	Bumblebee catfish		
		sp.			
	<i>Mystus</i>	<i>nigriceps</i>	Twospot catfish		
		<i>olyroides</i>			Endemic
		sp.			
Chacidae	<i>Chaca</i>	<i>bankanensis</i>	Angler catfish	LC	
Clariidae	<i>Clarias</i>	<i>meladerma</i>	Blackskin catfish	LC	
		<i>nieuhofii</i>	Slender walking catfish	LC	
		<i>teijsmanni</i>	Airbreathing catfish		
	<i>Encheloclarias</i>	<i>tapeinopterus</i>		VU	
Schilbeidae	<i>Pseudeutropius</i>	<i>moolenburghae</i>	Sun catfish		
Siluridae	<i>Kryptopterus</i>	sp.	Striped glass catfish		
	<i>Ompok</i>	<i>leiacanthus</i>		DD	
	<i>Silurichthys</i>	<i>ligniolus</i>	Brown leaf catfish		Endemic
		<i>phaiosoma</i>	Hasselt's leaf catfish		
	<i>Wallago</i>	<i>leeri</i>	Striped wallago catfish		
SYNBRANCHIFORMES					
Mastacembelidae	<i>Macrognathus</i>	<i>aculeatus</i>	Lesser spiny eel		
		<i>maculatus</i>	Frecklefin eel	LC	
Synbranchidae	<i>Monopterus</i>	<i>albus</i>	Asian swamp eel	LC	

¹ Potentially new species based on inspection in the field and of photographs. Requires specimen for confirmation.

Table A2. Abundances of each species for each month (Sep 2014 to Dec 2015) and habitat type (**R** = River, **F** = Forest). 'x' indicates that no survey was conducted.

Genus	Species	Sep		Oct		Nov		Dec		Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Nov		Dec		
		R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	R	F	
<i>Cyclocheilichthys</i>	<i>janthochir</i>	9	0	43	0	39	0	5	0	0	0	1	0	5	0	2	0	1	0	5	0	0	0	0	0	3	0	14	0	13	0	
<i>Desmopuntius</i>	<i>foerschi</i>	1241	0	0	0	12	0	274	0	29	0	133	0	271	0	340	0	146	0	113	0	2894	0	623	0	16	0	0	0	9	0	
<i>Desmopuntius</i>	<i>hexazona</i>	0	x	0	x	0	x	1	x	0	x	1	0	3	2	0	2	0	0	0	0	0	156	0	7	x	0	x	0	x	0	x
<i>Desmopuntius</i>	<i>johorensis</i>	0	0	1	0	0	0	0	0	0	0	1	0	4	0	2	0	1	0	6	0	72	0	3	0	0	0	1	0	0	0	
<i>Desmopuntius</i>	<i>rhombocellatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	83	0	0	0	0	0	0	0	0	0	
<i>Eirmotus</i>	<i>sp.</i>	60	0	98	0	5	0	3	0	10	0	14	0	3	0	6	0	3	0	4	0	89	0	770	0	9	0	4	0	31	0	
<i>Osteochilus</i>	<i>melanopleura</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	
<i>Osteochilus</i>	<i>spilurus</i>	1520	0	975	0	106	0	3839	0	189	0	573	0	1099	0	720	0	146	0	234	0	3155	0	4014	0	1021	0	64	0	211	0	
<i>Rasbora</i>	<i>cephalotaenia</i>	508	x	11	x	36	x	114	x	49	x	117	1	352	6	569	4	558	1	239	1	1220	0	536	x	3	x	0	x	4	x	
<i>Rasbora</i>	<i>dorciocetta</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Rasbora</i>	<i>kalochroma</i>	0	x	0	x	0	x	0	x	0	x	0	315	0	422	0	306	0	545	0	320	0	45	0	x	0	x	0	x	0	x	
<i>Striuntius</i>	<i>lineatus</i>	436	0	83	0	28	0	1	0	0	0	5	0	7	0	2	0	6	0	11	0	995	0	684	0	30	0	12	0	0	0	
<i>Trigonopoma</i>	<i>gracile</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Channa</i>	<i>bankanensis</i>	0	x	0	x	0	x	1	x	0	x	0	9	0	12	0	10	0	31	0	40	0	9	0	x	0	x	0	x	0	x	
<i>Channa</i>	<i>gachua</i>	0	x	0	x	1	x	0	x	0	x	0	21	0	26	0	29	1	33	0	61	1	4	1	x	0	x	0	x	0	x	
<i>Channa</i>	<i>melanoptera</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Channa</i>	<i>micropeltes</i>	0	x	0	x	0	x	0	x	32	x	0	0	0	0	0	0	0	0	0	1	0	0	0	0	x	0	x	0	x	0	x
<i>Channa</i>	<i>pleurophthalmus</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0	1	0	1	0	2	0	3	0	1	0	0	0	0	0	
<i>Helostoma</i>	<i>temminckii</i>	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	2	0	0	0	0	0	
<i>Nandus</i>	<i>nebulosus</i>	335	x	101	x	257	x	22	x	1	x	3	3	5	0	21	0	19	0	56	3	840	1	642	x	77	x	24	x	16	x	
<i>Belontia</i>	<i>hasselti</i>	6	x	7	x	4	x	0	x	6	x	0	54	5	31	7	10	15	8	23	6	57	0	17	x	59	x	0	x	0	x	

Genus	Species	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Nov	Dec
		R F	R F	R F	R F	R F	R F	R F	R F	R F	R F	R F	R F	R F	R F	R F
<i>Betta</i>	<i>anabatoides</i>	0 x	0 x	0 x	0 x	0 x	0 20	0 39	0 54	0 99	0 137	0 52	0 x	0 x	0 x	0 x
<i>Betta</i>	<i>foerschi</i>	0 x	0 x	0 x	0 x	0 x	0 1	0 0	0 0	0 1	0 3	0 1	0 x	0 x	0 x	0 x
<i>Luciocephalus</i>	<i>aura</i>	0 0	0 0	4 0	1 0	0 0	0 0	1 0	3 0	0 0	2 0	3 0	71 0	82 0	0 0	0 0
<i>Luciocephalus</i>	<i>pulcher</i>	58 x	17 x	1 x	0 x	0 x	1 5	0 1	0 0	2 2	2 0	48 0	0 x	21 x	2 x	0 x
<i>Sphaerichthys</i>	<i>acrostoma</i>	458 0	190 0	4 0	0 0	0 0	0 0	5 0	0 0	0 0	7 0	8199 0	1939 0	212 0	0 0	0 0
<i>Sphaerichthys</i>	<i>osphromenoides</i>	0 x	1 x	0 x	0 x	0 x	0 0	0 1	0 1	2 0	15 0	0 0	0 x	93 x	0 x	0 x
<i>Trichopodus</i>	<i>pectoralis</i>	0 0	0 0	1 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0
<i>Pristolepis</i>	<i>grootii</i>	20 0	16 0	6 0	0 0	17 0	7 0	7 0	43 0	123 0	82 0	940 0	434 0	35 0	23 0	2 0
<i>Leiocassis</i>	<i>micropogon</i>	118 0	1 0	26 0	39 0	52 0	43 0	87 0	6 0	12 0	24 0	35 0	10 0	0 0	10 0	8 0
<i>Leiocassis</i>	sp.	0 x	0 x	0 x	0 x	0 x	0 0	0 1	0 0	0 0	0 0	0 0	0 x	0 x	0 x	0 x
<i>Mystus</i>	<i>olyroides</i>	890 x	235 x	371 x	154 x	52 x	39 0	49 3	110 0	90 2	138 0	1182 0	1579 x	80 x	48 x	17 x
<i>Chaca</i>	<i>bankanensis</i>	0 0	3 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0	0 0
<i>Clarias</i>	<i>meladerma/ teijsmanni</i>	26 x	46 x	2 x	0 x	0 x	2 10	0 2	6 10	8 0	6 2	460 0	22 x	16 x	0 x	0 x
<i>Clarias</i>	<i>nieuhofii</i>	0 x	0 x	0 x	0 x	0 x	0 10	0 14	0 7	0 5	0 2	0 0	0 x	0 x	0 x	0 x
<i>Encheloclarias</i>	<i>tapeinopterus</i>	0 x	0 x	0 x	0 x	0 x	0 57	0 112	0 73	0 40	0 9	0 6	0 x	0 x	0 x	0 x
<i>Pseudeutropius</i>	<i>moolenburghae</i>	0 0	3 0	136 0	0 0	0 0	0 0	0 0	0 0	1 0	0 0	0 0	3 0	4 0	0 0	0 0
<i>Kryptopterus</i>	sp.	26 x	13 x	146 x	35 x	25 x	41 2	30 2	21 1	16 0	45 0	19 0	38 x	54 x	0 x	0 x
<i>Ompok</i>	<i>leiacanthus</i>	0 x	0 x	0 x	0 x	0 x	0 27	0 25	0 38	1 42	1 33	0 0	0 x	0 x	28 x	33 x
<i>Silurichthys</i>	<i>phaiosoma</i>	2 x	0 x	1 x	0 x	1 x	0 25	0 52	0 29	1 39	0 22	2 0	1 x	0 x	0 x	0 x
<i>Macrogathus</i>	<i>aculeatus</i>	0 x	0 x	0 x	0 x	0 x	0 0	0 0	0 0	0 0	0 0	3 0	0 x	0 x	0 x	0 x
<i>Macrogathus</i>	<i>maculatus</i>	3 x	4 x	12 x	0 x	0 x	0 1	0 2	1 1	0 1	0 0	16 0	6 x	3 x	1 x	0 x
<i>Monopterus</i>	<i>albus</i>	0 x	0 x	0 x	0 x	0 x	0 0	0 0	0 0	0 1	0 0	0 0	0 x	0 x	0 x	0 x

Table A3. Statistical analysis (Spearman rho, r_s or Pearson's product correlation (PPC) if indicated) of the Sebangau River environmental variables: catch *per* unit effort (CPUE), fish body size (SL) and species richness. These analyses are between monthly averages of each variable and exclude post-fire data; n=13 except for Secchi disk depth data analysis where n=8 and correlation analysis was conducted with environmental variables for the corresponding months only (February–September 2015). * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

	CPUE	Body size (SL)	Species richness	Average DO	Average pH	Average depth of river	Average Secchi disk depth	Average temperature	Average rainfall	Average P	Average NO ₃
Average DO	-0.148	-0.181	-0.450								
Average pH	-0.121	-0.071	-0.132	-0.154							
Average depth of river	-0.571*	0.000	-0.455	0.714**	0.132						
Average Secchi disk depth	-0.5	-0.024	-0.216	-0.19	0.238	0.119					
Average temperature	0.049	-0.250	0.586* (PPC)	-0.324	0.253	-0.159	0.248 (PPC)				
Average rainfall	-0.429	0.179	-0.478	0.718**	0.297	0.784**	0.108	-0.393			
Average P	-0.126	0.654*	-0.359	0.044	0.121	0.214	-0.238	-0.269	0.523		
Average NO ₃	0.184	-0.327	-0.169	0.757**	0.033	0.561*	-0.431	-0.432	0.656*	0.146	
Average NO ₂	0.3	-0.447	0.403	0.147	-0.107	0.158	-0.024	0.362	-0.204	-0.48	0.192