

## Conditions Influencing Waterbirds in the Vasse-Wonnerup Wetland, Western Australia: Implications for Water Regime Management



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**Author contributions:** Sora Marin-Estrella prepared all data for analysis liaising with DWER, Murdoch University and DBCA. Chandra Salgado Kent undertook data analysis. Glenn Hyndes project managed. All three co-authors contributed to project planning and design, interpretation and preparation of the report.

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## Executive Summary

The Vasse-Wonnerup Wetland located in the South-West of Australia is recognised as a “Wetland of International Importance” under the Ramsar Convention and hosts tens of thousands of waterbirds each year. The wetland comprises the Vasse and the Wonnerup Estuaries and their exiting channels. Two surge barriers were constructed between the estuaries and Wonnerup Inlet at the beginning of the 20<sup>th</sup> century to prevent seawater incursion to surrounding agricultural lands. Due to nutrient enrichment and altered flows caused by river diversions and land drainage, and the surge barriers reducing flushing water flows, the system has experienced episodic algal blooms, anoxia and fish deaths. Since the late 20<sup>th</sup> century, the surge barriers have been opened in summer as part of a water management regime to allow fresh sea water input into the system to improve conditions.

The main aim of this study was to investigate the relationship between the abundance and guild distribution of waterbirds, water regime and food sources within the Vasse-Wonnerup Wetland through the analysis of the Integrated Ecological Monitoring (IEM) datasets and monthly bird and water regime data. The IEM dataset was generated from the Integrated Ecological Monitoring project that lies within the Revitalising Geographie Waterways scheme that aims to improve water quality, waterway health and management of Geographie waterways funded by the state government’s Healthy Estuaries WA program.

All relationships were explored using Generalized Additive Models (GAMs) as they allow modelling of nonlinear functional relationships. Specifically, GAMs were used to predict macrophyte percent volume index (PVI), benthic macroinvertebrate and fish densities, and counts of waterbirds for each species as a function of abiotic and biotic predictor variables. Models were fitted for each biotic group or species that had sufficient non-zero observations to result in reliable models. In addition, only those conditions considered to be directly relevant to the biotic group being modelled that had sufficient non-zero values were included. For waterbirds, only species-level (and not guild-level) modelling was undertaken to avoid the risk of incorrectly generalising patterns of dominant species to all species belonging to the same guild. Most models were too complex given the sample size and data attributes to include interactions (e.g., waterbird guild and month as interacting factors).

To achieve the aims of this study, five specific objectives were addressed. These objectives and respective main outcomes are summarised below.

*Objective 1: Explore the relationships and identify key variables that influence interactions between water regime, aquatic plants, benthic macroinvertebrates, fish, waterbird abundance and dominant waterbird guild distribution.*

- a. Waterbirds were classified according to their movement patterns as *migrants*, *nomadic*, *locally nomadic*, and *sedentary*, as movement influences when waterbirds occupy the wetland. Seasonality was evident in one long-distance migrant (Common Greenshank), six nomadic species (Australian Pelican, Black-winged Stilt, Red-necked Avocet, Australian Shelduck and Grey Teal), two locally nomadic species (Pacific Black Duck and Black Swan), one sedentary species (Red-capped Plover), and the Silver Gull which belongs to several movement categories. Counts of all seasonal nomadic species in addition to the Pacific Black Duck were also associated with the area of the wetland covered by water.
- b. Within the wetland, six waterbird guilds were identified and included *surface dabblers and filter feeders* (comprised of ducks), *shoreline waterbirds* (comprised of a mixture of a duck, chat, crane, swamphen, rail and gull), *small waders* (comprised of a sandpiper, a stilt, a stint, a curlew,

sandpipers, plovers, godwits, and knots, among others), *large waders* (ibises, herons, egrets and a spoonbill), *deep waterbirds and divers* (ducks, cormorants, grebes, a darter, a pelican, and the Black Swan) and *aerial divers* (terns, an osprey, an eagle, a kite, a kingfisher, and a harrier). Within guilds, waterbird species varied in whether they were migrants, nomadic, locally nomadic or sedentary. Counts for 17 of 65 total species could be modelled (i.e., four surface dabbler and filter feeders, one shoreline waterbird, four small waders, two large waders, four deep waterbirds and divers, and two aerial divers). The most important conditions associated with waterbird counts for these species were as follows:

- Surface dabbler and filter feeder species counts were associated with month and available water area at the time of counts and one month prior. The exception to these patterns was the Australasian Shoveler, for which month was not identified as important, but salinity and wind speed were. Counts of two of the four species belonging to this guild, the Pacific Black Duck (a known herbivore) and the Australasian Shoveler (known to feed on macrophytes and invertebrates), were also associated with biotic variables. Counts of the Pacific Black Duck were associated with *Lamprothamnium* PVI and of the Australasian Shoveler with mean total macrophyte PVI.
- Shoreline waterbird counts, which included the silver gull, were associated with month and wind direction.
- Wader (small and large) counts were associated with available water area, surface turbidity and wind direction one month prior to counts. Month was also an important variable associated with small wader counts. Counts of one species of large wader, the White-faced Heron known to feed on invertebrates, was associated with mean Arthropoda (comprised 49 taxa of Insecta and 24 of Crustacea) density.
- Deep water and diver species counts were associated with wind direction, surface salinity and turbidity, and variability in wind direction and surface salinity a month prior to counts. Available water area was also an important variable associated with Australian Pelican counts. Two species from this guild were also associated with biotic variables. The Black Swan (an herbivore) was associated with mean total macrophytes PVI and the Australian Pelican (a piscivore) was associated with the variability in Atherinidae density.
- Aerial diver counts were associated with available water area and wind speed. One of the species from this guild, the piscivorous White-bellied sea eagle, was also associated with the variability in the total fish density.

The nature of the associations varied among species and are described within the report.

*Objective 2: Identify key variables that influence abundance of the four Ramsar listed species; i.e., the Black-winged Stilt (*Himantopus himantopus*), Red-necked Avocet (*Recurvirostra novaehollandiae*), Australian Shelduck (*Tadorna tadornoides*), and Australasian Shoveler (*Anas rhynchos*), and the Black Swan (*Cygnus atratus*).*

- c. Black-winged Stilt: The most important abiotic condition associated with Black-winged Stilt counts was month, with maximum counts in December-January. The peak in counts was associated with area of water available one month prior to counts, and more specifically with maximum area of water available between depths of 0.1 and 0.2 m one month prior to counts.
- d. Red-necked Avocet: The most important abiotic condition associated with Red-necked Avocet counts was month, with maximum counts in March-May and minimum counts in August-September. An increase in Red-necked Avocet counts was associated with an increase in monthly average water area for depths between 0.3 and 0.5 m one month prior to counts.

- e. Australian Shelduck: The most important conditions associated with Australian Shelduck counts was month, with highest counts in December and lowest in August. The highest counts were associate with monthly minimum area of available water over depths between 0.3 and 0.5 m of around 275,000 - 400,000 m<sup>2</sup>.
- f. Australasian Shoveler: The most important conditions associated with counts for this waterbird species were mean wind speed and maximum surface water salinity, with a peak in counts at monthly maximum salinities of around 30‰ and counts increasing with increasing wind speed, with high uncertainty above approximately 10 km hr<sup>-1</sup>. Australasian Shoveler counts also increased with increasing average total macrophyte density up to approximately at low volume indices.
- g. Black Swan: The most important conditions associated with Black Swan counts were ecological year (defined as the beginning of April to the end of March the following year), month, and minimum surface salinity. The highest Black Swan counts were between July and December. Black Swan counts varied among years, with lowest counts in the ecological year corresponding with the beginning of April 2016 to the end of March 2017. Decreasing Black Swan counts were associated with increasing monthly minimum surface salinities, with the steepest drop in numbers at salinities above approximately 20‰. Black Swan counts were also associated with the average total macrophytes PVI, with an increase in counts associated with increase in PVI at low volume indices.

*Objective 3: Explore the relationships between macrophytes and herbivore waterbird guilds (including possible time lags) using seasonal data and monthly data collected between March 2017 and March 2020.*

- h. For the two species of herbivore waterbirds with sufficient numbers to include in models, the most important biotic variables associated with counts were maximum value of *Lamprothamnium* PVI and average total macrophytes PVI.
- i. For Black Swans, the average total macrophytes PVI was the most important biotic variable, with increasing counts associated with increasing PVI at low volume indices.
- j. For the Pacific Black Duck, the maximum *Lamprothamnium* PVI was the most important biotic variable, with increasing counts associates with increasing PVI.

*Objective 4: Develop a seasonal conceptual model for the wetlands including water regime and ecological components based on findings from this analysis.*

- k. A seasonal conceptual model has been developed (see Figure 17), outlining the broad changes in the waterbird species in relation to seasonal changes in environmental conditions of the VWWS and key abiotic and biotic parameters. In addition, a process conceptual model (see Figure 18) has been developed to outline the important predictor variables that directly or indirectly influence waterbird counts.
- l. All fish and Atherinidae densities were associated with month and benthic macroinvertebrate densities; with increasing densities of fish associated with increasing total invertebrate density at lower densities, and peak fish densities associated with Austral summer months.
- m. Benthic macroinvertebrate densities generally varied with region and bottom depth, while total invertebrate and polychaete densities tended towards a negative association with PVI of macrophytes, and particularly *Ulva* sp.

- n. Macrophyte PVI for all species varied among regions. Time of year was important for *Stuckenia*, *Althenia*, *Lamprothamium* and *Ulva*, with peaks in PVI associated with austral summer months. Surface salinity and turbidity were important for *Rhizoclonium*, *Cladophora* and *Ruppia*. Increasing *Rhizoclonium* and *Cladophora* PVI was associated with increasing salinity up to approximately 40‰ and decreasing turbidity. Decreasing *Ruppia* PVI was associated with increasing salinity and turbidity above approx. 40‰ and 10 NTU, respectively.

These results are being integrated into the seasonal conceptual model to be presented in the final report.

*Objective 5: Use findings from this analysis to identify critical times or conditions for the ecology of the system that may be impacted (either positively or negatively) by manipulation of the water regime (either by increasing fresh water flows off the catchment or increasing seawater inflows through the surge barriers).*

Based on the available data, it was not possible to relate the outcomes from the models on waterbird counts to water management regime. The models could not account for regional differences in environmental conditions due to bird counts occurring only once per region per month. Consequently, it is difficult to recommend specific manipulation of water regimes to manage waterbirds in the system. Increasing the number of years, and where possible increasing the number of counts per month (particularly at times around the period of water regime manipulation) will likely increase the power of the modelling to more accurately predict conditions and impacts of water regime management on waterbird counts.

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## 1 Introduction

The Vasse-Wonnerup Wetland, situated in the South-West of Australia, is a highly modified coastal system that encompasses two estuaries, the Vasse and the Wonnerup Estuaries and their exiting channels. In 1908, two surge barriers were constructed between the estuaries and Wonnerup Inlet to avoid seawater incursions to adjacent agricultural lands (Lane *et al.* 1997). The gates changed the system from an estuarine environment to a winter-fresh/summer-saline lagoons (WRM 2007). Nevertheless, the Vasse-Wonnerup Wetland was nominated as an International Important Wetland under the Ramsar Convention in 1990. It regularly hosts between 25,000 and 35,000 waterbirds each year (WRM 2007).

Different human stressors, nutrient enrichment and altered flows due to river diversions and land drainage have significantly affected the wetland. The surge barriers act as a partial barrier, reducing flushing flows (Lane *et al.* 1997). As a result of all the above, episodic algal blooms, anoxia and fish deaths have occurred in the system (Lane *et al.* 1997, Tweedley *et al.* 2019a). These impacts are exacerbated in summer, when dry/high temperature conditions reduce water levels and increase salinity (WRM 2007, Tweedley *et al.*, 2014). Since 1988, the surge barriers have been managed and opened in summer to allow fresh sea water input into the system to ameliorate the above impacts (Department of Water and Environmental Regulation (DWER) n.d.).

Wetland management is difficult because it is essential to have deep knowledge of the wetland ecosystem function and the species dependent on it. The system needs to be managed at different temporal and spatial scales, to the advantage of diverse species, sometimes with competitive needs (Ma *et al.* 2009). At the same time, managers need to balance ecosystem integrity, water quality and communities' perceptions. Since hydrology (water levels) is a major factor affecting habitat use by waterbirds, it is an important aspect to manage (De la Cruz *et al.* 2018, Ma *et al.* 2009, Özgencil *et al.* 2019).

Bird surveys have been carried out since 1980 and a regular monthly bird monitoring program began in 2015 by the Department of Biodiversity Conservation and Attractions (DBCA). These surveys indicate that waterbird numbers and species using the wetlands vary substantially on a seasonal and annual basis and in relation to water regime (DBCA 2020). Previous studies in the Vasse-Wonnerup Wetland also suggested that waterbird numbers and diversity were correlated with habitat and food availability, which in turn were related to water levels and quality (Bamford and Bamford 1995). However, no detailed study has been carried out in the system to evaluate the relationship between seasonal water regime, waterbird numbers and their food sources.

The Integrated Ecological Monitoring project lies within the Revitalising Geographie Waterways scheme, which seeks to improve waterway health through water quality and management. It is funded by the state government's Healthy Estuaries WA program, which supports restoration and management activities across the regions. The Integrated Ecological Monitoring project ran from March 2017 to June 2020 and its main objective was to study the relationships among the different trophic levels (macrophytes, invertebrates, fish and waterbirds) of the Vasse-Wonnerup Wetlands and the effects that seasonal water quality and water level management have on these trophic levels.

The main objective of the present study, which is contained within the Integrated Ecological Monitoring, was to investigate the relationship between the abundance and guild distribution of waterbirds, water regime and food sources in the Vasse-Wonnerup Wetlands through the analysis of the Integrated Ecological Monitoring (IEM) datasets and monthly bird and water regime data.

The specific objectives were to:

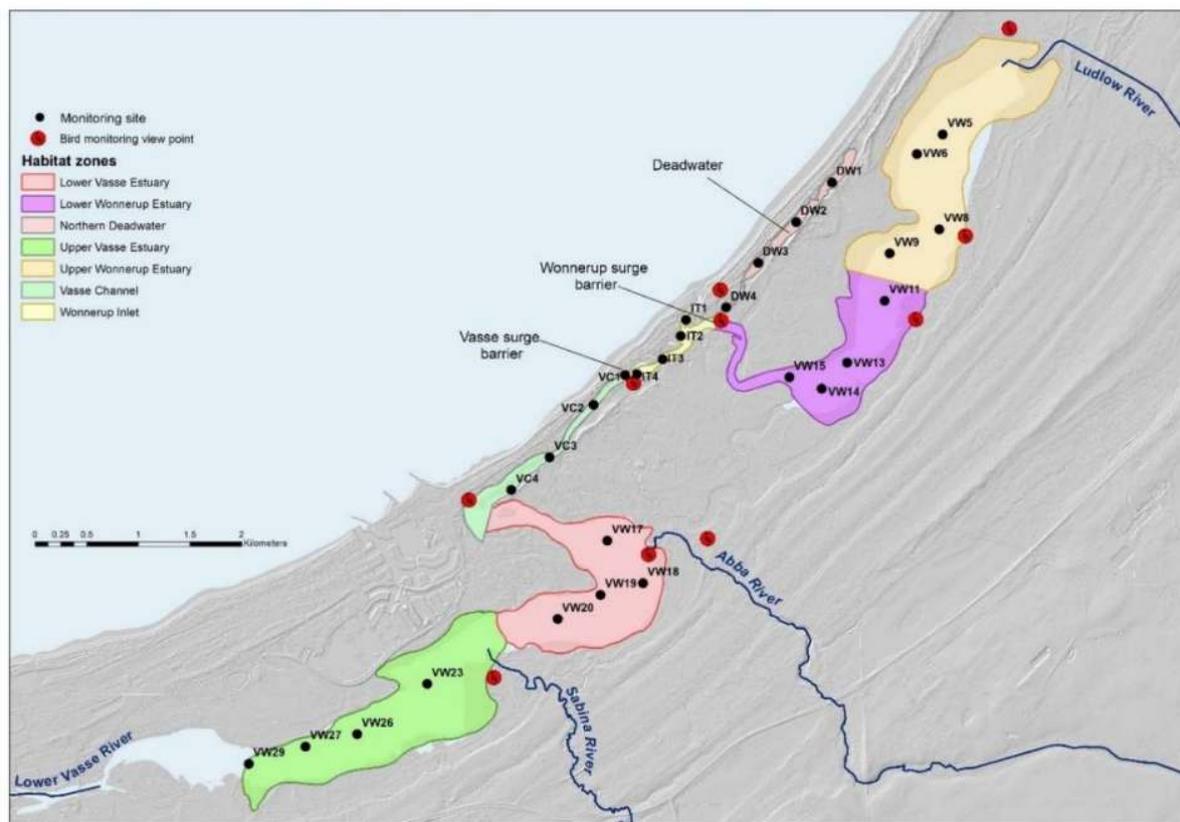
1. Explore the relationships and identify key variables that influence interactions between water regime, aquatic plants, benthic macroinvertebrates, fish, waterbird abundance and dominant guild distribution.
2. Identify key variables that influence abundance of the four Ramsar listed species; i.e., the Black-winged Stilt (*Himantopus himantopus*), Red-necked Avocet (*Recurvirostra novaehollandiae*), Australian Shelduck (*Tadorna tadornoides*), and Australasian Shoveler (*Anas rhynchos*), and Black Swan (*Cygnus atratus*).
3. Explore the relationships between macrophytes and herbivore waterbird guilds (including possible time lags) using seasonal data and monthly data collected between March 2017 and March 2020.
4. Develop a seasonal conceptual model for the wetlands including water regime and ecological components based on findings from this analysis.
5. Use findings from this analysis to identify critical times or conditions for the ecology of the system that may be impacted (either positively or negatively) by manipulation of the water regime (either by increasing fresh water flows off the catchment or increasing seawater inflows through the surge barriers).

## 2 Methods

### 2.1 Study area and period

Biotic and water quality conditions included in this study were measured at varying spatial and temporal resolutions in four regions within the Vasse-Wonnerup wetland between January 2017 (e.g., waterbird count data were available from this date) and March 2020. The regions (Figure 1) included the Lower Wonnerup (LW), Upper Wonnerup (UW), Lower Vasse (LV) and Upper Vasse (UV). Waterbird counts were undertaken at a regional scale (i.e., single counts per region) and were undertaken once a month. All other biota and water quality conditions were sampled at a finer spatial scale. Fish, benthic macro-invertebrates, macrophytes and water quality conditions were sampled at up to five replicate locations within four sites within each of the four regions (Figure 1). The biotic groups were measured two to six times a year, while water quality was measured every month.

Water level was measured at the Vasse and Wonnerup surge barriers every day, and daily area covered by water was estimated for the Vasse and Wonnerup estuaries, inferred from the daily surge barriers water level and provided by DWER. Daily water area included the total area of water covered within each region, each estuary, as well as the area of water corresponding to depths between 0-0.01, >0.01-0.05, >0.05-0.1, >0.1-0.2, >0.2-0.3, >0.3-0.5, and >0.5-1.0 m for each estuary. Daily wind speed and direction were obtained from the weather stations 009603 (Busselton Airport) and 009937 (Busselton Jetty) from the Australian Bureau of Meteorology (BOM).



**Figure 1. Vasse-Wonnerup wetland sample sites and regions in relation to the location of the surge barriers. Source: DWER.**

Waterbird data used in this report were collected by the South West Region, Department of Biodiversity, Conservation and Attractions (DBCA). All other biotic data were collected by Murdoch University (Paice & Chambers 2019, 2020; Tweedley & Cottingham 2019; Tweedley, Cottingham & Beatty 2019a, 2019b, 2020; Tweedley, Cottingham, Strachan & Beatty 2020), while water quality, water level, and area covered by water were collected or estimated by DWER. Details regarding the methods for surveying waterbirds and sampling fish, benthic macro-invertebrates, macrophytes, water quality, water level, and estimating area covered by water are presented in the following reports Paice & Chambers 2019, 2020; Tweedley & Cottingham 2019; Tweedley, Cottingham & Beatty 2019a, 2019b, 2020; Tweedley, Cottingham, Strachan & Beatty 2020. Details regarding the methods for surveying waterbirds and water quality were available from DWER. All data were provided by DWER.

## 2.2 Pre-processing and preparation of data

The datasets provided comprised a range of data, including those outside of the defined study area. Data from the replicate locations within sites within the four regions (i.e., LW, UW, LV, and UV) included in this study were extracted from the original datasets, and all species only observed outside the four regions were excluded.

For the waterbird dataset, the Maned Goose and Australian Wood Duck were combined since they are the same species. Waterbird species that were not considered to depend upon on wetlands and/or wetlands resources were also eliminated from the dataset (e.g., barn owl, little corella), to only include wetland-associated species. For the species included in the dataset, feeding guilds were

developed based on a modification of the classification used by DBCA for the wetland species (Lane *et al.* 2018, pers. comm., K. Williams) with slight modifications to closely reflect foraging ecology of the different waterbirds within the dataset. Waterbirds were grouped in relation to the water depth range they are commonly found in the wetland (information from Lane *et al.* 2018) and feeding behaviour.

- Shoreline: Present in shallow water areas (0 – 5 cm depth) and generalist feeders.
- Small waders: This group compromise migratory and resident waders from different families Scolopacidae, Charadriidae and Recurvirostridae. All feed in shallow water depth areas (0 – 10 cm depth) mostly on benthic invertebrates.
- Large waders: This guild contains herons, egrets, ibises and spoonbills. All of them feed in shallow to medium water depth areas (0 – 20 cm depth) on invertebrates and fishes.
- Deep water and divers: This group covers the waterbird species that need deeper water ranges to feed (20 – 150 cm water depth), like Black Swans and Cormorants. Some species were herbivores while others were piscivores.
- Surface dabblers and filter feeders: This guild group compromise most ducks, feeding on zooplankton (e.g. Australasian Shoveler, a filter feeder), invertebrates or aquatic plants.
- Aerial divers: This guild contains raptor species that localise their prey while flying and species that fly and plunge in water to capture their prey like terns and White-bellied Sea Eagle. They feed mostly on fish.

For waterbird species with sufficient occupancy to include in models (see Section 2.3), movement patterns and known diet composition were also determined, as these influenced occupancy of the wetland and informed model fitting. Waterbirds were classified according to their movement patterns as *migratory*, *nomadic*, *locally nomadic*, or *sedentary* (Hansen *et al.* 2015, Kingsford and Norman, 2002, Kingsford *et al.* 2010 Roshier *et al.* 2002). Migratory included those that have seasonally predictable movement patterns and travel between distant breeding and non-breeding grounds following known paths or flyways used in the outgoing and return journey (Chan 2001, Roshier and Reid 2003). Nomadic species included those that have no fixed breeding grounds and movements occur when conditions are suitable in particular areas and there are no specific movement paths (Chan 2001, Roshier and Reid 2003). Sedentary included waterbirds that remain in the same locality throughout the year (Chan 2001, Roshier and Reid 2003) (Table 2). The known diet composition included the following categories: vegetation, benthic invertebrates, nekton invertebrate, fish and carrion.

For fish, macrophytes, and water quality measures, all records corresponding to dry conditions or that were not sampled because the water level was too low were excluded from the dataset. For the fish and macroinvertebrate datasets, only those with abundance reported as densities were considered in analyses as these provided standardised abundance values. For the fish dataset, most data resulted from seine net sampling. A small number of records consisting of qualitative observations were considered not directly comparable to seine net samples. Such records were removed as their accuracy and precision were unknown and there were only a few (eight observations).

All species were distinguished from each other throughout the study where possible except for those in the family Atherinidae and within the macrophyte genera. The species belonging to the Atherinidae were included in the grouping 'Atherinidae' between 2017 and autumn 2019, and thereafter were identified at family rather than species. For this reason, all analyses presented here (apart from descriptive statistics) included 'Atherinidae' as a grouping of species to allow for the entire dataset over the study period to be analysed together. For macrophytes, *Ruppia* was not identified to species

level at the beginning of the study, thus subsequent species level records for this genus were combined into *Ruppia* to allow all data to be analysed and directly compared with the other macrophytes, of which the data provided were also at genus level. For the large number of invertebrates sampled, many could not be identified at species level, and many were at different stages in their life history. Consequently, macroinvertebrates groups consisted of either the taxonomic grouping, the species, or the species and life history stage.

In addition to the groupings described above, the total macrophyte (both aquatic angiosperms and macroalgae), benthic macroinvertebrates, and fish were also calculated, as were totals for each macroinvertebrate phyla identified in the study area. These groups were included in waterbird models (see Section 2.3 below), as waterbirds are known to feed on a range of species and groupings may be better predictors than specific species. Because waterbirds were sampled at a different temporal (i.e., month) and spatial scale (i.e., Region) than the other biotic groups, the monthly mean, maximum, minimum, and standard deviation (as a measure of variability) per Region for macrophyte, benthic macroinvertebrates, and fish groupings (e.g., Total Macrophytes, Total Invertebrates, Total Fish, Total Molluscs, Total Arthropods, Total Atherinidae, *Ruppia*...) and species were also calculated for use in waterbird models (see Section 2.3 below). Finally, one month lag abiotic conditions and macrophyte values were calculated for inclusion as predictors in models of waterbirds. The total potential variables available for modelling relationships within the ecosystem included several hundred.

Water area available for specific water depth ranges was expected to be associated with waterbirds numbers (De la Cruz *et al.* 2018). To calculate the water area per region available for different water depths, two excel files provided by DWER were used. The first file “Vasse-WonnerupHeight-Area-VolumeRelationships-1.xlsx” presented the water height - area - volume relationships for Vasse Estuary, Wonnerup Estuary and Wonnerup Inlet and it was prepared by Peta Kelsey (DWER) on 20 August 2020. For more information on the Geographical Information System (GIS) methodology used, please access the file or contact DWER. Of the wetland’s sections with independent information on correlations on water height - area - volume, only four were selected to match the regions of the present study. They were as follows: Abba section was considered the Low Vasse region, Sabina section was considered the Upper Vasse region, Wonnerup South section was considered the Low Wonnerup Region and Wonnerup North section was considered the North Wonnerup Region.

The other file used to correlate water depth to available water area per region for different water depth ranges was “Vasse\_Wone\_areas\_volumes\_example.xlsx” provided by Linda Kalnejais (DWER). This file presented the relationship of water height, the two-dimensional total water area and the available two-dimensional water area for different water depth ranges for each estuary separately. However, the information was not provided by region. The water depth ranges were 0 - 0.01 m, 0.01 - 0.05 m, 0.05 - 0.1 m, 0.1 - 0.2 m, 0.2 - 0.3 m, 0.3 - 0.5 m and 0.5 - 1.0 m.

### 2.3 Models for identifying key variables associated with aquatic plants, benthic macroinvertebrate, fish, waterbird abundance including Ramsar-listed species and dominant guild distribution

Generalized Additive Models (GAMs) are some of the most powerful approaches currently available for modelling nonlinear functional relationships between covariates and outcomes (Wood, 2017). GAMs were used to predict macrophyte percent volume index (PVI), benthic macroinvertebrate and fish densities, and counts of waterbird species belonging to the different water guilds as a function of

abiotic and biotic predictor variables. For waterbirds, species-level was used rather than guild-level, as there was variability among species within guilds in their relationships with abiotic and biotic variables. By modelling individual species, the risk of incorrectly generalising patterns of dominant species to all species belonging to the same guild was avoided. An alternative would have been to include guild as an interacting factor, however, such models would have been too complex given the sample size and data attributes and would not have converged. Convergence is when the change in estimates from successive approximations (i.e., iteration to interaction of fitting the model) decreases such that they eventually converge and a solution is reached. Lack of convergence is an indication that the data do not fit the model well as a result of too many poorly fitting observations. As fitting more complex models was not an option, guild-level patterns were identified instead by describing any similar patterns among species within guilds. Models were constructed for each biotic group or species that had sufficient non-zero observations to result in model reliability (i.e., >40 surveys resulting in non-zero observations).

Abiotic and biotic predictors included biologically relevant variables (see Table 1 for list of predictors for each biotic group) for which there were sufficient non-zero observations to allow models to converge. Direct biological relevance was determined using evidence in published peer-reviewed literature and expert knowledge. For water quality predictors, data from samples taken at the surface and bottom of the water column were available, however, data for the bottom were too limited in sample size to include in models. Consequently, for outcome variables in which conditions at the bottom of the water column were considered more relevant, data from the surface were used instead. Biotic predictors included all taxa (with sufficient non-zero observations) from biotic groups that were considered directly relevant to the outcome variable. Where there was insufficient information for the diet for most species being predicted belonging to a biotic group (e.g., fish), all species from potential biotic groups forming their diets or important as habitat were included as predictors. Consequently, only abiotic groups were included as predictors of macrophytes PVI, while macrophytes were included as predictors of benthic macro-invertebrate densities, and macrophytes and benthic macro-invertebrates were included as predictors of fish densities. For waterbirds, there was considerable information available on the diet of the different species, hence only biotic groups known to make up the diet of the species were included in models predicting bird counts (see Table 4 for a list of biotic groups making up waterbird diets). One month lag abiotic conditions and macrophyte values were included as predictors in models of waterbirds.

Full models including indirect effects and interactions were too complex to fit due to the large number of potential predictor variables influencing species of each biotic group. In addition, many candidate predictor variables were collinear. To reduce model complexity only those predictors considered highly likely to be directly relevant to the response variables were included (see Table 1 for likely relevance of predictor variables). While Region was not considered to directly influence response variables (i.e., since the attributes corresponding to each region, themselves, would likely be the drivers), models that were not already too complex were fitted with Region. These included models for all biotic groups except for waterbirds. Models that included region and site as a random variable were too complex to fit, which would have allowed the expected mean values (i.e., the intercept) to vary among Regions and Sites. Ecological year, defined as the 30<sup>th</sup> March to 1<sup>st</sup> April of the following year, was specified as a categorical ('parametric') parameter in waterbird species models. The definition of Ecological Year was based on expert advice regarding the most biologically meaningful (e.g., migration, reproduction) 12-month cycle for most waterbird species occupying the Vasse-Wonnerup wetland (pers. comm., K. Williams). Month was included in models of all biotic groups as a continuous predictor, particularly as seasonality was expected for some species being modelled.

Temporal autocorrelation in residuals of fitted GAMs was evaluated, and as autocorrelation was not a notable feature, autocorrelation structure was not included in what were already complex models.

**Table 1. Biologically relevant predictors considered as potentially directly influencing each of the biotic groups (macrophytes, benthic macro-invertebrates, fish, and waterbirds) that had sufficient observations to include in analyses (e.g., while many water quality variables were measured at the surface and bottom of the water column, sufficient observations were only available for surface measures).**

Predictors	Macrophytes	Benthic macro-invertebrates	Fish	Waterbirds
Month				
Water level (measured at the gates)				
Bottom depth				
Salinity				
Dissolved Organic Nitrogen				
Water temperature				
Filterable reactive phosphorus				
Turbidity				
Total Suspended Solids (TSS)				
Chl a				
Ecological Year				
Wind speed				
Wind direction				
Mean area (in 2D) covered by water and water depth ranges (i.e., 0-0.01, >0.01-0.05, >0.05-0.1, >0.1-0.2, >0.2-0.3, >0.3-0.5, >0.5-1)				
Monthly lag values for biologically relevant environmental variables				
Lag mean area (in 2D) covered by water and water depth ranges				
Macrophyte percentage volume index (total and specific to species)				*
Benthic macroinvertebrate relative abundance (total and specific to species)				*
Fish relative abundance (total and specific to species)				*

\* Macrophyte, macro-invertebrates and/or fish were included in models for waterbird species known to have a diet composed of species within the biotic group.

	May be directly biologically relevant
	Likely directly biologically relevant

While including only those predictors considered highly likely to be directly relevant to the outcome variables substantially reduced the total number of candidate predictors, a large number remained, many of which were collinear. Consequently, a full-subsets modelling approach was taken (Fisher *et al.* 2018). Full-subsets information theoretic approaches (Burnham & Anderson, 2002) provide an alternative to stepwise model selection (Mundry & Nunn, 2009), which can be problematic when a large number of candidate predictors are to be included in models and some of which are collinear. Information theoretic approaches involve the construction of a complete model set from which to compare all models using criterion such as the Akaike information criterion (AIC), AIC corrected for small sample sizes (AICc, Hurvich & Tsai, 1989), and Bayesian information criterion (Wit, van Heuvel, & Romeijn, 2012). Information theoretic approaches typically are used to select a “best”, most “parsimonious”, or “optimal” model of all candidate models. By considering all combinations of variables, the relative importance of different variables can be explored (by summing model weights for each variable; Burnham & Anderson, 2002). The main advantage of this approach is that a full model does not need to be fitted as a starting point for model comparisons, meaning that a greater number of predictors can potentially be explored than might be able to be otherwise (Fisher *et al.* 2018). The approach used here also allowed automatic removal of collinear predictors to avoid unreliable estimates resulting from collinear predictors occurring in the same model.

For waterbird models, automatic removal is particularly relevant for abiotic variables related to the area of the wetland covered by water at different depth ranges, since the areas of water at certain depth ranges is collinear with those at other depth ranges. For example, the area of water covering depths between 0-0.01 m is highly collinear with the area covering depths between >0.01-0.05 m. The full-subsets approach compares models combining each non-collinear set of variables at a time. In this example, models including the variable ‘*area of water at depths between 0-0.01 m*’ would be compared with models including the ‘*area of water at depths between >0.01-0.05 m*’. The most parsimonious of these models that reduces the information theoretic criterion by a predetermined value is then selected as the *optimal* model. Consequently, final optimal models may not necessarily select a depth range that directly reflects the area used by feeding waterbirds (for example), rather it may be linked more directly with a depth range that is more sensitive to changes within the system hydrodynamics (resulting from the shape and bathymetry of the wetland, water regime, precipitation, etc.), which in turn is linked to waterbirds.

The conditional distributions of the response variables were specified by the distribution family; with Tweedie specified in models predicting macrophyte PVI and benthic macroinvertebrate and fish densities, and negative binomial (with a log-link function) specified for waterbird species. The number of basis functions,  $K$ , was constrained to 5 for macrophyte, benthic macro-invertebrate and fish species (and groups), and 4 for waterbird species to allow sufficient variation in smoothers while retaining reasonable computation time. Smooth functions were fitted using isotropic smoothers based on thin plate regression splines with penalties given as 1 (e.g., the order of the derivatives; Wood 2003). The smooth for Wind Direction used a cyclical cubic spline smooth term basis type. Models were fitted using Restricted Maximum Likelihood Estimation (RMLE).

Interaction between factors and continuous variables were considered where possible, but only included interactions with Ecological Year for models predicting waterbirds. All biotic group models included 5 predictor variables at a time, except for those for waterbirds which were limited by sample size. To maximise power of waterbird prediction, models for waterbirds were split into two separate sets: the first included abiotic (environmental conditions) and the second included biotic variables

(fish and invertebrate densities and macrophyte variables). Splitting them allowed for the maximum possible sample size of abiotic predictors to be used, since abiotic predictors were collected at temporal resolution that matched waterbird counts, while biotic predictors were sampled quarterly (i.e., a lower temporal resolution). Models predicting waterbird counts included only two predictors at a time for models with abiotic predictors, and between 4 and 5 were for models with biotic predictors due to limited sample size. The result was a total of 48 optimal models resulting from over 500,000 candidate models.

Before fitting models, data were explored for outliers that would overly influence outputs using histograms and scatter plots. Outliers were removed and the most relevant of collinear predictors retained. The goodness-of-fit of each model was evaluated by confirming that no clear relations between the residuals and fitted values were visible in a scatterplot. In addition, residuals were assessed using QQ plots of the relationship between the deviance residuals (Wood 2017) and their theoretical expectation for the chosen family. Overall fit was also evaluated using a plot of response versus fitted values.

Relationships between response and predictor variables were evaluated based on the summed model weights for each variable and model outputs, including the effective degrees of freedom (EDF), model statistics, p-values and partial residual plots. The summed model weights provided information on the relative importance of predictor variables (on a scale of 0 to 1) compared to other candidate variables in predicting the response variables. Relative importance does not provide information on how large the effect of the variable is whether it is statistically significant. The model outputs provide this information. EDFs provide information about the complexity of a penalized smooth term (i.e., how “wiggly” the term is in its relationship with the response variable). The model statistic and p-value provide information about how important effects were and whether they were significant. For all models that included 4 to 5 terms and included or verged on including all possible non-collinear variables, variables that were significant at  $p < 0.05$  level and were relatively important were identified as those associated with the response variable. For waterbird models including abiotic variables, as a maximum of only two terms could be included, and for many species more than two terms were likely significant, variables that were significant and had a relative importance  $> 0.3$  were identified as having an association with the response variables as this value was consistent with the relative importance of variables identified as significant. ‘Explained variance’ was also reported for all models, which is used to measure the discrepancy between a model and observed data. Explained variance is the part of the model’s total variance that was explained by the explanatory variables and is expressed as a percentage. The remainder of the variance remains unexplained. A high percentage of explained variance suggests a stronger association between response and explanatory variables included in the model.

For more complex models such as those fitted, ‘partial dependence plots’ are used to interpret the relationship between the response variable and predictor variables. Partial dependence plots provide information regarding the relationship between the response variable and predictors after eliminating the effect of the other predictor variables. More specifically, in the example of models with bird counts as the response variable and Month as a predictor (i.e., explanatory variable), partial dependence plots show the change in the mean predicted bird counts on the y axis as the Month changes over its distribution on the x axis. The y axis label that has an ‘s’ in it indicates it is smoothed’ function over months. The means on the y axis are centered and are on the scale of the link function (since the smoothed variables must sum to zero in a GAM). They are accompanied by 95% confidence intervals.

These plots were used to interpret the Model results. The horizontal lines along the bottom of each plot, called a 'rug plot', show the values of the predictor variables for each smooth. Plots for factors show the estimated effect for each factor (i.e., parametric term) level compared to the first of those levels shown as zero.

## 2.4 Software

All analyses and associated visualisations were produced using R v4.0.0 (R Core Team 2018) in RStudio Version 1.1.419 - © 2009-2018 RStudio, Inc. The autocorrelation function was estimated using the *Stats* (R Core Team 2018) package, *MGCV* (Wood, 2020) was used to fit the GAMs called through the package *FSSgam* to apply the full-subsets approach, and LOESS (Locally Estimated Scatterplot Smoothing) curves with 95% CIs were plotted using *ggplot* (Wickham, 2016) in R.

## 3 Results

### 3.1 Conditions influencing waterbirds counted

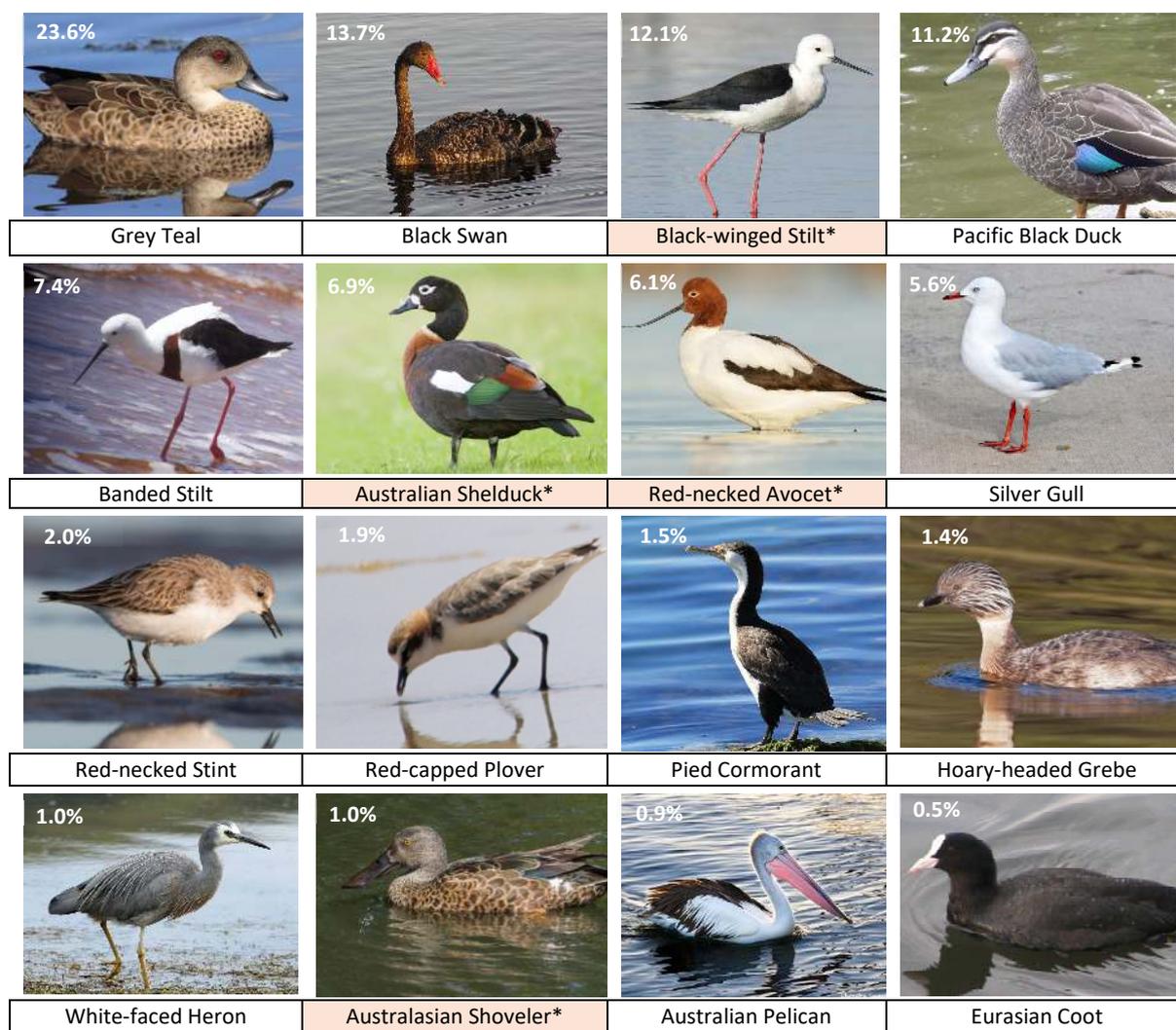
Of a total of 97 bird species detected during surveys between 2017 and 2020 in the Vasse-Wonnerup wetland, 65 were identified as waterbirds occupying (either seasonally or year-round) and accessing resources from the wetland (e.g., feed on macrophytes, invertebrates and/or fish within the system, etc.). These were organised into six waterbird guilds, including the surface dabbler and filter feeder, shoreline, small wader, large wader, deep water and diver and aerial diver waterbird guilds. Surface dabblers and filter feeders in this study were all ducks (Figure 2). Shoreline species were a mixture and included a duck, chat, crake, swamphen, rail and gull. Small waders comprised sandpipers, a stilt, a stint, a curlew, plovers, godwits, and knots, among other species. Large waders were the ibises, herons, egrets and a spoonbill. Deep waterbirds and divers included ducks, cormorants, grebes, a darter, a pelican, and the Black Swan. The aerial divers included terns, an osprey, eagle, kite, kingfisher, and a harrier.

The total number of waterbird counts during surveys was 244,822. The most frequent detections were of the Grey Teal (23.6%), the Black Swan (13.7%), the Black-winged Stilt (a RAMSAR listed species; 12.1%) and the Pacific Black Duck (11.2%), with all other species making up less than 10% of the total counted (see Figure 3, Appendix A. Birds – Figure 19-Figure 26. Mean number of waterbirds ( $\pm$  SE) per year in the Upper Wonnerup between 2017-2020 for species detected less than 15 or fewer times.). The other three RAMSAR listed species, the Australian Shelduck, the Red-necked Avocet, and the Australasian Shoveler, made up 6.9, 6.1, and 1.0% of the total, respectively (Figure 3).

Surface dabblers & filter feeders	Shoreline waterbirds	Small wader	Large wader	Deep water birds & divers	Aerial divers
Grey Teal Duck Chestnut Teal Duck Australasian Shoveler* Pacific Black Duck Australian Shelduck* Pink-eared Duck Freckled Duck Silver Gull	Buff-banded Rail Australian Spotted Crane Purple Swamphen Australian Wood Duck White-fronted Chat	Sharp-tailed Sandpiper Black-winged Stilt* Red Knot Great Knot Eastern Curlew Bar-tailed Godwit Black-tailed Godwit Pacific Golden Plover Red-capped Plover Greater Sand Plover Black-fronted Dotterel Red-necked Stint Australian Pied Curlew Sandpiper Wood Sandpiper Marsh Sandpiper Terek Sandpiper Banded Stilt Red-necked Avocet* Common Sandpiper Common Greenshank	Australian White Ibis Straw-necked Ibis Yellow-billed Spoonbill Glossy Ibis Little Egret White-faced Heron Great Eastern Egret White-necked Heron Hoary-headed Grebe	Black Swan Eurasian Coot Musk Duck Hardhead Duck Blue-billed Duck Australasian Grebe Great Crested Grebe Little Black Cormorant Little Pied Cormorant Australasian Darter Australian Pelican Great Cormorant Pied Cormorant	Eastern Osprey Crested Tern Caspian Tern Sacred Kingfisher Whiskered Tern White-winged Black Tern Whistling Kite White-bellied Sea-Eagle Swamp Harrier

\* = Ramsar-listed species.

Figure 2. Waterbirds detected during the study and their feeding guilds.



**Figure 3. Most often detected species (\* = RAMSAR listed) in the Vasse-Wonnerup wetland during surveys conducted between 2017-2020 across all four sampled regions. Percentage of the total detections are denoted in white text, with the total of species in the figure making up 97% of detections. Source: Red-necked Avocet, Hoary-Headed Grebe, and Australian Shelduck by J. J. Harrison; White-faced Heron by Glen Fergus; Australian Pied Cormorant by Julien Carnot; Black-winged Stilt by Raju Kasambe; Eurasian Coot by R. H. Walpole; and Australasian Shoveler by Francesco Veronesi, all licenced under CC by attribution-non-Commercial; Black Swan, Pacific Black Duck, Silver Gull licenced under CC; Red-necked Stint by Jane Putland; Australian Pelican by Chandra Salgado Kent; and Red-capped Plover and Banded Stilt by Sora Marin Estrella.**

### 3.1.1 Relationship between waterbird counts and environmental conditions

Of the 65 waterbird species, 17 had sufficient non-zero survey counts to attempt modelling of their abundance, including all four Ramsar-listed species (Table 2). Important temporal and environmental conditions for waterbirds included ecological year, month, area of water within different depth ranges available, surface salinity, surface turbidity and wind speed and direction (Figure 4). Important abiotic and biotic variables over the study period in relation to surge barrier gate management are plotted in Figure 28.



**Table 2. Waterbirds species with sufficient non-zero survey counts for modelling. Categories and groups based on their biology, ecology, behaviour and expert advice (Kim Williams, DBCA). Sources of information used to compile breeding, foraging and movement category for each species is in the bibliography section. ■ = Surface dabbler & filter feeder, ■ = shoreline, ■ = small wader, ■ = large wader, ■ = deep water & diver, and ■ = aerial diver waterbird guilds.**

Common name	Scientific name	Feeding guild	Known diet composition	Movement patterns	V-W breeding	Ramsar-listed species
Australasian Shoveler	<i>Anas rhynchos</i>	surface dabblers & filter feeders	benthic invertebrates, nekton invertebrate, vegetation	nomadic		YES
Australian Shelduck	<i>Tadorna tadornoides</i>	surface dabblers & filter feeders	vegetation, benthic invertebrates	nomadic	Yes	YES
Grey Teal Duck	<i>Anas gracilis</i>	surface dabblers & filter feeders	benthic invertebrates, nekton invertebrates, vegetation	nomadic	Yes	No
Pacific Black Duck	<i>Anas superciliosa</i>	surface dabblers & filter feeders	vegetation	locally nomadic	Yes	No
Silver Gull	<i>Chroicocephalus novaehollandiae</i>	shoreline	benthic invertebrates	various		No
Black-winged Stilt	<i>Himantopus himantopus</i>	small wader	benthic invertebrates	nomadic	Yes?	YES
Common Greenshank	<i>Tringa nebularia</i>	small wader	benthic invertebrates, nekton invertebrates	migratory		No
Red-capped Plover	<i>Charadrius ruficapillus</i>	small wader	benthic invertebrates	sedentary	Yes	No
Red-necked Avocet	<i>Recurvirostra novaehollandiae</i>	small wader	benthic invertebrates, nekton invertebrates	nomadic		YES
Australian White Ibis	<i>Threskiornis molucca</i>	large wader	benthic invertebrates	sedentary		No
White-faced Heron	<i>Egretta novaehollandiae</i>	large wader	fish, nekton invertebrates	locally nomadic		No
Australian Pelican	<i>Pelecanus conspicillatus</i>	deep water & divers	fish	nomadic		No
Black Swan	<i>Cygnus atratus</i>	deep water & divers	vegetation	locally nomadic	Yes	No
Little Pied Cormorant	<i>Microcarbo melanoleucos</i>	deep water & divers	fish, nekton invertebrates	locally nomadic		No
Pied Cormorant	<i>Phalacrocorax varius</i>	deep water & divers	fish	sedentary		No
Whistling Kite	<i>Haliastur sphenurus</i>	aerial divers	Carrion, fish	sedentary		No
White-bellied Sea-Eagle	<i>Haliaeetus leucogaster</i>	aerial divers	fish	sedentary		No

The “surface dabbler and filter feeder” waterbird guild was comprised mostly of duck species. Four of the seven detected species during the study (Figure 2) had sufficient non-zero observations to include in models (i.e., the Grey Teal, the Australian Shelduck, the Australasian Shoveler and the Pacific Black Duck). Two of these were Ramsar-listed species, which were the Australian Shelduck and the Australasian Shoveler. The variable that was most commonly identified as important in models across these species was month (Figure 4), with a peak in Grey Teal, Australian Shelduck, and Pacific Black Duck counts associated with December-January, and a trough associated with August-September (Figure 5 and Figure 6). The Australasian Shoveler was the only duck species to have significant differences in expected counts among ecological years, with lower and more variable counts in the ecological year 2017-2018 than in other years. This species was also the only species to have counts significantly associated with salinity and wind speed. Based on the model, a peak in counts can be expected at monthly maximum salinities of around 30‰, with counts dropping above and below ~30‰. Counts can be expected to increase with increasing minimum wind speed, with high uncertainty above approx. 10 km hr<sup>-1</sup> (due to fewer observations in high wind speed conditions). The area covered by water corresponding with particular depth ranges was important in models for all ducks, and most notably the area corresponding to depths between 0.2 and 0.5 m, and 1-month lag values for depths between 0.01 and 0.05 m. Based on the model, Grey Teal and Australian Shelduck counts can be expected to be highest at around 375,000 m<sup>2</sup> monthly mean and 275,000-400,000 m<sup>2</sup> one-month lag monthly minimum estimated area for depths between 0.3 and 0.5 m, respectively. Australian Shelduck counts can be expected to be highest at 350,000 m<sup>2</sup> monthly maximum values for the same water depth range one month prior to counts (e.g., lag value). An increase in Pacific Black Duck counts was associated with increasing monthly maximum area covered with water for depths between 0.2 and 0.3 m and increasing monthly variability (e.g., standard deviation) in area of water for depths between 0 and 0.01 m. In the model, decreasing counts, however, was associated with increasing variability in the monthly maximum area of water for depths between 0.01 and 0.05 m one month prior to counts (e.g., lag value). Neither turbidity nor wind direction featured as important variables in models for any of these species.

The “shoreline” waterbird guild comprised six detected species during the study (e.g., Silver Gull, Buff-banded Rail, Australian Spotted Crake, Purple Swamphen, Australian Wood Duck, and White-fronted Chat; Figure 2). The silver gull was the only of these with sufficient non-zero observations to include in models. Modelled Silver Gull counts were associated with month and wind direction (Figure 4). According to models, a peak in silver gull counts can be expected between January-March and a trough in August-September (Figure 5). In the model, highest counts were associated with monthly minimum wind prevailing from northerly directions, and decreased with shifts towards easterlies, southerlies, and westerlies, respectively. Salinity, turbidity and area of water (regardless of depth range) did not feature as important variables in models predicting silver gull counts.



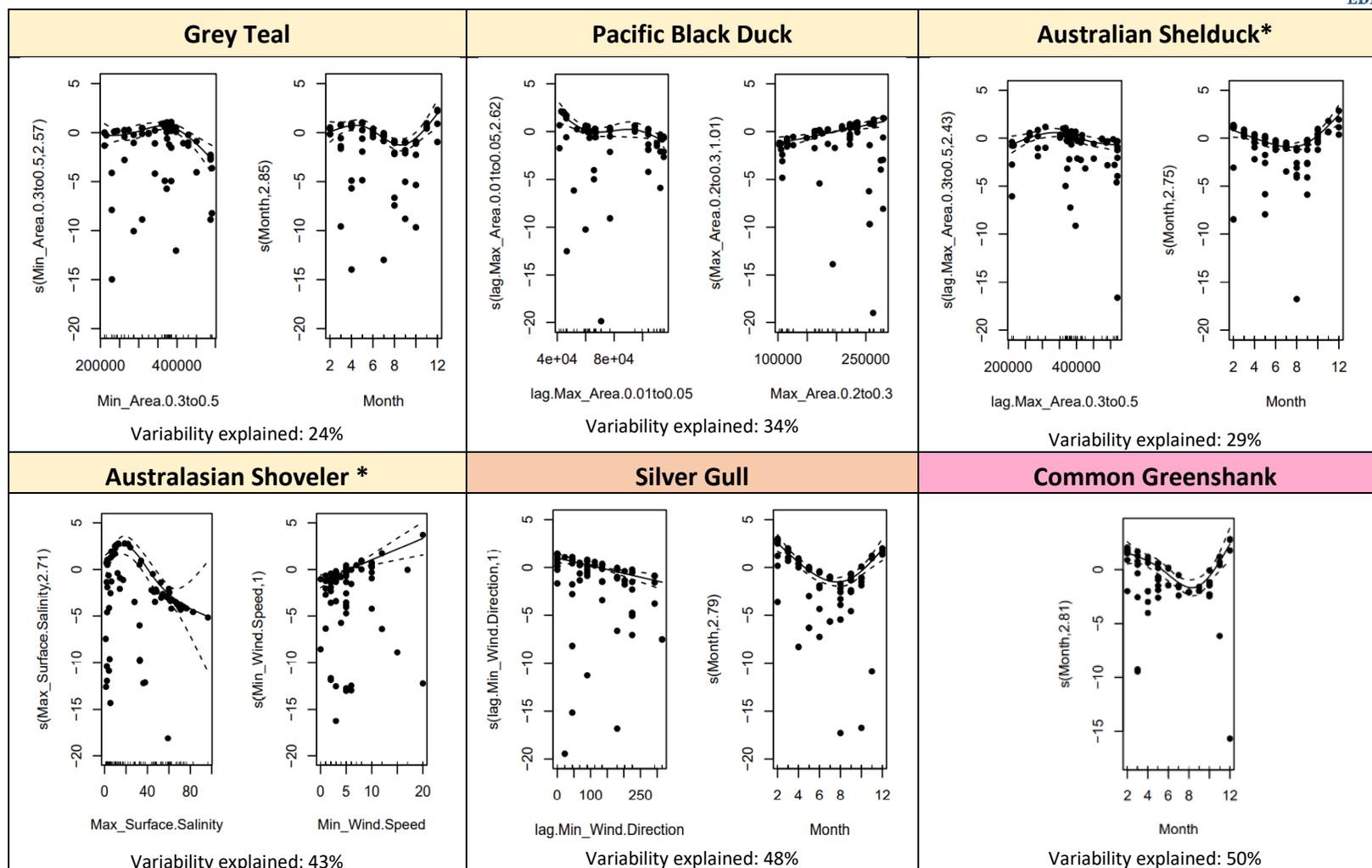


Figure 5. Plots showing the smooth components of the fitted GAM predicting waterbird counts as a function of the two most important significant abiotic predictor variables between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands for smooths illustrating upper and lower lines are added at 2 standard errors above and below the estimate. Confidence bands include the uncertainty about the overall mean. Note: the x axis has a 'rug plot' which indicate values at which surveys were undertaken. ■ = Surface dabbler & filter feeder, ■ = shoreline, ■ = small wader, ■ = large wader, ■ = deep water & diver, and ■ = aerial diver waterbird guilds. \* = Ramsar-listed species.

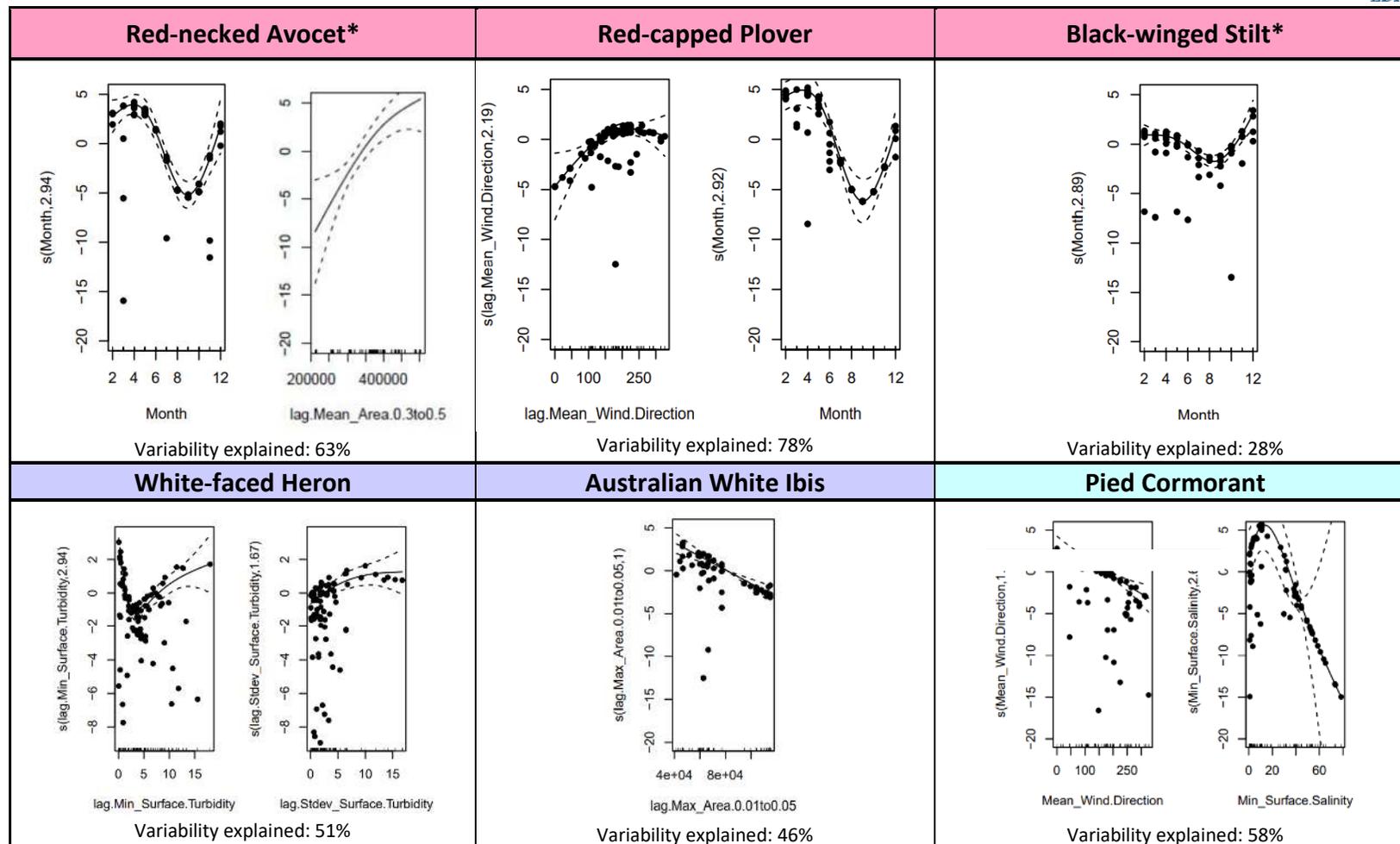


Figure 5 continued...

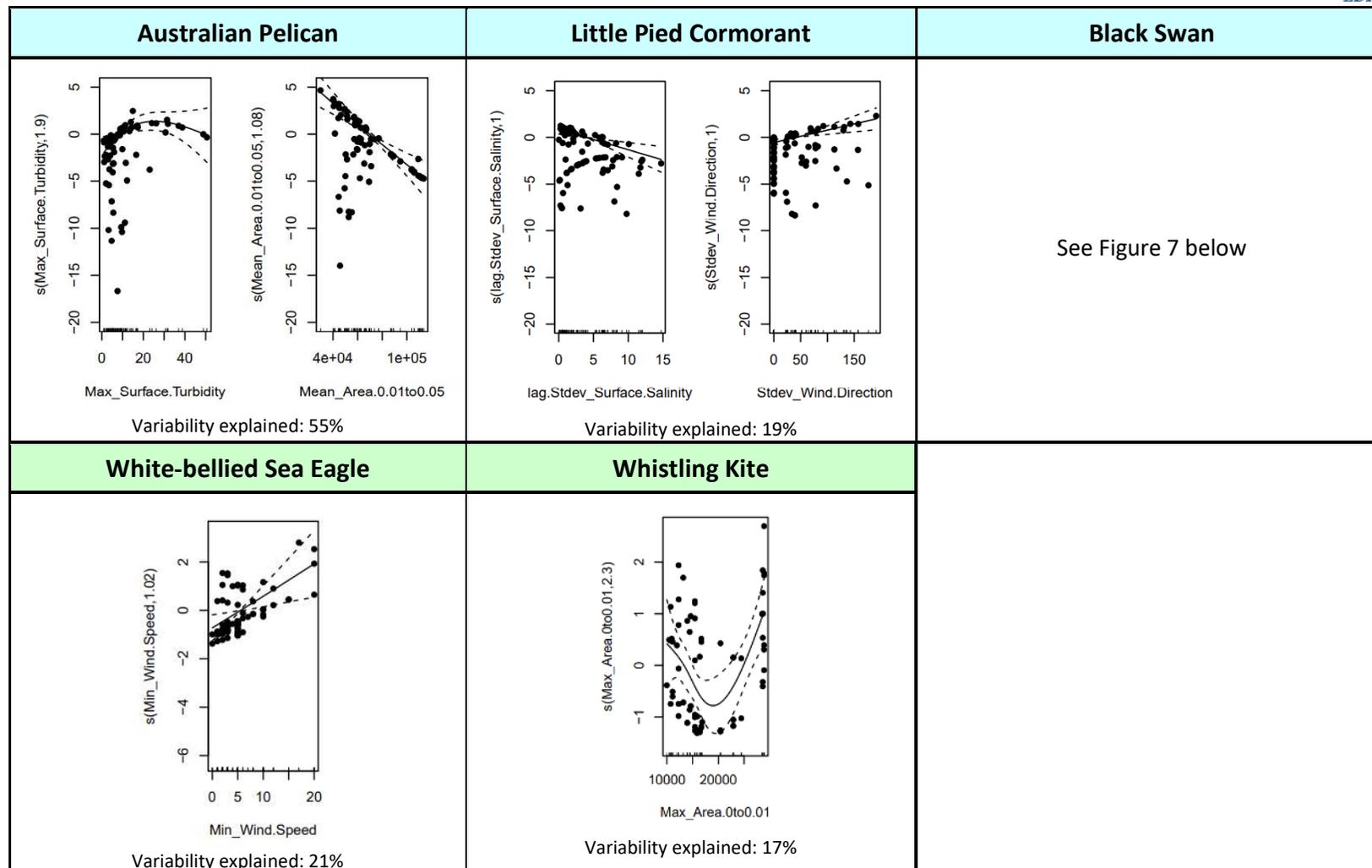
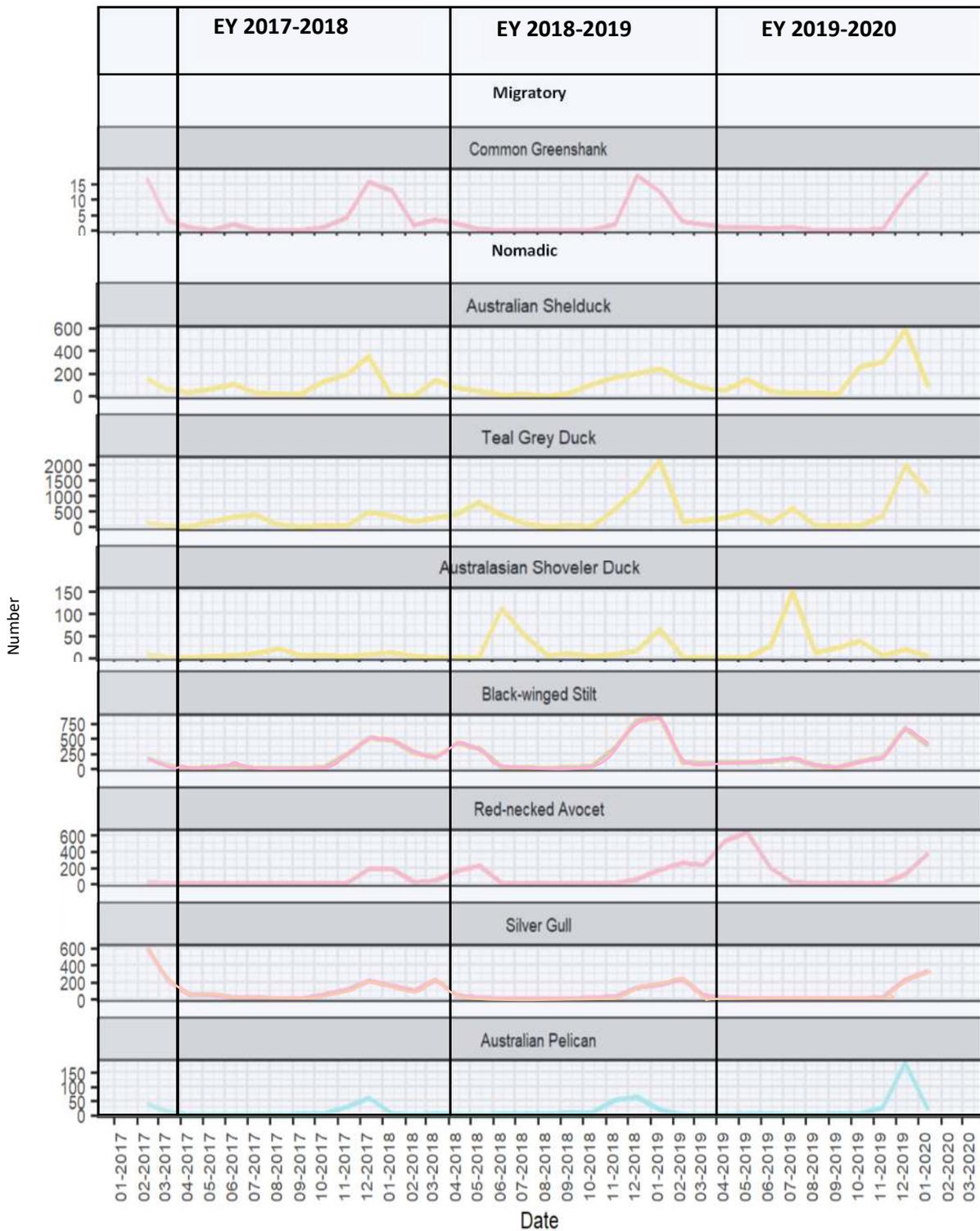


Figure 5 continued...



**Figure 6.** Counts of migratory, nomadic, locally nomadic, and sedentary water species belonging to the six waterbird guilds identified over the study period. Gate management is illustrated in the bottom panel.

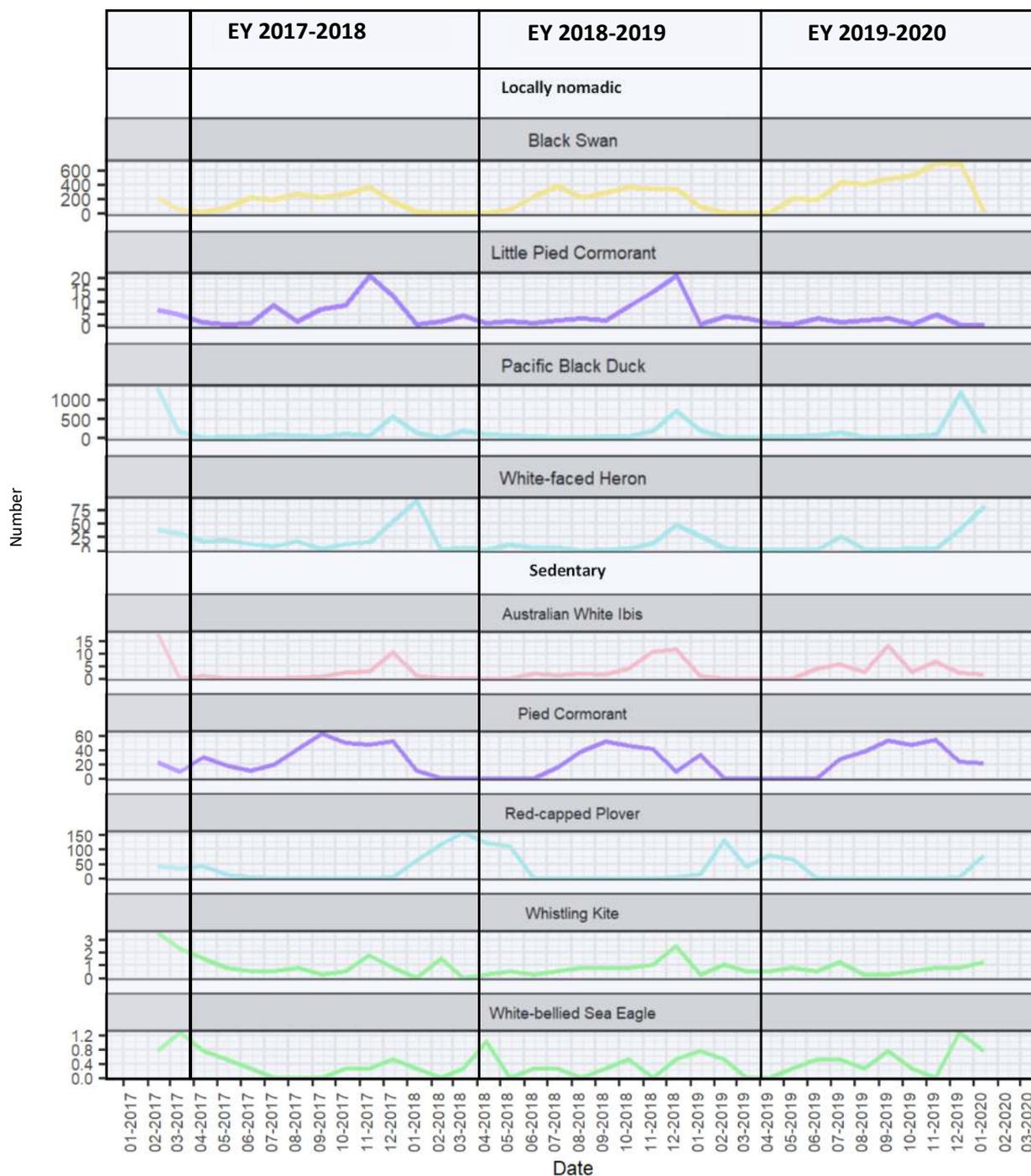


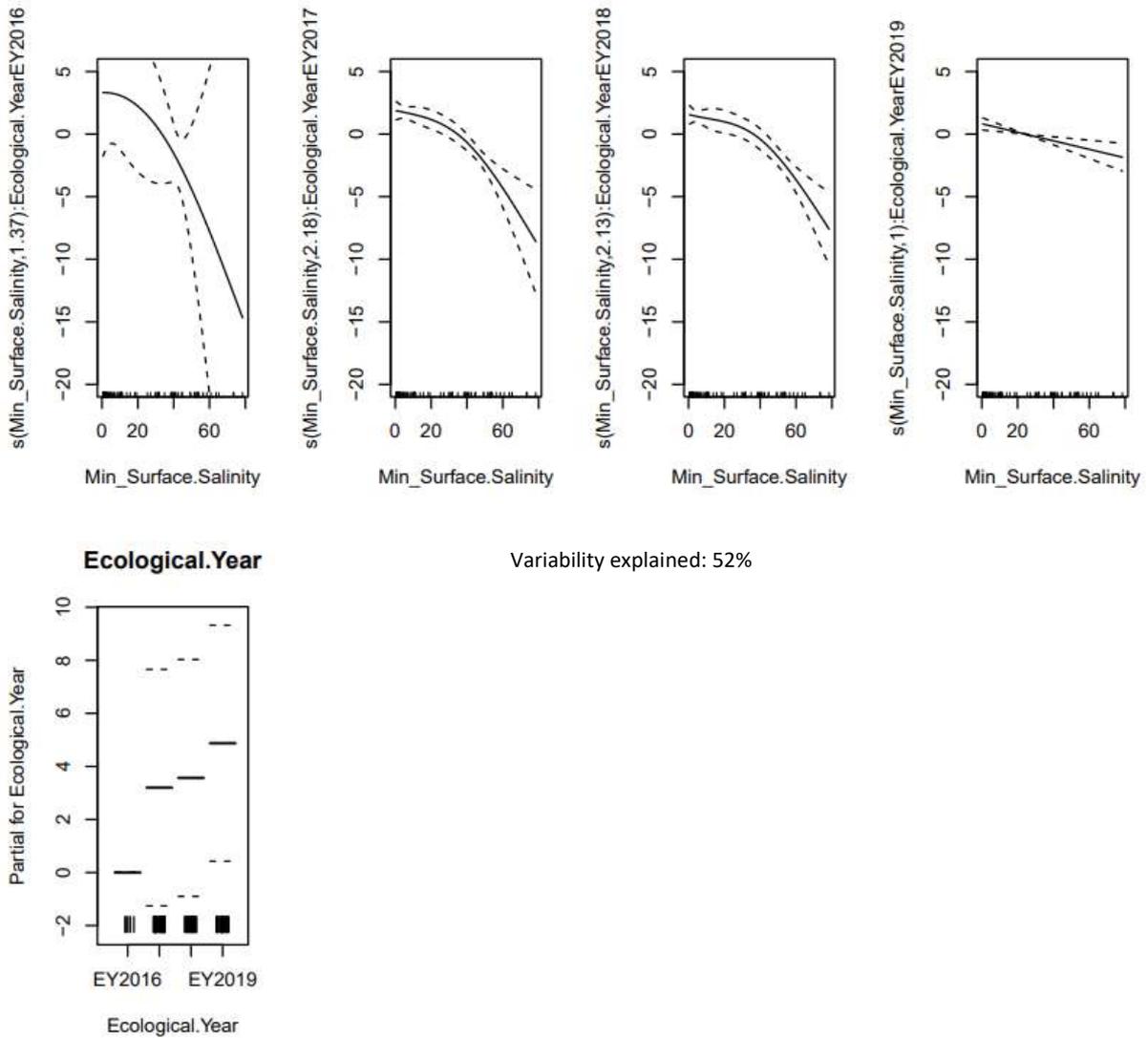
Figure 6 continued...

The “small wader” waterbird guild included 21 species detected (see Figure 2). Of these, the Common Greenshank, Red-necked Avocet, Red-capped Plover, and Black-winged Stilt had sufficient non-zero observations to include in models. The Red-necked Avocet and Black-winged Stilt are the Ramsar-listed species of this waterbird guild. As for the surface dabblers and filter feeders, the variable that was most commonly identified as important across these species was month (Figure 4). Consistent with most surface dabblers, filter feeders and the shoreline Silver Gull, a trough in counts of all species

was associated with September (Figure 5). The peak in Common Greenshank and Black-winged Stilt counts was also similar to most species belonging to the waterbird guilds so far discussed, i.e., December. Based on the model, peaks in Red-necked Avocet and Red-capped Plover counts, however can be expected in March-April, which is later than for previously discussed species. The Red-capped Plover was the only species in which modelled counts were associated with wind direction, with a peak in counts associated with mean and maximum prevailing directional values consistent with the SSW (200 degrees). The area of water available for particular depth ranges was important in models for most small waders. More specifically, highest Common Greenshank counts were associated with monthly standard deviation values of approx. 20,000 m<sup>2</sup> for depths between 0.3 and 0.5 m. According to the model, increasing numbers of Red-necked Avocets are expected with an increased mean water area towards 400,000 m<sup>2</sup> for depths 0.3-0.5 one month prior the counts. A peak in Black-winged Stilt counts can be expected at monthly standard deviation values of approx. 15,000 m<sup>2</sup> area for depths between 0.3 to 0.5 m. A peak in counts was also associated with a 1-month lag in monthly minimum values of approx. 12,000 m<sup>2</sup> for depths between 0.01 to 0.05 m. Salinity and turbidity did not feature in models for these species.

The “large waders” included eight species (see Figure 2) of which the White-faced Heron and the Australian White Ibis had sufficient non-zero observations to include in models. Modelled Australian White Ibis counts varied among years, with particularly low counts in the ecological year 2018-2019 (April 2018 to March 2019 (Figure 4, Figure 5). In contrast to the majority of the surface dabblers and filter feeders and small waders, neither month nor did salinity, wind direction or wind speed did feature as important in predicting counts for either of these species. Based on the models, White-faced Heron counts can be expected to increase with increasing 1-month lag monthly minimum and monthly standard deviation surface turbidity NTU. The area of water available for particular depth ranges was important in models for both species. Australian White Ibis counts varied with lag in monthly mean and maximum area for water between 0 to 0.05 m. The most important of these relationships involved decreasing counts associated with increasing 1-month lag monthly maximum area at ~150,000 m<sup>2</sup> for depths between 0.01 and 0.05 m.

Of the 14 species detected during the study belonging to the “deep water and diver” waterbird guild (see Figure 2), four species including the Australian Pelican, the Black Swan and the Pied and Little Pied Cormorant, had sufficient non-zero observations to include in models. Month featured as particularly important in models for the Black Swan, with a peak in Swan counts associated with the months between July and December (Figure 6). Black Swan counts varied among years, with lowest counts associated with the ecological year corresponding with the April 2016-March 2017. Surface salinity, surface turbidity, and wind direction featured as important predictor variables for counts of this waterbird guild. Based on the models, Black Swan counts can be expected to decrease with increasing monthly minimum salinity, with the steepest drop in numbers at salinities above ~20 ‰. An interaction between year and salinity indicates that the relationship with salinity is expected to have depended upon ecological year, with the last of these years (2019-2020) having a gentler decline in counts with increasing monthly minimum salinity than for previous ecological years (Figure 7). Models suggest that Pied Cormorant counts can also be expected to decrease with increasing salinity, but the pattern is more of a peak at monthly minimum salinities around 13 ‰, with a drop below and above this value (Figure 5). Pelican counts can be expected to increase with increasing monthly maximum surface turbidity up to ~15 NTU, after which there are few observations to allow for certainty in the relationship (Figure 5). Similarly, Swan counts can be expected to increase with increasing monthly maximum turbidity NTU. Based on models, greatest Pied Cormorant counts can be expected to be associated with winds prevailing from the north (Figure 5). Area covered by water was only important for Pelican counts, with decreasing counts associated with increasing monthly mean area for depths between 0.01-0.05 (Figure 5).



**Figure 7.** Plot showing the smooth components of the fitted GAM predicting Black Swan counts as a function of the interaction between ecological year and min surface salinity between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands for smooths illustrating upper and lower lines are added at 2 standard errors above and below the estimate. Confidence bands include the uncertainty about the overall mean. Note: the x axis has a ‘rug plot’ which indicate values at which surveys were undertaken.

The “aerial diver” waterbird guild comprised nine detected species (see Figure 2), two of which had sufficient non-zero observations to include in models. Neither ecological year, nor month, salinity, turbidity, or wind direction featured as important variables in models for these species (Figure 4). Models suggested that increasing White-bellied Sea Eagle counts can be expected with increasing minimum monthly wind speed (Figure 5). Increasing eagle counts may be expected to increase with increasing monthly variability (standard deviation) in area of water up to approx. 25,000 m<sup>2</sup> covering depths between 0.2 and 0.3 m, and then drop at values beyond this. According to models, Whistling Kite counts may be expected to decrease slightly with increasing monthly maximum area up to ~20,000 m<sup>2</sup> for depths up to 0.01 m, and then increase slightly at values beyond this (Figure 5).

See Table 3 for a summary of the key outcomes for the importance of predictor variables for waterbird counts. Optimal models that included two variables (the maximum possible given the sample size) explained 24% of variability in Grey Teal counts, 29% in Australian Shelduck counts, 43% in Australasian Shoveler counts, 34% in Pacific Black Duck counts, 48% in Silver Gull, 50% in Common Greenshank, 63% in Red-necked Avocet counts, 78% in Red-capped Plover counts, 28% in Black-winged Stilt counts, 51% in White-faced Heron counts, 46% in Australian White Ibis counts, 58% in Pied Cormorant counts, 55% in Australian Pelican counts, 19% in Little Pied Cormorant counts, 52% in Black Swan counts, 21% in White-bellied Sea Eagle counts, and 17% in Whistling Kite counts.

### *3.1.2 Relationship between waterbird counts and biota potentially comprising their diet*

Thirteen waterbird species had sufficient non-zero values to include in models with biotic predictors potentially comprising their diet (see Figure 8). Macrophyte PVI was important in models predicting Pacific Black Duck, Black Swan, Australasian Shoveler, and the Grey Teal, with increasing counts associated with increasing PVI for the first two species (Figure 9). Macrophyte PVI metrics identified as important included regional maximum *Lamprothamnium* for the Pacific Black Duck and total regional mean macrophyte PVI for the Black Swan. There was no strong effect apparent resulting from 1-month lag macrophyte PVI values for any species. Counts for Australasian Shoveler and the Grey Teal were fewer in non-zero counts, and results are less reliable, thus no further details are given here.

Arthropods were important in models predicting White-faced Heron counts, as were fish in models predicting Australian Pelican and the White-bellied Sea Eagle counts, with increasing counts associated with increasing densities of these biotic groups. Metrics that resulted in optimal models included the maximum arthropods densities for the White-faced Heron (Figure 9). In contrast, the regional standard deviation of total fish and the Atherinidae densities were the variables resulting in optimal models for the White-bellied Sea Eagle and the Australian Pelican, respectively.

### Species with diet composed of vegetation

	Num. non-zero observations	Mean_Stuckenia	Mean_Athenia	Mean_Lamprothamnium	Mean_Ulva.sp.	Mean_Rhizoclonium	Mean_Cladophora	Mean_Ruppia	Max_Stuckenia	Max_Athenia	Max_Lamprothamnium	Max_Ulva.sp.	Max_Rhizoclonium	Max_Cladophora	Max_Ruppia	Stdev_Stuckenia	Stdev_Athenia	Stdev_Lamprothamnium	Stdev_Ulva.sp.	Stdev_Rhizoclonium	Stdev_Cladophora	Stdev_Ruppia	MacrophytesTotalMean	MacrophytesTotalMax	MacrophytesTotalMin	MacrophytesTotalSE
Pacific black duck	118	0.08	0.09	0.11	0.05	0.07	0.07	0.00	0.00	0.02	0.15	0.07	0.07	0.06	0.00	0.00	0.01	0.14	0.08	0.07	0.07	0.00	0.03	0.01	0.01	0.01
Black swan	124	0.00	0.00	0.00	0.00	0.12	0.09	0.02	0.00	0.00	0.00	0.00	0.16	0.12	0.07	0.00	0.00	0.00	0.00	0.15	0.13	0.06	0.58	0.10	0.05	0.10

### Species with diet composed of macrophytes & invertebrates

	Num. non-zero observations	Mean_Ruppia	Max_Ruppia	Stdev_Ruppia	MacrophytesTotalMean	MacrophytesTotalMax	MacrophytesTotalSE	Mean_Capitella.capitata	Max_Capitella.capitata	Stdev_Capitella.capitata	InvTotalMean	InvTotalSE
Australian shoveler duck*	76	0.01	0.01	0.01	0.74	0.03	0.08	0.04	0.04	0.04	0.14	0.01
Teal grey duck	119	0.09	0.10	0.09	0.07	0.06	0.08	0.06	0.06	0.06	0.15	0.05

### Species with diet composed of invertebrates only and invertebrates and fish

	Num. non-zero observations	Mean_Capitella.capitata	Max_Capitella.capitata	Stdev_Capitella.capitata	InvTotalMean	InvTotalSE	Mean_Annelida	Mean_Arthropoda	Mean_Mollusca	Max_Annelida	Max_Arthropoda	Max_Mollusca	Min_Annelida	Stdev_Annelida	Stdev_Arthropoda	Stdev_Mollusca
White-faced heron	114	0.00	0.10	0.10	0.00	0.00	0.00	0.00	0.10	0.00	0.53	0.10	0.00	0.00	0.00	0.10
Australian white ibis	62	0.00	0.10	0.10	0.00	0.00	0.00	0.10	0.20	0.00	0.00	0.10	0.00	0.00	0.10	0.10
Red capped plover	63	0.00	0.00	0.00	0.10	0.00	0.00	0.10	0.10	0.10	0.10	0.10	0.00	0.10	0.10	0.00
Black-winged stilt*	114	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.00	0.10	0.10	0.10	0.00	0.10	0.10	0.00
Red-necked stint	41	0.00	0.10	0.10	0.10	0.10	0.00	0.10	0.10	0.00	0.10	0.10	0.00	0.10	0.10	0.10

### Species with diet composed of fish

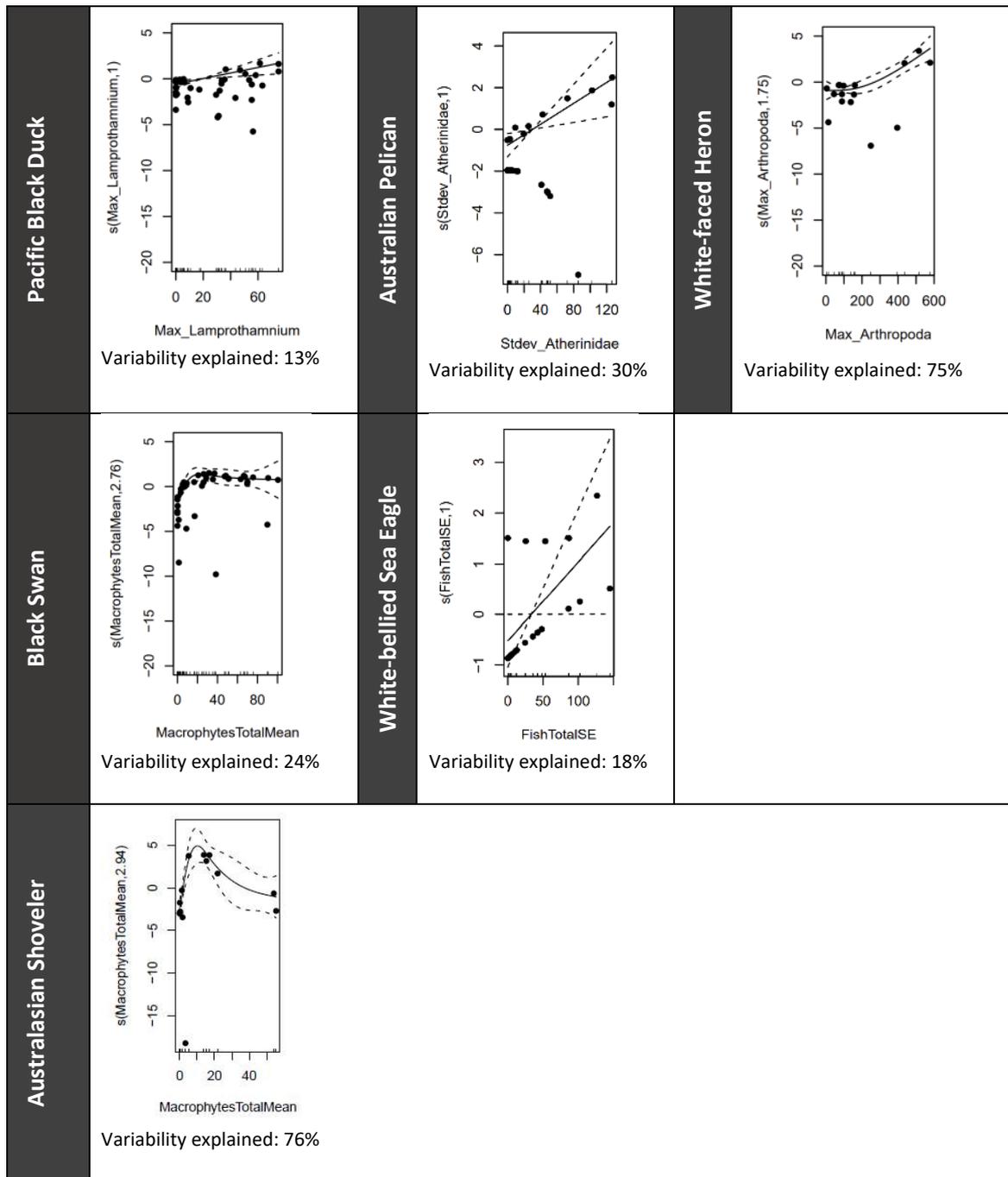
	Num. non-zero observations	Mean_Atherinidae	Max_Atherinidae	Stdev_Atherinidae	FishTotalMean	Total Max Fish	Total Std. Dev. Fish
Australian Pelican	68	0.25	0.17	0.10	0.19	0.17	0.11
Pied Cormorant	61	0.10	0.10	0.10	0.11	0.10	0.10
Whistling Kite	74	0.13	0.11	0.11	0.11	0.10	0.10
White bellied sea eagle	41	0.22	0.18	0.17	0.12	0.13	0.12

**Relative importance scale**

0.0  
0.2  
0.4  
0.6  
0.8  
1.0

Significant at p-value < 0.05 in optimal model

**Figure 8. Relative variable importance scores (0-1) from full subsets GAM analyses exploring the relationship between potential diet composition (Macrophyte PVI, invertebrate density, and fish density) and waterbird counts in the Vasse-Wonnerup wetland during surveys conducted between 2017-2020. All predictor biotic groups included mean, maximum, minimum, and standard deviation values as variables, and those for macrophytes also included their corresponding 1-month lag values undertaken in separate full-subsets models and not presented in these tables). Note: All models included a maximum of five predictor variables except for the model with invertebrates only and invertebrates and fish diet composition which had four. \* = Ramsar-listed species. ■ = Surface dabbler & filter feeder, ■ = shoreline, ■ = small wader, ■ = large wader, ■ = deep water & diver, and ■ = aerial diver waterbird guilds. \* = Ramsar-listed species.**



**Figure 9.** Plot showing the smooth components of the fitted GAM predicting waterbirds as a function of diet components between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands illustrate the uncertainty about the overall mean ( $\pm 2$  standard errors). A ‘rug plot’ on the x axis illustrates samples taken.

See Table 3 for a summary of the key outcomes for the importance of predictor variables for waterbird counts. Optimal models for biotic parameters explained 13% of variability in Pacific Black Duck counts, 24% in Black Swan counts, 76% in Australasian Shoveler counts, 75% in White-faced Heron counts, 30% in Australian Pelican counts, and 18% in White-bellied Sea Eagle counts.

**Table 3. Summary of the relationship between the most important predictor variables and waterbird counts based on GAMs.**

Movement	Feeding Guild	Common name	Ramsar listed species	Food	Season	Month	Water area	Wind conditions	Salinity	Turbidity	Ecological Year	
nomadic	surface dabblers & filter feeders	Australian Shelduck	YES		Dry							
					Wetting							
					Wet	It presents a trough in counts around September						
					Drying	Maximum numbers in December	Peak associated with maximum available water area for depths 0.3-0.5 m one month prior the counts					
nomadic	surface dabblers & filter feeders	Grey Teal Duck	No	Total invertebrates (mean)	Dry							
					Wetting							
					Wet							
					Drying	It presents a trough in counts around September and maximum numbers in December	Peak in numbers associated with available water area for depths 0.3-0.5 m					
nomadic	surface dabblers & filter feeders	Australasia n Shoveler	YES	Total macrophytes (mean), total invertebrates (mean)	Dry			Counts can be expected to increase with increasing minimum wind speed	Peak in counts at salinities of around 30‰		Lower and more variable counts in the ecological year 2017-2018	
					Wetting							
					Wet							
					Drying							
locally nomadic	surface dabblers & filter feeders	Pacific Black Duck	No	Laprotanium (maximum )	Dry	Shows maximums in December. Weak seasonality	Peak in numbers associated with available water area for depths 0.2-0.3 m. Decreasing counts associated with increasing monthly maximum area of water for depths between 0.01 and 0.05 m one month prior to counts					
					Wetting							
					Wet							
					Drying							
various	shoreline	Silver Gull	No		Dry	Maximum numbers in January-February		Numbers were higher with North winds happening one month prior the counts				
					Wetting							
					Wet	Trough in counts around September						
					Drying							

Table 3 continued...

Movement	Feeding Guild	Common name	Ramsar listed species	Food	Season	Month	Water area	Wind conditions	Salinity	Turbidity	Ecological Year
nomadic	small wader	Black-winged Stilt	YES		Dry	Maximum numbers in December.	A peak in counts was also associated with the existence of a maximum water area for depth a month prior the counts . This is correlated with posterior moderate increment a month later (December) of mean water area for shallow water depths from 0.01 to 0.1 m				
					Wetting						
					Wet	Trough in counts around September					
					Drying						
migratory	small wader	Common Greenshank	No		Dry	Maximum numbers in December					
					Wetting						
					Wet						
					Drying						
sedentary	small wader	Red-capped Plover	No		Dry	Substantially high numbers in March-April related with breeding period		Numbers were higher with North winds happening one month prior the counts.			
					Wetting						
					Wet	Trough in counts around September					
					Drying						
nomadic	small wader	Red-necked Avocet	YES		Dry	High numbers in March-April	Increasing mean water area for depths 0.3-0.5m one month prior the counts correlated with posterior increments a month later of mean water area for shallow water depths from 0.01 to 0.1 m				
					Wetting						
					Wet	Trough in counts around September					
					Drying						

Table 3 continued...

Movement	Feeding Guild	Common name	Ramsar listed species	Food	Season	Month	Water area	Wind conditions	Salinity	Turbidity	Ecological Year	
sedentary	large wader	Australian White Ibis	No	Mollusca (mean)	Dry					Counts can be expected to increase with increasing turbidity one month prior count and with variability of turbidity on the month the count was done	Low counts in ecological year 2018-2019	
					Wetting							
					Wet							
					Drying							
locally nomadic	large wader	White-faced Heron	No	Arthropoda (mean)	Dry		Numbers decreased with increasing of really shallow areas one month prior to count					
					Wetting							
					Wet							
					Drying							
nomadic*	deep water & divers	Australian Pelican	No	Atherinidae (StDev)	Dry	Shows maximums in December. Weak seasonality	Numbers decreased with drying of the wetland			Numbers increased within a low turbidity range		
					Wetting							
					Wet							
					Drying							
sedentary	deep water & divers	Pied Cormorant	No		Dry			Numbers increased with N winds	Peak numbers at 13‰.			
					Wetting							
					Wet							
					Drying							
locally nomadic	deep water & divers	Little Pied Cormorant	No		Dry			Numbers increased with variability of S winds				
					Wetting							
					Wet							
					Drying							
locally nomadic	deep water & divers	Black Swan	No	Total macrophytes (mean)	Dry							
					Wetting							
					Wet							High numbers related with breeding period
					Drying							High numbers in December

Table 3 continued...

Movement	Feeding Guild	Common name	Ramsar listed species	Food	Season	Month	Water area	Wind conditions	Salinity	Turbidity	Ecological Year
sedentary	aerial divers	Whistling Kite	No		Dry		Counts decrease slightly with increasing area up to approx. 20,000 m <sup>2</sup> for depths between 0 to 0.01 up to, and then increase slightly at values beyond this				
					Wetting						
					Wet						
					Drying						
sedentary	aerial divers	White-bellied Sea Eagle	No	Total fish (StDev)	Dry			Numbers increased with wind speed			
					Wetting						
					Wet						
					Drying						

### 3.2 Conditions influencing relative abundance of fish species

Sampling of fish between 2017 and 2020 in the Vasse-Wonnerup wetland yielded 9 species (see Appendix C. Fish - Table 6) and an estimated total number of 43,595 fish. The most abundant taxa, when adjusted for sample sizes, were those of the family Atherinidae (82.4%), *L. wallacei* (12.9%), and the Gobiidae *P. olorum* (4.5%) (see Appendix C. Fish - Figure 30-30 and Table 6).



Figure 10. Most often sampled taxa of fish caught in the Vasse-Wonnerup wetland between 2017-2020 across all four sampled regions (see Table 6). Percentage of the grand total are denoted in white text, with the total of species in the figure making up 99.8% of fish sampled. Note: *A. elongata*, *L. wallacei*, *A. mugiloides* and *A. vaigiensis* were pooled into the family Atherinidae for all analyses due to fish in this family being identified only to family level between Jul 2017 and Jul 2019. Source: Erik Schlogl / iNaturalist.org. License: CC by Attribution-NonCommercial/Rudie H. Kuitert

Of the fish taxa collected, only Atherinidae had sufficient non-zero survey counts for robust modelling. The total numbers of fish across all taxa were also included in models. Of the variables used in the models, month and invertebrate abundance were found to be relatively important in models of the total number of fish and Atherinidae (see Figure 11). Due to the dominance of Atherinidae in the fish samples, the models for the total number of fish and Atherinidae demonstrated similar relationships with month (see Figure 11, Figure 12). A peak in densities was associated with summer months with a drop during winter months. An increase in total fish and in the family Atherinidae was associated with an increase in invertebrate densities up to a relative abundance of ~1,000 invertebrates per 0.1 m<sup>2</sup>. Final optimal models explained 60% of the density of Atherinidae and 54% of Total Fish.

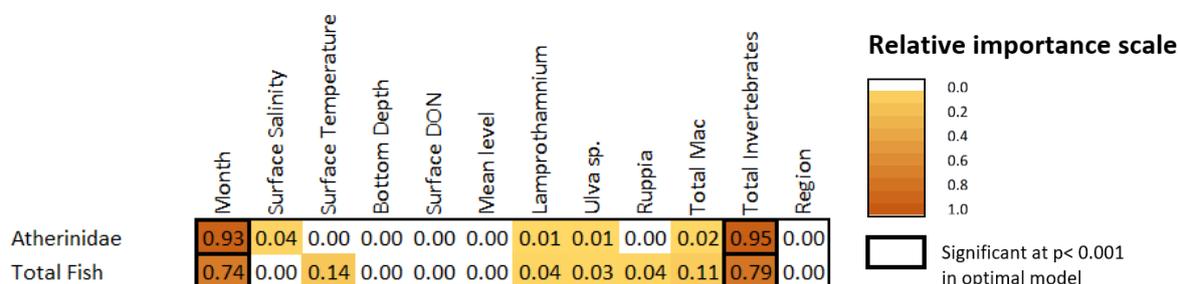
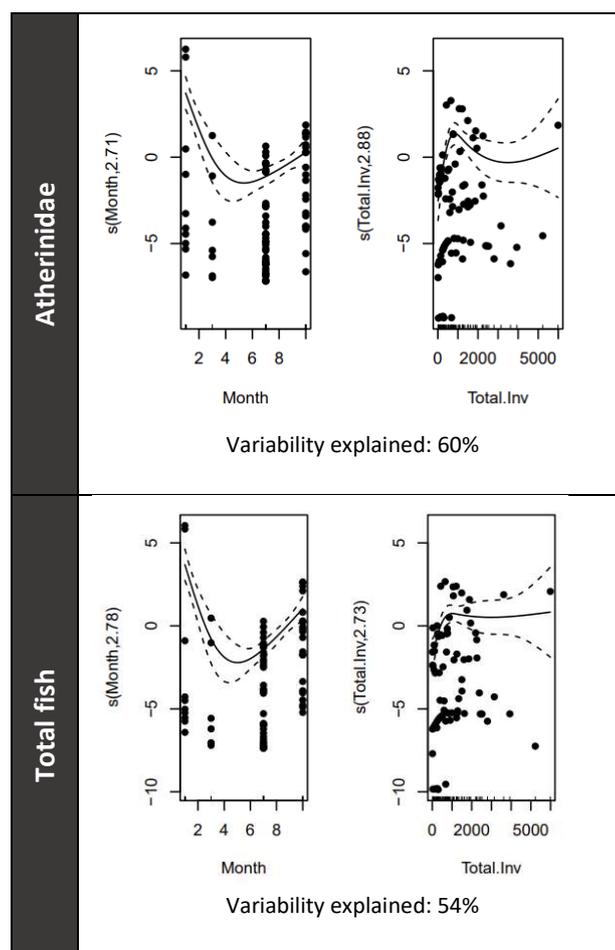


Figure 11. Relative variable importance scores (0-1) from full subsets GAM analyses exploring the relationship between environmental conditions and abundance per 100 m<sup>2</sup> for fish in the Vasse-Wonnerup wetland during surveys conducted between 2017-2020. Note: All models included a maximum of five predictor variables and only total fish and Atherinidae had sufficient non-zero observations to be included in models.



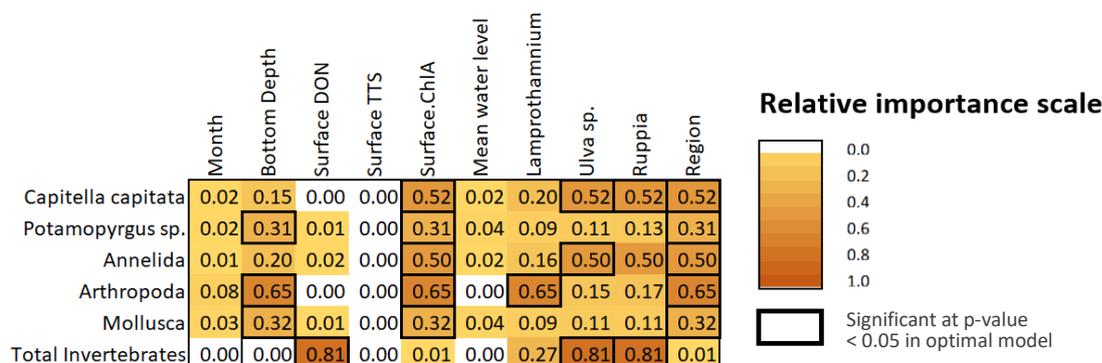
**Figure 12.** Plot showing the smooth components of the fitted GAM predicting total number of fish per 100 m<sup>2</sup> as a function of environmental variables and biotic groups with sufficient non-zero values between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands illustrate the uncertainty about the overall mean ( $\pm 2$  standard errors). A ‘rug plot’ on the x axis illustrates samples taken.

### 3.3 Conditions influencing relative abundance of benthic macro-invertebrate species

Of the benthic macro-invertebrates sampled between 2017 and 2020 throughout the Vasse-Wonnerup wetland, 94 taxa/life stages were identified (see Appendix D. Invertebrates -Table 7). The total number of benthic macro-invertebrates was 239,924 individuals after abundances per sample had been standardised to number of individuals per 0.1 m<sup>2</sup>. The species most frequently identified was the polychaete *Capitella capitata* (67.6%), followed by the gastropod *Potamopyrgus* sp. (10.4%), the chironomid *Procladius* sp. (4.6%) and the ostracod *Mytilocypris tasmanica chapmani* (4.3%), with all other species comprising less than 3% of the total benthic macro-invertebrates sampled (Appendix C. Fish Invertebrates).

Of the 94 groups, only two had sufficient non-zero survey counts to model their abundances against possible influencing variables. Pooling of taxa/life stages into broader taxonomic levels allowed abundances of Annelida, Arthropoda, and Mollusca to be modelled against those variables. Region, Surface Chl A, Bottom depth, and cover of the macrophytes *Ulva* and *Ruppia* were often important

variables, while surface DON was important for predicting the total abundances of invertebrates (see Figure 13).



**Figure 13. Relative variable importance scores (0-1) from full subsets GAM analyses exploring the relationship between environmental conditions and abundance 0.1 m<sup>2</sup> for benthic macro-invertebrates in the Vasse-Wonnerup wetland during surveys conducted between 2017-2020. Note: All models included a maximum of five predictor variables, and species, taxonomic group and total invertebrates with sufficient non-zero observations were included in models.**

While surface Chlorophyll *a* (Chl A) concentration and Region were generally important predictor variables for all modelled taxonomic groups, the importance of other predictor variables differed across those groups (Figure 10). The percent volume index (PVI) of the macrophytes *Ruppia* and *Ulva* were also important predictor variables for the polychaete *Capitella capitata* and Annelida more broadly. There was a general negative response of these worms with increasing Chlorophyll *a* and macrophyte PVI (see Figure 14). Greatest densities for Annelida and *C. capitata* were associated with the lower region of both Vasse and Wonnerup, and the Upper region of the Vasse.

In addition to Chlorophyll *a* and Region, Bottom depth was an important predictor variable for the gastropod and Mollusca more broadly (Figure 10). High densities of Mollusca and *Potamopyrgus sp.* were associated with the Upper Vasse and in the Lower Wannerrup. Mollusca and *Potamopyrgus sp.* displayed a uni-model relationship with Bottom depth and Chl A concentration, increasing to mid-levels before declining with greater depth and concentrations, respectively, and then decreased (although the decrease was associated with greater uncertainty due to fewer observations at greater depths and Chl A concentrations).

Arthropoda showed a similar uni-model relationship with Bottom depth to molluscs, while greatest densities were associated with the Upper Vasse and Upper Wonnerup (Figure 11). More broadly, a decrease in Total Invertebrate densities was associated with increasing PVI of *Ruppia* and *Ulva*, and dissolved organic nitrogen (DON) concentration.

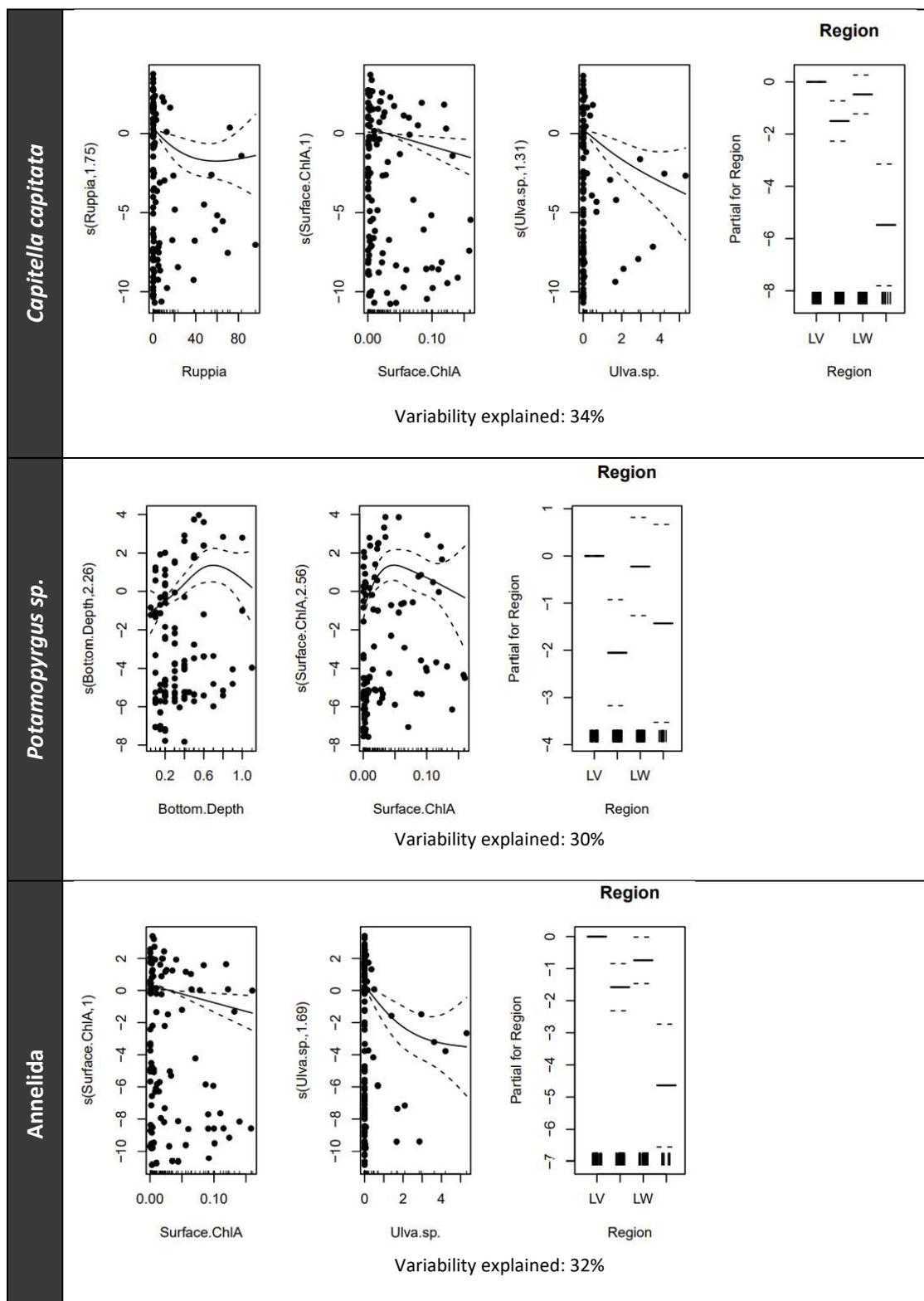


Figure 14. Plot showing the smooth components of the fitted GAM predicting total number of benthic macro-invertebrates per 0.1 m<sup>2</sup> as a function of environmental variables, biotic groups with sufficient non-zero values and region (Lower Vasse, Upper Vasse, Lower Wonnerup, Upper Wonnerup) between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands illustrate the uncertainty about the overall mean ( $\pm 2$  standard errors). A 'rug plot' on the x axis illustrates samples taken.

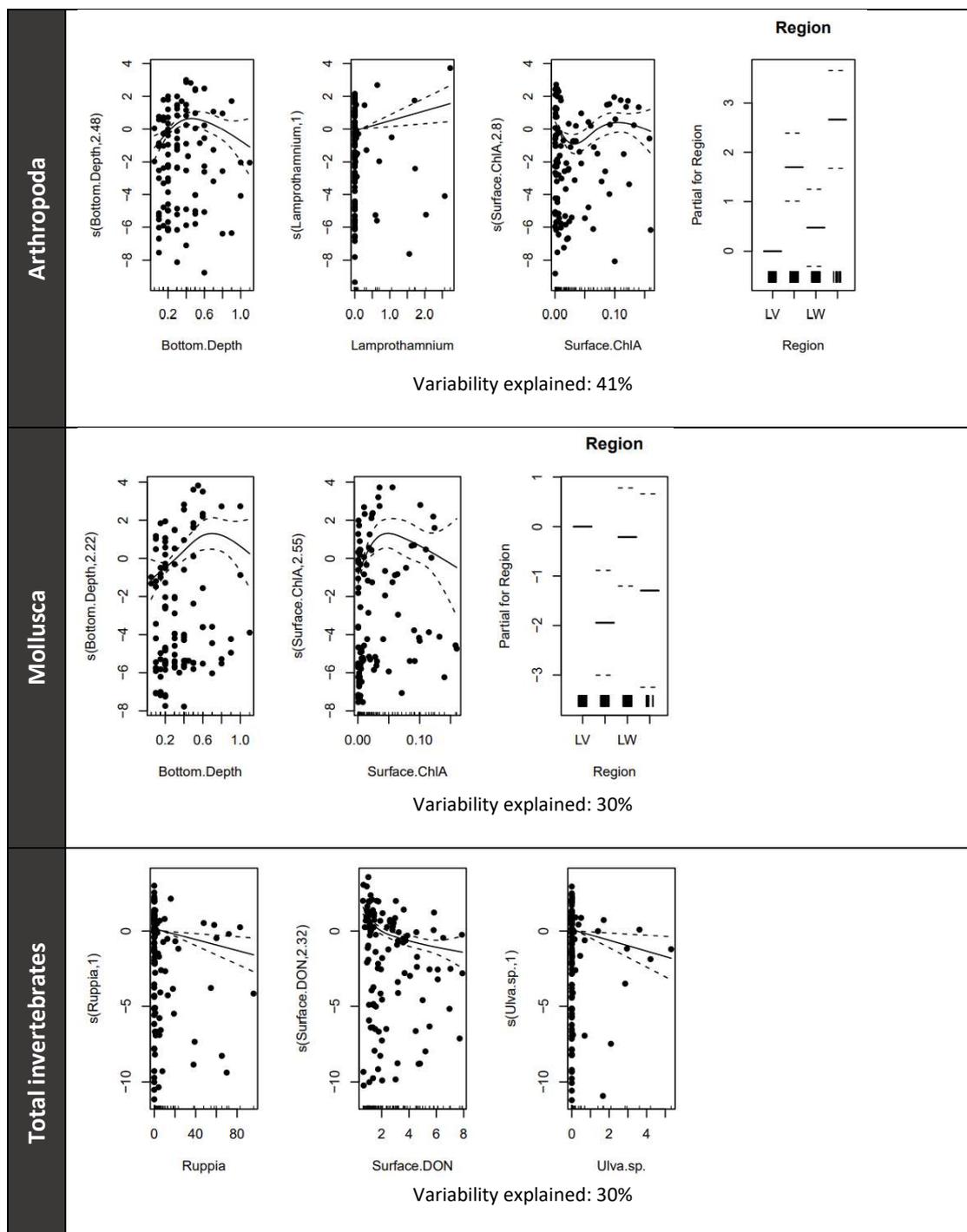


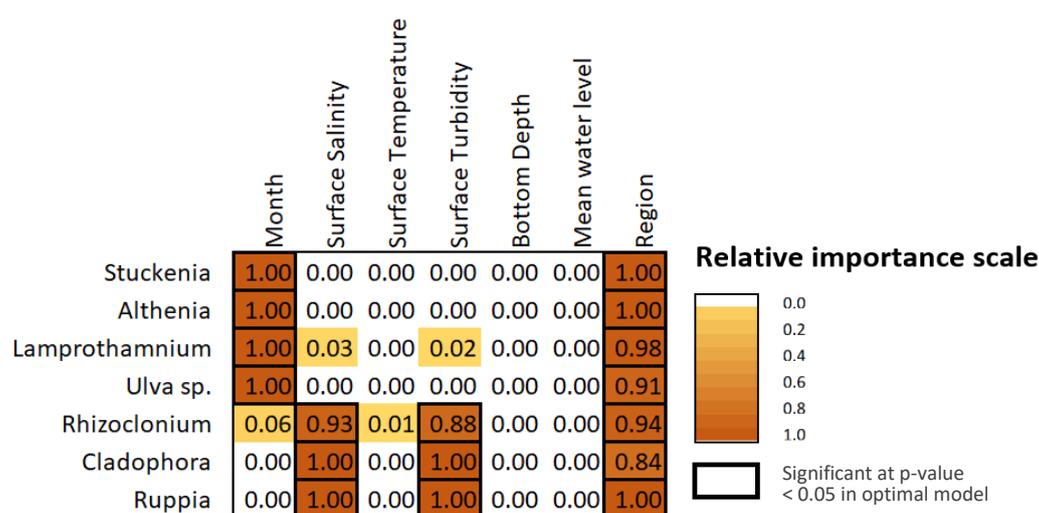
Figure 14 continued...

Final optimal models explained 34% of the density of *Capitella capitata*, 32% of Annelida, 41% of Arthropoda, 30% of Mollusca, 30% of Total Invertebrates.

### 3.4 Conditions influencing macrophytes

Nine taxa of macrophytes were recorded between 2017 and 2020 at sites in the Vasse-Wonnerup wetland, which included a combination of species and genus levels (see Appendix E. Macrophytes-Table 8). The seagrass *Ruppia* had the highest percent volume index (PVI) (24.6% of the total), followed by *Lamprothamnium* (17.9%), *Stuckenia* (9.7%), *Ulva* (9.6%), *Cladophora*. (7.3%), *Rhizoclonium* (3.9%), and *Althenia* (3.8%).

Since there was inconsistency in species level being recorded within genus and across genera, analyses have focused on genus level, resulting in six genera being included in models. Of directly relevant variables with sufficient observations to include in the models, region was consistently an important predictor variable for the PVI of all genera, while month was important for *Stuckenia*, *Althenia*, *Lamprothamnium* and *Ulva* and surface salinity and turbidity were important for *Rhizoclonium*, *Cladophora* and *Ruppia* (Figure 12).



**Figure 15. Relative variable importance scores (0-1) from full subsets GAM analyses exploring the relationship between environmental conditions and percent volume index (PVI) for macrophyte species in the Vasse-Wonnerup wetland during surveys conducted between 2017-2020. Note: All models included a maximum of five predictor variables.**

Macrophytes varied among regions with *Ruppia* having the highest PVI in the Lower Wonnerup and lowest in the Upper Vasse (Figure 15). *Lamprothamnium* and *Althenia* had the highest PVI in the Lower Vasse and the Lower and Upper Wonnerup, and the lowest in the Upper Vasse, while *Stuckenia*, *Rhizoclonium*, *Ulva*, and *Cladophora* had the highest PVI in the Lower and Upper Vasse and the Lower Wonnerup and lowest in the Upper Wonnerup (Figure 16). For those genera where month was an important predictor variable, the relationships varied across genera. *Stuckenia* and *Althenia* PVI increased in austral summer months and decreased in the winter, while *Ulva* PVI increased through the winter months, peaking in the late spring/early summer and dropping to a low in the late summer. Similar variability occurred across genera for which surface salinity and turbidity were shown to be important (Figure 15). *Ruppia* PVI decreased with increasing salinity, particularly at concentrations above ~40-50‰ *Rhizoclonium* and *Cladophora* increased with increasing salinity until 60‰, after which there were too few observations for reliable inference. The PVI of *Ruppia* and *Cladophora* decreased with increasing turbidity, with the sharpest drop between 0 and 15 NTU, while the PVI for *Rhizoclonium* responded negatively to increasing turbidity until about 15 NTU after which it responded positively to increasing turbidity. Final optimal models explained 50% of the variability in PVI for

*Stuckenia*, 66% for *Althenia*, 46% for *Ulva*, 27% for *Rhizoclonium*, 29% for *Cladophora*, and 38% for *Ruppia*.

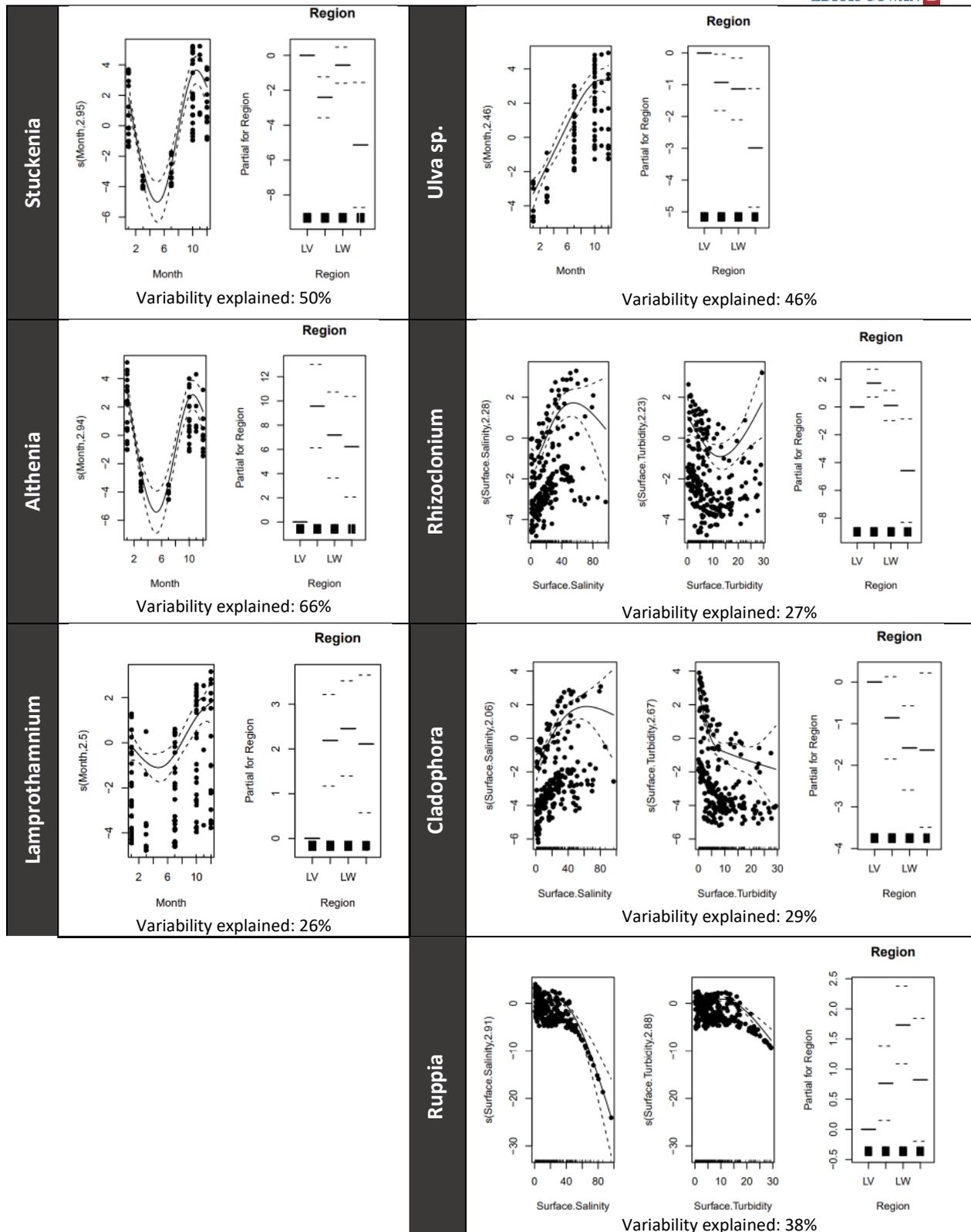
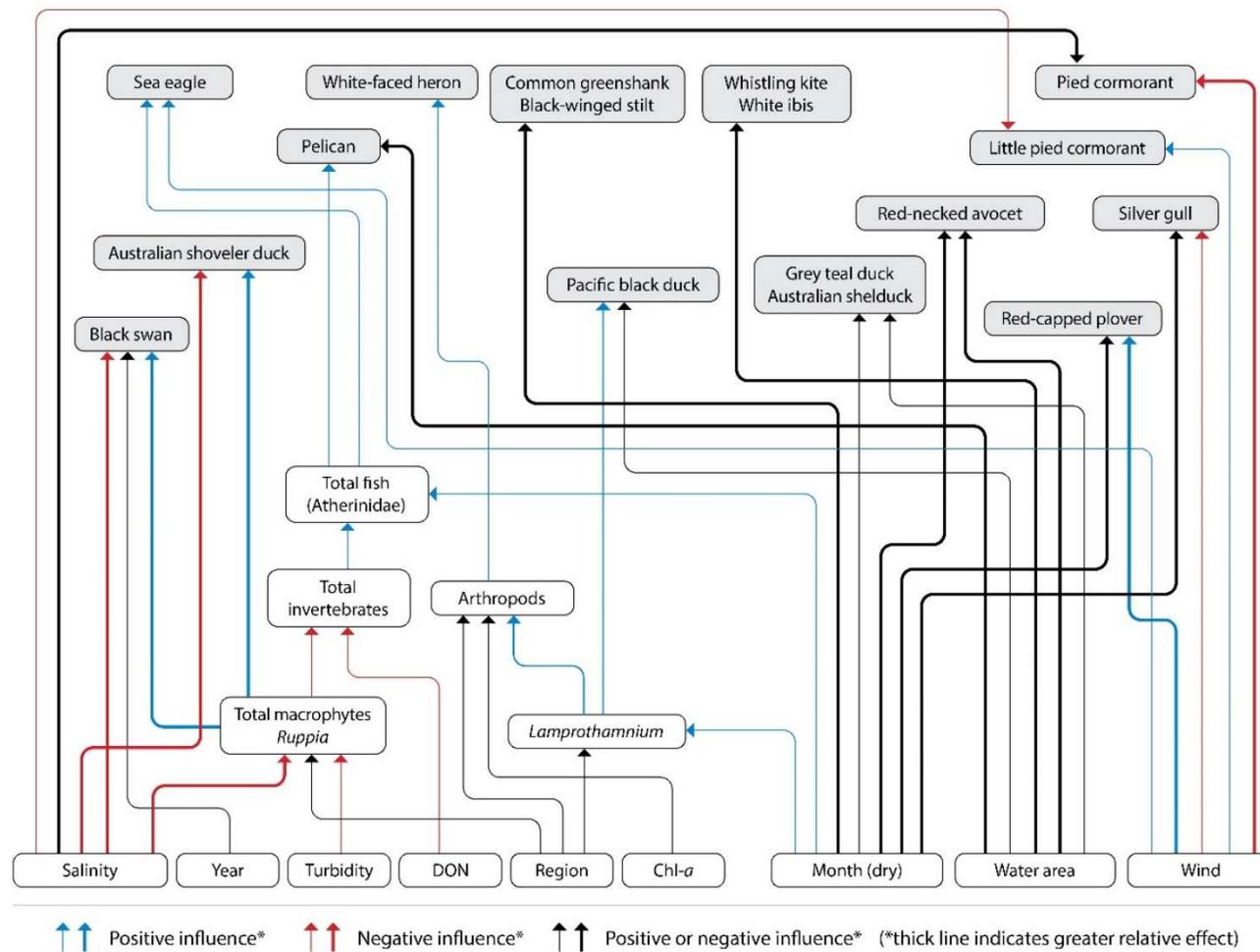


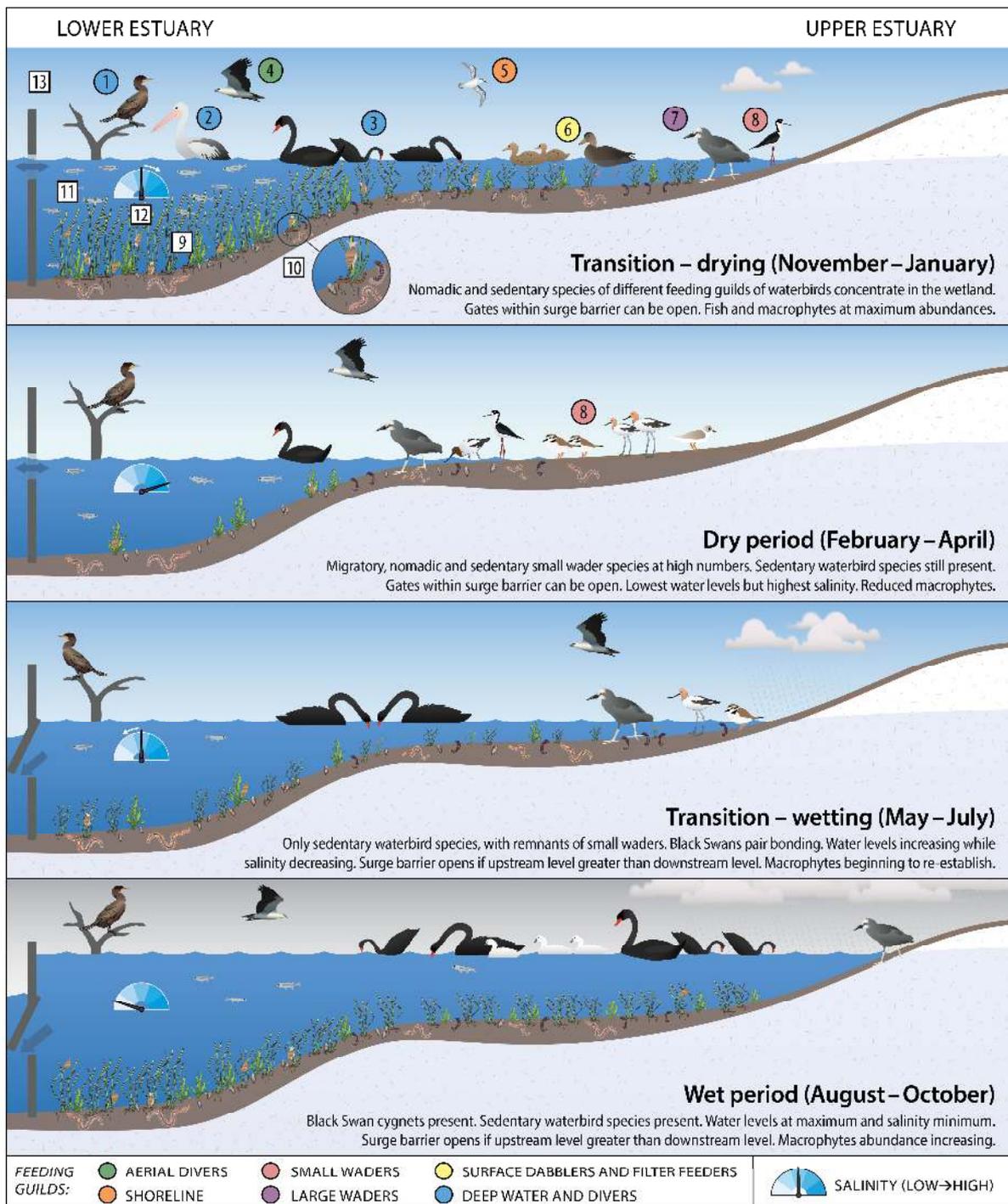
Figure 16. Plot showing the smooth components of the fitted GAM predicting macrophytes percent volume index (PVI) as a function of environmental variables, month, and region (Lower Vasse, Upper Vasse, Lower Wonnerup, Upper Wonnerup) between 2017 and 2020. Values in the vertical axis title represent the estimated degrees of freedom (edf) smooth term. Confidence bands for smooths illustrating upper and lower lines are added at 2 standard errors above and below the estimate. Confidence bands illustrate the uncertainty about the overall mean ( $\pm 2$  standard errors). A ‘rug plot’ on the x axis illustrates samples taken.

## 4 Seasonal conceptual model

Based on the outcomes of the model results presented above, a conceptual model is presented in Figure 17 illustrating the relative importance of variables to waterbird counts. The conceptual model highlights the importance of month, water area available, wind and salinity on waterbird counts as well as more complex pathways of influence of other abiotic variables. These pathways are illustrated pictorially in Figure 18 through seasonal ecosystem attributes.



**Figure 17. Conceptual model outlining the important predictor variables that were found directly or indirectly associated with waterbird counts in the VWWS based on GAMs results.**



**Figure 18. General seasonal conceptual model illustrating significant abiotic and biotic ecosystem attributes associated with different feeding guilds of waterbirds, and their relationships, in the VVWS. 1 = sedentary deep water and divers, 2 = nomadic deep water and divers, 3 = locally nomadic deep water and divers, 4 = sedentary aerial divers, 5 = various movement shoreline, 6 = nomadic surface dabblers and filter feeders, 7 = locally nomadic/sedentary large waders, 8 = migratory/nomadic/sedentary small waders 9 = macrophytes, 10 = macroinvertebrates, 11= fish, 12 = salinity, and 13 = surge barrier.**

## 5 Discussion

### 5.1 Important times and conditions for waterbirds in the Vasse-Wonnerup wetland

#### 5.1.1 Seasonality and movement

The most important and consistent predictor of waterbird counts identified from this study was month of the year. This result illustrates the importance of seasonal processes for several species in the wetland, such as migration, breeding, the condition of the Vasse-Wonnerup wetland, and/or external factors such as the availability and condition of other wetlands waterbirds may access. All species for which month was identified as important in models were nomadic or migratory, apart from the sedentary Red-capped Plover and the locally nomadic Black Swan. Many waterbird species in Australia are nomadic and respond to a stochastic climate that controls irregular and sometimes ephemeral resource availability (Kingsford and Norman 2002, Kingsford *et al.* 2010, Norman and Chambers 2010). Irregular heavy rain episodes in the continent's interior make previous unsuitable areas wetlands with plentiful food resources available (Roshier *et al.* 2002). Large numbers of individuals, sometimes in the thousands, belonging to a wide range of species of waterbirds concentrate in these wetlands, many of them to breed (Pedler *et al.* 2014, Pedler *et al.* 2018). Thus, counts of these nomadic birds in coastal wetlands can be unpredictable and dependent on conditions external to systems such as the Vasse-Wonnerup wetland.

In Western Australia (WA), wetlands are highly dynamic (Department of Environment and Conservation (DEC) 2012), and episodic/intermittent and seasonal lakes and wetlands are dependent on intense local rainfall to create habitats for waterbirds (Halse *et al.* 1998, Halse *et al.* 2005). The South West region of WA has a Mediterranean climate, with dry summers and wet winters (Halse *et al.* 1993). In the South West interior in particular, rainfall occurs mostly during Austral winter months (Pedler *et al.* 2018). The wetlands in the South West region are also highly dynamic with a strong seasonal hydrological regime, with winter flooding and summer drawdown/dry period (Davis and Froend 1999, DEC 2012). It is in winter, therefore, when many of the episodic and seasonal wetlands are flooded and available for the nomadic waterbird species (Chapman and Lane 1997). More specifically, flooding of the wetland begins in early winter and maximum depths of the waterbody are reached in late winter (August) to early spring (September) (Halse *et al.* 1993, Davis and Froend 1999), with a drawdown or drying period in summer-autumn (Davis and Froend 1999, DEC 2012). At the same time, coastal wetlands play an important role as refuge during dry/summer periods (Loyn *et al.* 1994, Kingsford and Norman 2002, Wen *et al.* 2016). The Vasse-Wonnerup wetland, thus, likely acts as a refuge in dry/summer periods for many nomadic species. The increase in numbers of many species in December with a trough in counts of all species in August-September when maximum depths for ephemeral and seasonal wetlands occur (Figure 17), indicates that there is certain synchrony among species, as has been observed in nomadic species in other wetlands in eastern Australia (Bino *et al.* 2020).

The Red-necked Avocet is the only nomadic species whose maximum numbers were predicted to occur later, between March-May, according to the optimal model. Notably, while models with an interaction between Ecological Year and Month could not be fitted for this species, the peak in empirical counts was shifted by a couple of months between the 2018-2019 and the 2019-2020 ecological year. A larger dataset over a greater number of years would further inform the timing and variability in occupancy of Red-necked Avocets in the system. Timing and variability observed in this study may reflect changes in conditions of a larger geographical area as well as within the system and the nomadic nature of Red-necked Avocets arriving to locations with suitable habitat. In comparison,

month was not featured as important in Australasian Shoveler models, but ecological year was, with the lowest and more variable count in the 2018-2019 ecological year. The variability observed in this study in the timing of nomadic species in conjunction with some overlaps in timing of relatively high empirical counts suggests that broader geographic conditions likely influence the importance of the Vasse-Wonnerup on any given year. For example, when some nomadic species were low in numbers (2017), the average rainfall across Western Australia was relatively high, while in 2019 when higher peaks in counts were observed, average conditions across Australia were much drier. No further inferences can be made without evaluating patterns over a longer period (over many years) and broader regional conditions.

Although the Australian Pelican is considered a nomadic species, our results indicate weak seasonality indicating potential behaviour consistent with locally nomadic species (Black Swan, Figure 17). In a relatively recent study on pelicans in coastal regions of South Australia, banded pelicans were recovered mostly within 300 km of their breeding site and actively maintained their coastal breeding colonies even during periods of inland floods (Johnstone *et al.* 2015). The close distance of permanent coastal wetlands with abundant food resources such as the Leschenault and Peel-Harvey Estuaries (Raines *et al.* 2000, Hale and Butcher 2007, Peel-Harvey Catchment Council 2019) makes short-distance movements of Australian Pelicans feasible among these systems, reducing the seasonal role of the Vasse-Wonnerup wetland as a summer refuge for this species. Another species presenting a weak seasonality is the Pacific Black Duck, a locally nomadic species.

For long distance migrants, like the Common Greenshank, the seasonal use of wetlands in Australia is mostly regulated by the periods of their annual cycle and arrival to the Australian continent where they spend their non-breeding, pre-migration and post-migration periods (September-October to February-April) (Tulp *et al.* 1994, Wilson *et al.* 2007, Minton *et al.* 2013). This aligns with the higher counts of this species in the Vasse-Wonnerup wetland in the summer months and lowest counts in late winter (Figure 17). It is important to note that long distance migratory shorebird species have slightly different migratory phenologies, with some species beginning their migration earlier than others (Tulp *et al.* 1994, Wilson *et al.* 2007, Minton *et al.* 2013).

The other two species for which month was identified as important in models were the Red-capped Plover (sedentary) and the Black Swan (locally nomadic), with both breeding in the Vasse-Wonnerup system. The Red-capped Plover is an opportunistic breeder, breeding most of the year if conditions are favourable except in May (Halse and Jaensch 1989, Holland and Minton 2012). Red-capped Plover's high numbers in March-April (Figure 17) could be related to its breeding and dispersal of individuals depending upon conditions in the Vasse-Wonnerup (Red-capped Plover breed in the Vasse-Wonnerup between February and April. Kim Williams, per. comm.), and perhaps surrounding systems. Whether dispersal is only within the system or to surrounding systems is unknown. Regardless, the possible temporal patterns may be related to movement to surrounding habitats with more suitable conditions at the time counts were undertaken. Black Swans breed in the Vasse-Wonnerup between July and September (Kim Williams, per. comm.), with the Vasse-Wonnerup colony being the largest regular breeding colony of Black Swans in WA. This is likely to explain, at least in part, the importance of month as a predictor for swan counts. For example, the peak numbers of Black Swans were found between July and December (Figure 17). Also, the Vasse-Wonnerup wetland is likely to provide a summer refuge for locally nomadic Black Swans from drying inland wetland systems.

These results do not preclude other waterbird species that use the Vasse-Wonnerup system also utilising the system seasonally. Indeed, the empirical counts suggest that other species also occupy the wetland seasonally. However, the limitations in data (e.g., single monthly counts over three years)

used to evaluate patterns in the present study likely did not provide models with sufficient power to detect these relationships if they do exist. Importantly, as more intensive and extended drought periods are projected due to climate change, coastal refuges like the Vasse-Wonnerup will become more important for a range of waterbird species in the future (Wen *et al.* 2016).

### 5.1.2 Conditions within the Vasse-Wonnerup wetland

#### Available water area

Apart from month, the available water area for different depths at the time of counts or one month prior were the most recurrent important predictor variables for surface dabbler and filter-feeders, small and large wader counts. In addition, for all species from other feeding guilds that showed seasonality, excluding those that breed in the system and long-distance migratory species, conditions related to the available area of water were also important predictor (Figure 18).

Models for surface dabblers and filter-feeder species models suggested that peak counts were associated with available water area for water depths between >0.2-0.5 m. More specifically, the peaks in numbers for the Pacific Black Duck and Grey Teal were associated with the area of water available for depths between >0.2-0.3 m (increasing with increasing area) and >0.3-0.5 m (decreasing with increasing area, although the effect was small), respectively. In other words, greater counts of these species in December-January coincide with the area of water at deeper ranges decreasing and the shallower depth ranges increasing in area (Figure 17).

For small waders, it was the available water area for different water depths one month prior to the counts that was associated with their peak numbers. Increasing numbers of Red-necked Avocets were associated with an increased mean water area for depths between >0.3-0.5 one month prior to counts. While models with an interaction between Water Area and Month could not be fitted for this species, it is likely that there would have been an interaction between these variables. Empirical observations of Red-necked Avocets indicated that higher counts occurred in the drying and/or dry period (sometime between Jan and May). The association with depths between >0.3-0.5 m one month prior, only occurred when the system was approaching the drying period. For the Black-winged Stilt, a peak in counts was associated with a maximum water area of 100,000 m<sup>2</sup> at depths between >0.1-0.2 m a month prior to counts. Similar to the Red-necked Avocet, while models with an interaction between Water Area and Month could not be fitted, it is likely that there would have been an interaction. Empirical observations of Black-winged Stilts indicated that higher counts occurred in the drying period (December). The association with depths between >0.1-0.2 m one month prior only occurred when the system was approaching the drying period. During the drying period there were greater areas in depth ranges these species are expected to feed in, >0.01 to 0.1 m (Estrella *et al.* 2015, Ntiamoa-Baidu *et al.* 1998, Estrella pers. obs.), which indicates that their feeding grounds were inundated prior to their arrival. Therefore, for these two species, conditions that likely influenced food availability leading up to their arrival in large numbers were important, at least for the three years examined in this study. By arriving just after the transition period from a system dominated by deep water areas to one dominated by shallow water areas (Figure 17), these species may maximise their seasonal occupancy over which suitable shallow water habitat is available to feed in.

Increasing counts of the Australian White Ibis were associated with decreasing maximum water areas at 0.01-0.05 m depths the month before. Relatively large maximum areas of water for these depths are mostly found in the “drying/dry” period, from December to April, at the commencement of which coincides with peak Australian White Ibis counts. Australian White Ibis feed mostly on benthic invertebrates in the Vasse-Wonnerup wetland (Lane *et al.* 2018). According to DBCA, their optimal

foraging habitat for benthic invertebrates is between depth ranges between 0.05 and 0.14 m. Similar to the small waders, by arriving just after the transition period, from a system dominated by deep water areas to one dominated by shallow water areas (Figure 17), these species may maximise their seasonal occupancy over which suitable shallow water habitat is available to feed in.

Australian Pelican counts increased with decreasing shallow water areas, particularly for depths between 0.01-0.05 m. The feeding behaviour of this piscivore, which scoops fish with its beak's pouch from the surface, would require deeper waters to feed (between 0.38-1 m; see Lane *et al.* 2018). While the model results would seem contrary to the foraging behaviour of this species, the dominant peak in numbers occurred in 2019, which coincidentally was associated with larger areas of deeper water and smaller areas of shallower waters than previous years. In the case of the aerial diver, the Whistling Kite, a significant association with area of water was identified in the model, however the effect was very small. This result may be due to the fact that numbers counted during any single monthly count consisted of only several individuals, which may have been the same individuals given the species is sedentary and territorial.

While specific areas for depth ranges and times (i.e., one-month lag or at time of counts) were identified in models, collinear variables including some areas at different depth ranges or the same depth ranges but at different times are also likely to be important for these species. Where possible, these relationships have been discussed above to assist in interpretation. The relationship in areas at different depth ranges, such as variables included here, are likely reflective of the total hydrological attributes of the system that ultimately influences waterbirds occupying the wetland.

#### Turbidity, salinity and wind speed and direction

Turbidity, salinity, and wind speed and direction, were important variables included in the optimal two-variable models associated with counts of the Australasian Shoveler, large and small waders and deep water and diver species (Figure 18). While the Australasian Shoveler is a highly specialized zooplankton filter feeder (Crome 1985), in the Vasse-Wonnerup wetlands, it also feeds on macrophytes (Lane *et al.* 2018). Australasian Shoveler numbers peaked at salinities of ~30‰. Incidentally, total macrophyte PVI was important in predicting the species' counts, and *Ruppia* spp., *Cladophora* and *Rizoclonium* sp. presented high PVI at salinities below 40‰ (see results). However, peak numbers of Australasian Shoveler did not coincide with peak PVI, hence, it is likely that its relationship with salinity and macrophytes is complex and depends upon other conditions in the system. The species numbers were positively related to increasing minimum wind speeds, although the effect was not large. It is unclear whether numbers, which mostly appeared to increase in June based on the empirical data, were directly influenced by wind or whether another seasonal driver that coincided with high winds during the early wet period caused this result. If wind is a direct driver, it may be that its role in sediment resuspension in shallow systems (Bever *et al.* 2018) may result in increased planktonic invertebrate availability (through resuspension), with waterbirds concentrating and feeding on high prey densities that accumulate in the opposite direction of the wind (Verkuil *et al.* 2003, Estrella *et al.* 2015).

High numbers of Red-capped Plovers and Silver Gulls were associated with different wind directions in the month prior to the counts. For the Red-capped Plover, counts peaked during the period when winds dominate from the south and south-east (around September to May) and occurred prior and during peak counts (around March). For the Silver Gull, minimum wind direction was to the north which occurred one month prior to the general south-easterly direction of the mean wind direction during the time counts were highest (around January). This period is when wetlands are beginning to dry and optimal foraging habitat increases in availability where waterbirds may concentrate. The

drying and dry periods (November-January and February-April, respectively) are also when winds prevail predominantly from the south and south-west in Australia's southwest coast. In comparison, the increasing number of White-bellied Sea Eagle with wind speed could be related to the higher capacity to glide under high wind conditions (Yates *et al.* 2001) and spend more time in the air through this energy efficient mode of travel. It is unlikely that wind associated with seasonality explains the relationship, as there was no obvious seasonality in counts of the Sea Eagle, the species is sedentary, and it may be that the same few individuals were observed (and were more detectable when in the air).

White-faced Heron numbers decreased with increasing turbidity up to 2.5 NTU, and then increased with increasing turbidity above 2.5 NTU. While turbidity may impact on prey capture success by Herons and Egrets (Cezilly 1992), the effect resulting from the models was small, and without a larger sample, the reasons for the association are unclear.

Black Swan numbers were associated with salinity (Figure 18), with decreasing numbers above 20‰. As the Black Swans have nasal salt glands to regulate salt concentrations (Kingsford & Norman 2002), it is unlikely that the relationship is a direct one with salinity. Rather, the association is likely linked to their food source, including *Ruppia*. *Ruppia* (which is the dominant macrophyte in the Vasse-Wonnerup) decreased with increasing salinity and *Ruppia megacarpa* seeds germinate at salinities up to 30‰ (Kim *et al.* 2013). At the same time, it is possible that high numbers of Black Swans grazing affect negatively the abundance of their food source, macrophytes, in the wetland (per. comm. C. O'Dea), reaching a threshold where feeding in the system is not profitable anymore.

While Pied Cormorants had peak numbers in the Vasse-Wonnerup at water salinities of 13‰ (brackish water), it is unlikely that salinity itself was directly related to cormorant numbers as Pied Cormorants and Little Pied Cormorants are well adapted to high salinity environments. They have nasal salt glands which are excretory glands that help to maintain internal osmotic homeostasis under saline or hyper saline conditions (Siegel-Causey 1990). While models did not identify seasonality in this sedentary species, empirical counts were consistently highest during the winter months when precipitation is highest and salinity low. As this species is a deep-water diver, able to sustain dives at many meters depth, and opportunistically feed on small, slow-swimming, benthic-feeding fish (Trayler *et al.* 1989), their optimal habitat may be at its maximum when the area of deep water within the wetland is largest. This is during the wet period typified by relatively low salinity levels. Wind speed and direction, which were also important predictors of cormorant counts (Figure 18), may be associated with winter storms when cormorants' counts were highest. However, wind conditions could influence cormorant behaviour through their effects on wetland conditions (Van Eerden & Voslamber 1995) associated with foraging opportunities (Gremillet *et al.* 2012). For another deep water and diver species, the Australian Pelican, the increase in numbers at mid to high water turbidity could be related to the reduced fish antipredator behaviour in turbid waters (Abrahams and Kattenfeld 1997, Kimbell and Morrell 2015), as well as Pelicans' capacity to tactile feed in murky water using their bills and pouches (Australian Museum n.d.), although it is likely that the relationship is complex as the effect was very small.

#### Potential food availability

Variability in Atherinidae densities was shown to be an important predictor variable for Australian Pelican counts, although the observations for which this was based were few. While this seabird species targets a range of fish species, it is possible that small species such as Atherinidae could form part of its diet as they can catch multiple small fish at a time by expanding their throat pouches. The association with the variability of this prey species may be related to the schooling nature of this

species and its patchy distribution. However, it should be recognised that the fish sampling was biased towards smaller species, and thus the fish species and densities may not reflect food availability for pelicans in the system. Similarly, the White-bellied Sea Eagle most likely targets larger species than those collected by seine net and are not able to catch small fish species such as *Atherinidae*.

The association of Black Swan counts and total macrophyte PVI (Figure 18) is supported by Black Swans having a broad diet in the Vasse-Wonnerup wetland (Kissane 2019) and/or a seasonal change in diet from *Ruppia* to fringing vegetation in the dry season (Chambers *et al.* 2018). For the Pacific Black Duck, its numbers were associated with high *Lamprothamnium* PVI, which reached maximum values in December. December is also when maximum numbers of the Pacific Black Duck, the only nomadic herbivore of the waterbirds modelled, were found in the system.

Further inferences on the role that food resources have on the numbers of waterbirds in the Vasse-Wonnerup are not possible without more targeted foraging ecology studies.

## 5.2 Recommendations regarding manipulation of the water regime

Based on analyses using the three-year dataset that was available for this study, the conditions influencing waterbird counts, including water depth or area, was found to be complex and varied across species that use the VWWS (Figure 18). Modelling indicated that many of the small waders and surface dabblers and filter feeders demonstrated a strong association with month, which reflects the nomadic or migratory nature of many of these species and the importance of conditions external to the VWWS, such as rainfall patterns at inland wetlands (see Section 5.1 for more detail). While water depth was not identified by the models as an important factor, different components of water areas at specific depths were important for a range of these species. Since water depth and water area were collinear, both could not be included in the same models, and water area always came on top in terms of its importance, likely reflecting the availability of their habitat and foraging resources. Interestingly, of the nomadic, small wader species, only the Red-necked Avocet appeared to be influenced by water area, and the counts of this Ramsar listed species were influenced by the water area in the preceding month. This differed from the Common Greenshank, the only species out of the 37 long-distance migratory wader species that occur in Australia (protected under several bilateral and international agreements: JAMBA 1974, CAMBA 1986, ROKAMBA 2007, Bonn Convention, Ramsar Convention), which was not associated with water regimes. Furthermore, while area of specific depths appeared to influence a range of surface dabblers and filter feeders (3 of 4 are Ramsar listed), the depth and timing of those areas differed across species. It should be recognised that other co-linear variables, including area of water at different depth ranges, are likely to also be associated with waterbird counts, but the most important of these for prediction were selected in the full subsets modelling approach. In general, while there are some similarities among species, others vary in their association with hydrological conditions at different times.

On a cautionary note, the ability of the models to identify all important factors is limited by the three-year data set and the relatively coarse spatial scale (and to a lesser extent temporal scale) for bird counts. Relating the waterbird counts to water management regime was not possible, even though the Vasse estuary is managed via the surge barrier while the Wonnerup estuary is not. The models could not account for regional differences in environmental conditions due to bird counts occurring only once per region per month. Consequently, it is difficult to recommend specific manipulation of water regimes to manage waterbirds in the system. Increasing the number of years, and where possible increasing the number of counts per month (particularly at times around the period of water

regime manipulation) will likely increase the power of the modelling to more accurately predict conditions and impacts of water regime management on waterbird counts (see recommendations for future monitoring below).

### 5.3 Limitations of this study and recommendations for future monitoring

A range of analytical constraints reduced the capability of models to detect patterns that may have been present and to explain variability in waterbird counts. The most significant was the mismatch in the scale of spatial and temporal sampling between waterbirds and biotic groups for models with biotic predictors. The coarser spatial scale of waterbird sampling and coarser temporal scale of biotic group sampling meant that each dataset needed to be adapted so that scales were comparable. Waterbird data were restricted to only include quarterly counts for models including biotic data, and biotic data were reduced through calculations of means, minimums, maximums and standard deviations for each of only four regions. The resulting small sample size significantly reduced the power models had to detect statistically significant relationships. The mismatch in scales also meant that models were split into two, one including biotic predictors and one including abiotic predictors, to avoid losing power since abiotic predictors were collected on the same temporal scale as waterbird counts. In addition, the time of the month waterbird counts, and biotic sampling were undertaken did not necessarily coincide. With high variability in waterbirds available for detection during counts (i.e., occupying the wetland) and in their behaviours that may affect detectability, numbers could have potentially varied significantly between the times counts were undertaken and times other biotic groups were sampled. Ideally sampling would occur simultaneously or over proximal dates.

While waterbird models with abiotic predictors had a larger sample size than those with biotic predictors did, the high number of zero-counts (i.e., where the number of waterbirds counted for a species during a survey was '0') and the high variability of waterbird counts from month to month meant that single monthly counts still represented a relatively small sample size, particularly when considering seasonality and water regime that varied over the three years of the study. For example, to evaluate the effect of month on waterbird counts, only three counts for each month were available to fit the models. Any interactions between month and ecological year were not possible to evaluate with this small sample size. In addition, with small sample sizes, the risk of overfitting GAMs can increase. Models in this study associated with a high percentage of variability explained tended towards a small sample size. Consequently, while the number of basis functions (i.e., the maximum possible degrees of freedom allowed for a smooth term in the model) was constrained to 5 (and thereby constraining how 'wiggly' the fitted model was), overfitting (i.e., too much 'wiggleness') may have occurred. Increasing the counts within years and the overall number of years over which the data extended would help overcome this issue.

In general, models had to be simplified to include few variables at a time (and Region was not included at all for waterbird models), no random variables, and mostly no interactions so that convergence could be reached. Simplified models with a relatively small samples size (that are not overfitted) and high numbers of zero-counts are more limited in their ability to explain observed variability in the data than those that can include all or most important predictors and have a larger sample of non-zero counts. Many of the models for waterbird species in this study explained only a small percentage of variability in the data. For example, optimal models for the Grey Teal, the Australian Shelduck, the Pacific Black Duck, the Black-winged Stilt, the Little Pied Cormorant, the White-bellied Sea Eagle, and the Whistling Kite that included abiotic predictors explained between 17% and 34% of the variability. While conditions associated with waterbird counts are discussed in length above, their effect on counts may be relatively small.

Future analysis of existing long-term waterbird counts, if available, would increase the power of these models, although predictor variables to be included are best if available for the same dates or proximal dates to waterbird counts. Biotic sampling should include all potential prey of species modelled, and relevant potential prey groups. In this study, the small seine net used to catch prey species was biased towards small, slower moving fish species such as gobies (Gobiidae) and hardyheads (Atherinidae) rather than larger and more mobile species (e.g., mullet species) that are likely to form the prey of species such as pelicans and raptors. Future sampling of potential prey should use larger, less size-selective seine nets. Furthermore, studies on the diet of waterbirds occupying the Vasse-Wonnerup wetland would help inform predator-prey relationships.

Ultimately, knowledge regarding times and conditions identified as important for the ecology of the system that may be impacted by manipulation of the water regime is a desired key outcome of this study. To more directly evaluate the effects of water regime during critical times, a study incorporating a greater number of counts (identified through power analyses) matched with sampling of predictor variables before, during and after water regime manipulation is recommended. Such a study should focus on the transition drying period, when: (1) large numbers of nomadic Ramsar listed and migratory species immigrate into and occupy the VWWS; (2) water levels recede; and (3) the gates in the surge barrages are opened (at least for the three-year study period). Additionally, manipulation of the water regime through altering the surge barrage openings at different intervals and times over a long-term period while conducting intensive monitoring before, during and after would further inform water regime management. We recommend drawing on the skills and knowledge of managers and scientists, including biologists and statisticians, in a workshop to ensure that spatial and temporal scales, sampling methods and key predictors are incorporated that will allow predictive models to achieve key questions.

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## Appendix A. Waterbirds

**Table 4. All waterbirds included in the study, their feeding guilds, and known biotic composition of diet.**

Common name	Feeding Guild	Biotic composition of diet
Australian Wood Duck	shoreline	vegetation
Pacific Black Duck	surface dabblers & filter feeders	vegetation
Black Swan	deep water & divers	vegetation
Eurasian Coot	deep water & divers	vegetation
Purple Swampphen	shoreline	vegetation
Australian Shelduck	surface dabblers & filter feeders	vegetation, benthic invertebrates
Musk Duck	deep water & divers	vegetation, benthic invertebrates
Hardhead Duck	deep water & divers	vegetation, benthic invertebrates
Blue-billed Duck	deep water & divers	vegetation, benthic invertebrates
Grey Teal	surface dabblers & filter feeders	vegetation, benthic invertebrates, nekton invertebrates
Chestnut Teal Duck	surface dabblers & filter feeders	vegetation, benthic invertebrates, nekton invertebrates
Australasian Shoveler	surface dabblers & filter feeders	vegetation, benthic invertebrates, nekton invertebrates
Freckled Duck	surface dabblers & filter feeders	vegetation, benthic invertebrates, nekton invertebrates
Silver Gull	shoreline	benthic invertebrates
Buff-banded Rail	shoreline	benthic invertebrates
Australian Spotted Crake	shoreline	benthic invertebrates
Sharp-tailed Sandpiper	small wader	benthic invertebrates
Black-winged Stilt	small wader	benthic invertebrates
Red Knot	small wader	benthic invertebrates
Great Knot	small wader	benthic invertebrates
Eastern Curlew	small wader	benthic invertebrates
Bar-tailed Godwit	small wader	benthic invertebrates
Black-tailed Godwit	small wader	benthic invertebrates
Pacific Golden Plover	small wader	benthic invertebrates
Red-capped Plover	small wader	benthic invertebrates
Greater Sand Plover	small wader	benthic invertebrates
Black-fronted Dotterel	small wader	benthic invertebrates
Red-necked Stint	small wader	benthic invertebrates
Australian Pied Oystercatcher	small wader	benthic invertebrates
Curlew Sandpiper	small wader	benthic invertebrates
Australian White Ibis	large wader	benthic invertebrates
Straw-necked Ibis	large wader	benthic invertebrates
White-fronted Chat	shoreline	invertebrates
Hoary-headed Grebe	deep water & divers	invertebrates
Yellow-billed Spoonbill	large wader	nekton invertebrates
Pink-eared Duck	surface dabblers & filter feeders	benthic invertebrates, nekton invertebrates
Wood Sandpiper	small wader	benthic invertebrates, nekton invertebrates
Marsh Sandpiper	small wader	benthic invertebrates, nekton invertebrates
Glossy Ibis	large wader	benthic invertebrates, nekton invertebrates
Terek Sandpiper	small wader	benthic invertebrates, nekton invertebrates
Banded Stilt	small wader	benthic invertebrates, nekton invertebrates
Red-necked Avocet	small wader	benthic invertebrates, nekton invertebrates
Common Sandpiper	small wader	benthic invertebrates, nekton invertebrates, fish
Common Greenshank	small wader	benthic invertebrates, nekton invertebrates, fish
Little Egret	large wader	fish, nekton invertebrates
White-faced Heron	large wader	fish, nekton invertebrates
Eastern Great Egret	large wader	fish, nekton invertebrates
Australasian Grebe	deep water & divers	fish, nekton invertebrates
Great Crested Grebe	deep water & divers	fish, nekton invertebrates
Little Black Cormorant	deep water & divers	fish, benthic invertebrates
Little Pied Cormorant	deep water & divers	fish, nekton invertebrates
Australasian Darter	deep water & divers	fish, nekton invertebrates
White-necked Heron	large wader	fish

■ = Ramsar-listed species

**Table 4 continued...**

Australian Pelican	deep water & divers	fish
Great Cormorant	deep water & divers	fish
Pied Cormorant	deep water & divers	fish
Eastern Osprey	aerial divers	fish
Crested Tern	aerial divers	fish
Caspian Tern	aerial divers	fish
Sacred Kingfisher	aerial divers	fish
Whiskered Tern	aerial divers	fish
White-winged Black Tern	aerial divers	fish
Whistling Kite	aerial divers	fish
White-bellied Sea-Eagle	aerial divers	fish
Swamp Harrier	aerial divers	birds, eggs, fish, invertebrates

**Table 5. Waterbird total counts, number of surveys, and mean counts per survey across all four sampled regions (i.e., the Lower Wonnerup, Upper Wonnerup, Lower Vasse and Upper Vasse) in the Vasse-Wonnerup wetland between 2017-2020.**

Species	Counts	Num. of surveys	Mean total counts per survey
Grand Total	244,822	157	1,559.38
Grey Teal	57,849	157	368.46
Black Swan	33,620	157	214.14
Black-winged Stilt	29,546	157	188.19
Pacific Black Duck	27,472	157	174.98
Banded Stilt	18,175	157	115.76
Australian Shelduck	16,923	157	107.79
Red-necked Avocet	14,963	157	95.31
Silver Gull	13,723	157	87.41
Red-necked Stint	4,785	157	30.48
Red-capped Plover	4,591	157	29.24
Pied Cormorant	3,601	157	22.94
Hoary-headed Grebe	3,490	157	22.23
White-faced Heron	2,501	157	15.93
Australasian Shoveler	2,406	157	15.32
Australian Pelican	2,319	157	14.77
Eurasian Coot	1,303	157	8.3
Hardhead Duck	1,083	157	6.9
Sharp-tailed Sandpiper	915	157	5.83
Little Pied Cormorant	672	157	4.28
Yellow-billed Spoonbill	646	157	4.11
Common Greenshank	594	157	3.78
Musk Duck	574	157	3.66
Australian White Ibis	559	157	3.56
Little Black Cormorant	414	157	2.64
Eastern Great Egret	362	157	2.31
Pink eared Duck	289	157	1.84
White-fronted Chat	230	157	1.46
Straw-necked Ibis	230	157	1.46
Whistling Kite	132	157	0.84
White-necked Heron	93	157	0.59
Glossy Ibis	78	157	0.5
Caspian Tern	69	157	0.44
Crested Tern	63	157	0.4
Whiskered Tern	59	157	0.38
White-bellied Sea Eagle	58	157	0.37
Little Egret	48	157	0.31
Wood Sandpiper	44	157	0.28
Black-tailed Godwit	42	157	0.27
Curlew Sandpiper	35	157	0.22
Black-fronted Dotterel	29	157	0.18
Australian Wood Duck	28	157	0.18
Teal Chestnut Duck	26	157	0.17
Swamp Harrier	26	157	0.17
Blue-billed Duck	25	157	0.16
Australasian Grebe	20	157	0.13
Great Knot	20	157	0.13
Bar-tailed Godwit	19	157	0.12
Purple Swamphen	12	157	0.08
Freckled Duck	11	157	0.07
Marsh Sandpiper	8	157	0.05
Greater Sand Plover	5	157	0.03
Great Cormorant	4	157	0.03
Spotted Crake	4	157	0.03
Sacred Kingfisher	4	157	0.03
Osprey	4	157	0.03
White Winged Black Tern	4	157	0.03
Pied Oystercatcher	3	157	0.02

■ = Ramsar-listed species

**Table 5 continued...**

Pacific Golden Plover	3	157	0.02
Buff-banded Rail	2	157	0.01
Common Sandpiper	2	157	0.01
Terek Sandpiper	2	157	0.01
Eastern Curlew	1	157	0.01
Australasian Darter	1	157	0.01
Crested Grebe Great	1	157	0.01
Red Knot	1	157	0.01
Unidentified medium wader	1	157	0.01

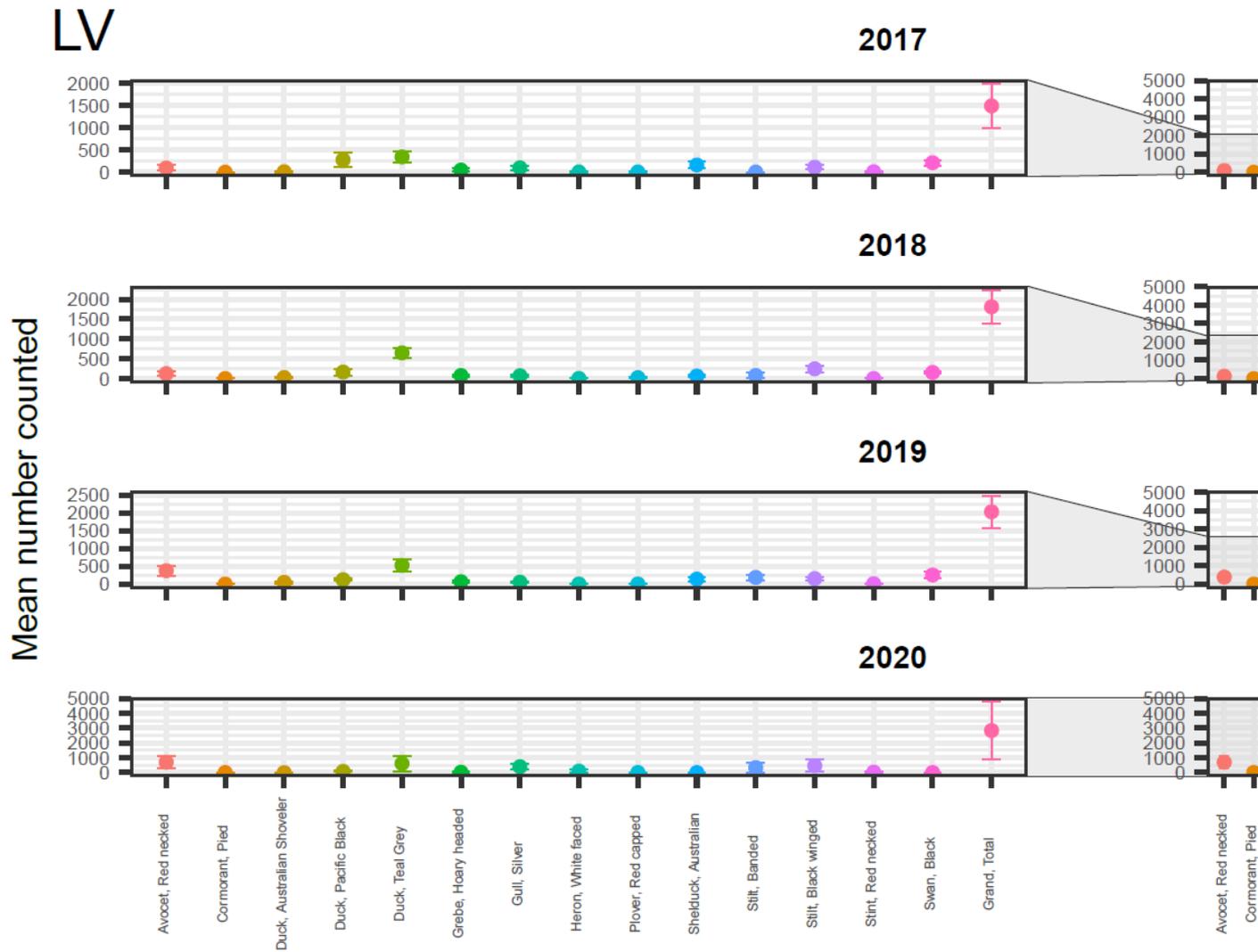


Figure 19. Mean number of waterbirds ( $\pm$  SE) per year in the Lower Vasse between 2017-2020 for species detected more than once.

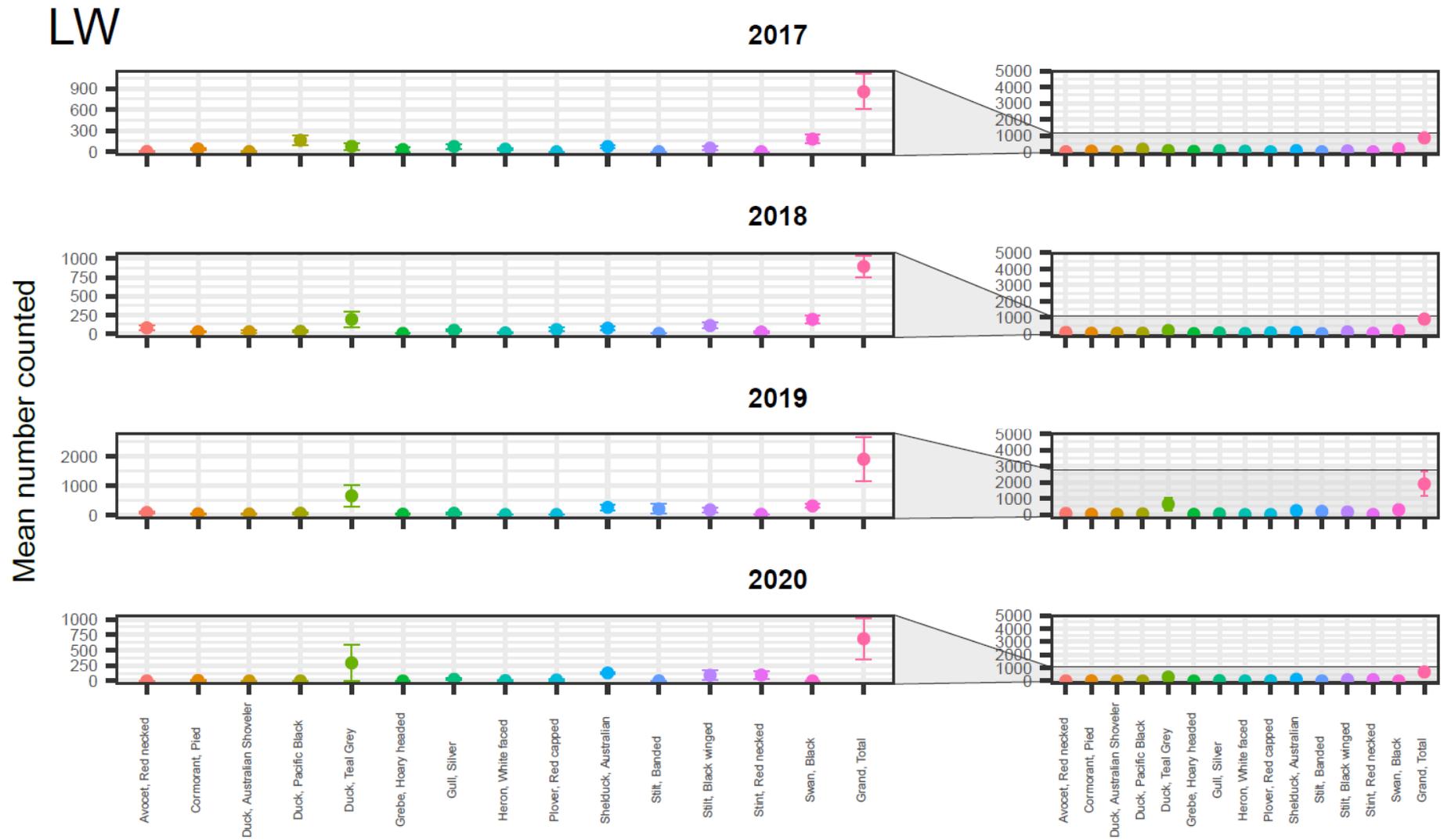


Figure 20. Mean number of waterbirds ( $\pm$  SE) per year in the Lower Wonerup between 2017-2020 for species detected more than 15 times.

# UV

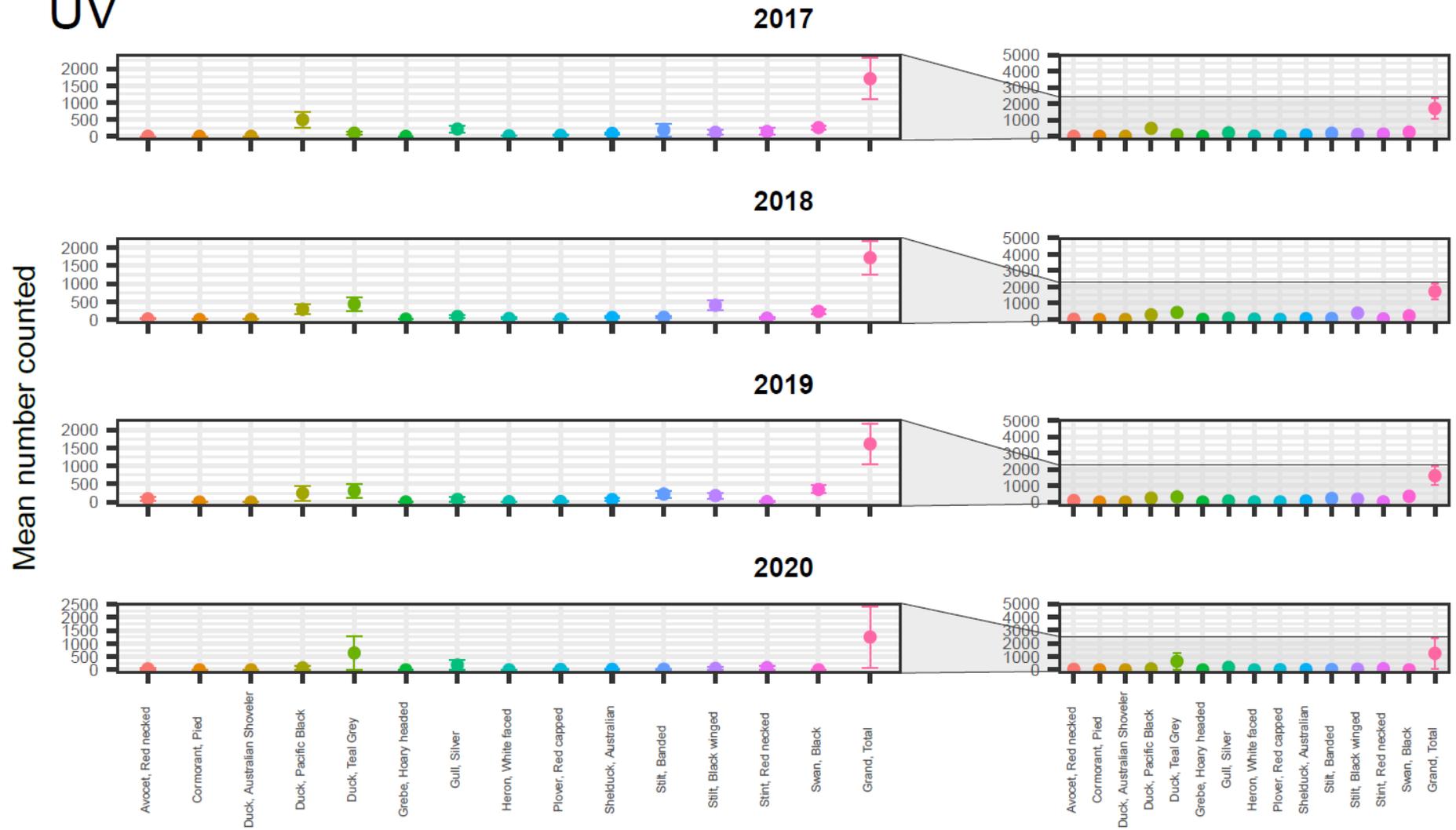


Figure 21. Mean number of waterbirds ( $\pm$  SE) per year in the Upper Vasse between 2017-2020 for species detected more than 15 times.

# UW

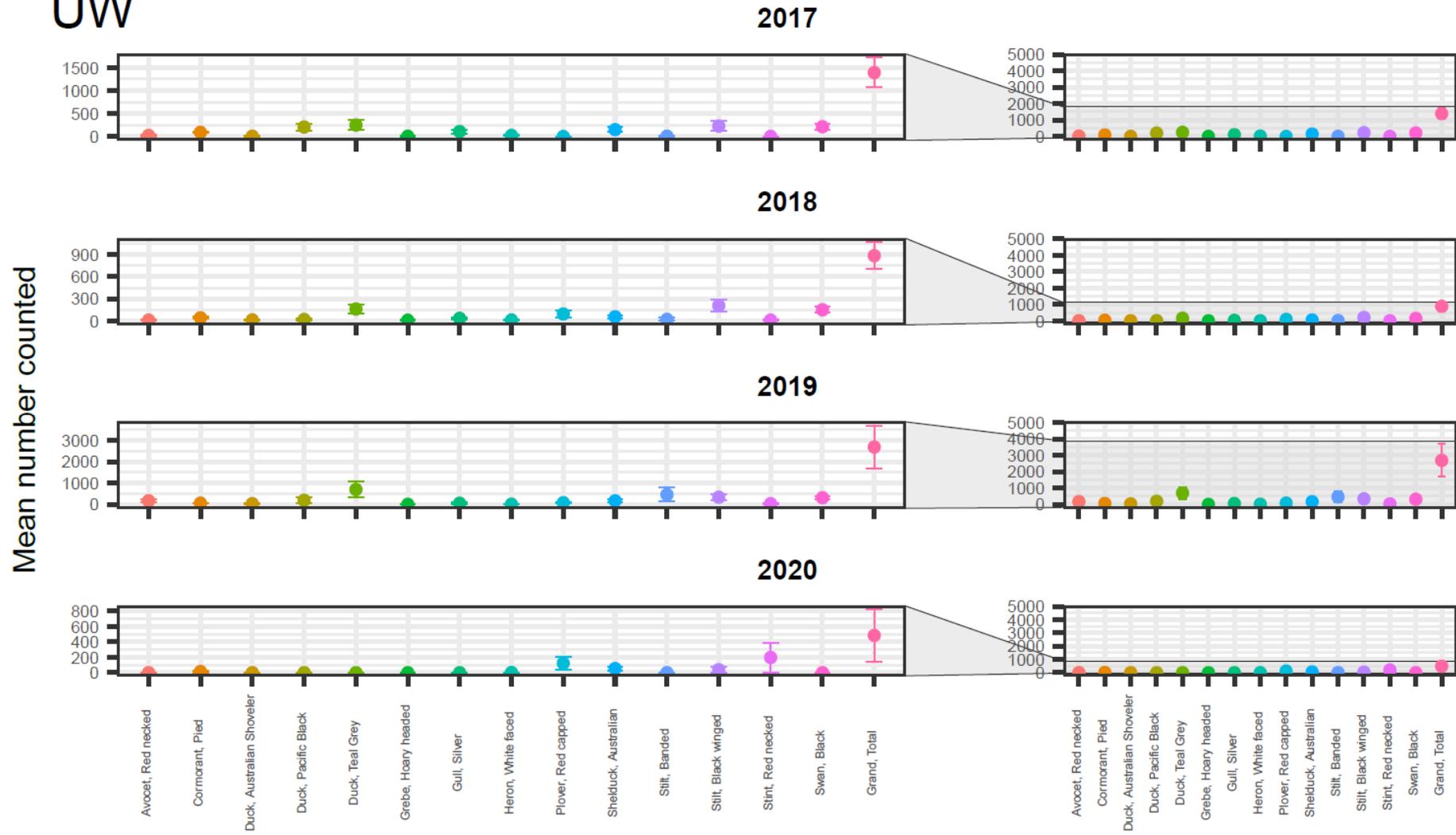


Figure 22. Mean number of waterbirds ( $\pm$  SE) per year in the Upper Wonerup between 2017-2020 for species detected more than 15 times.

LV

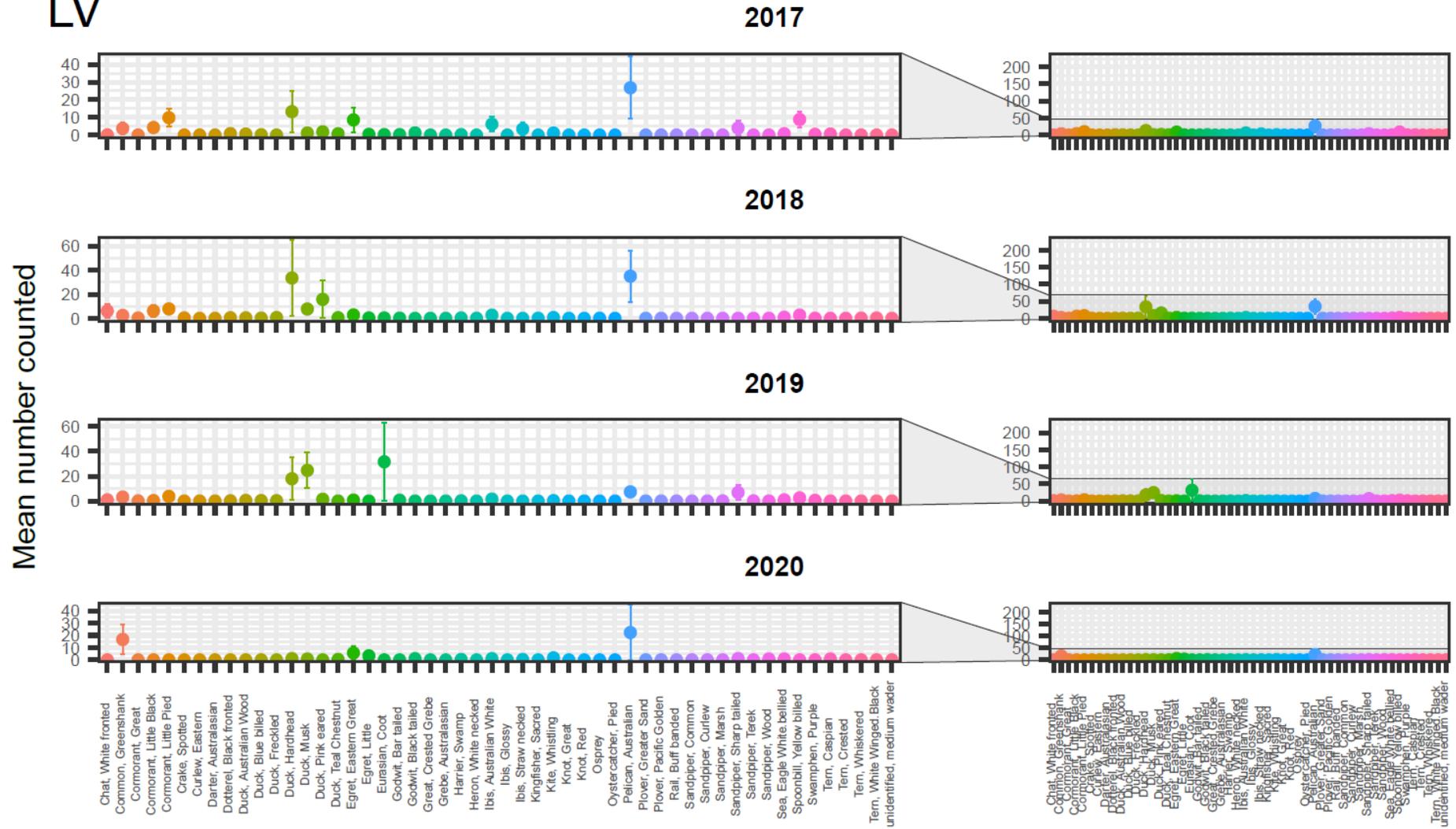


Figure 23. Mean number of waterbirds ( $\pm$  SE) per year in the Lower Vasse between 2017-2020 for species detected less than 15 or fewer times.

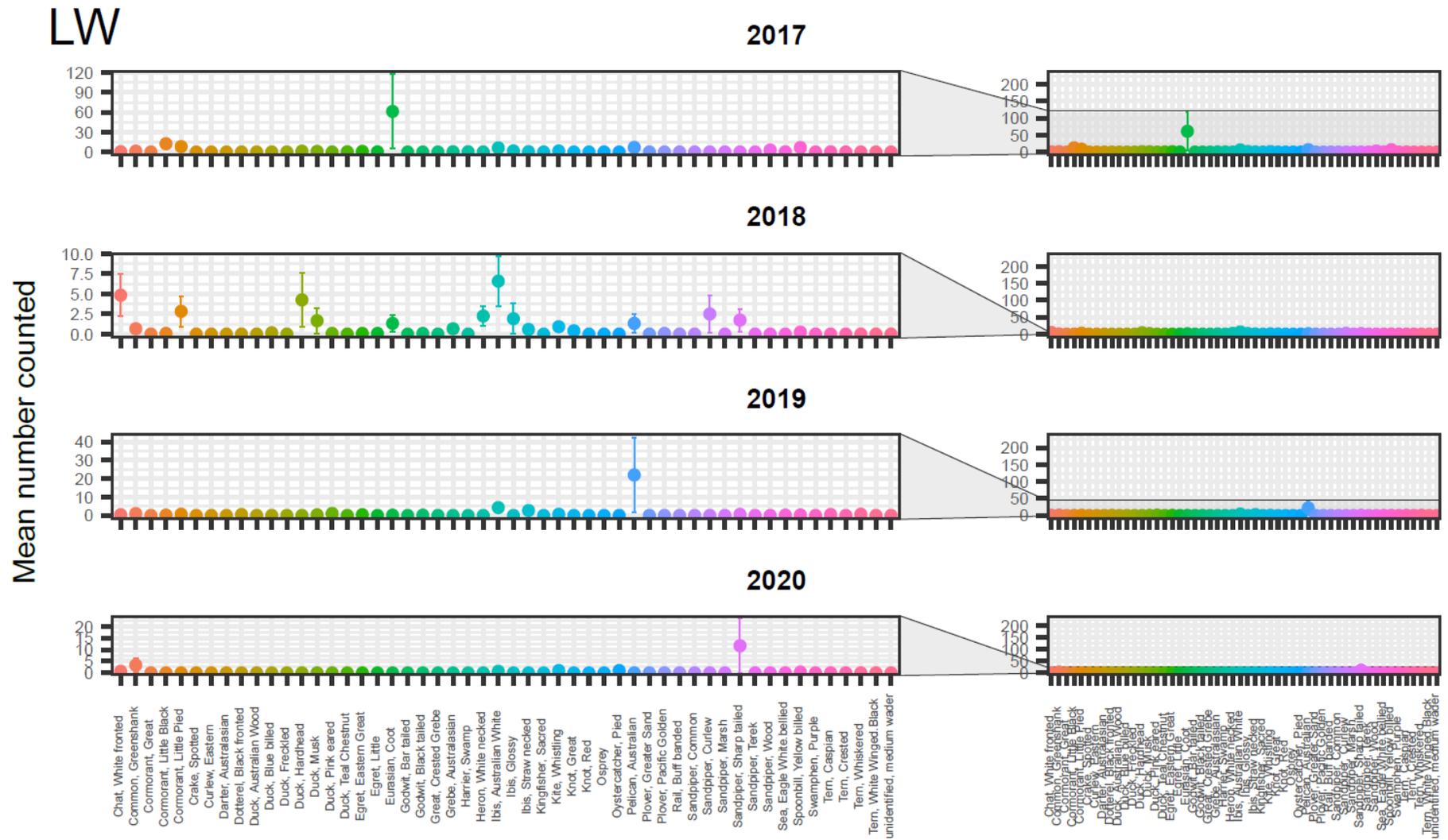


Figure 24. Mean number of waterbirds ( $\pm$  SE) per year in the Lower Wonnerup between 2017-2020 for species detected less than 15 or fewer times.

# UV

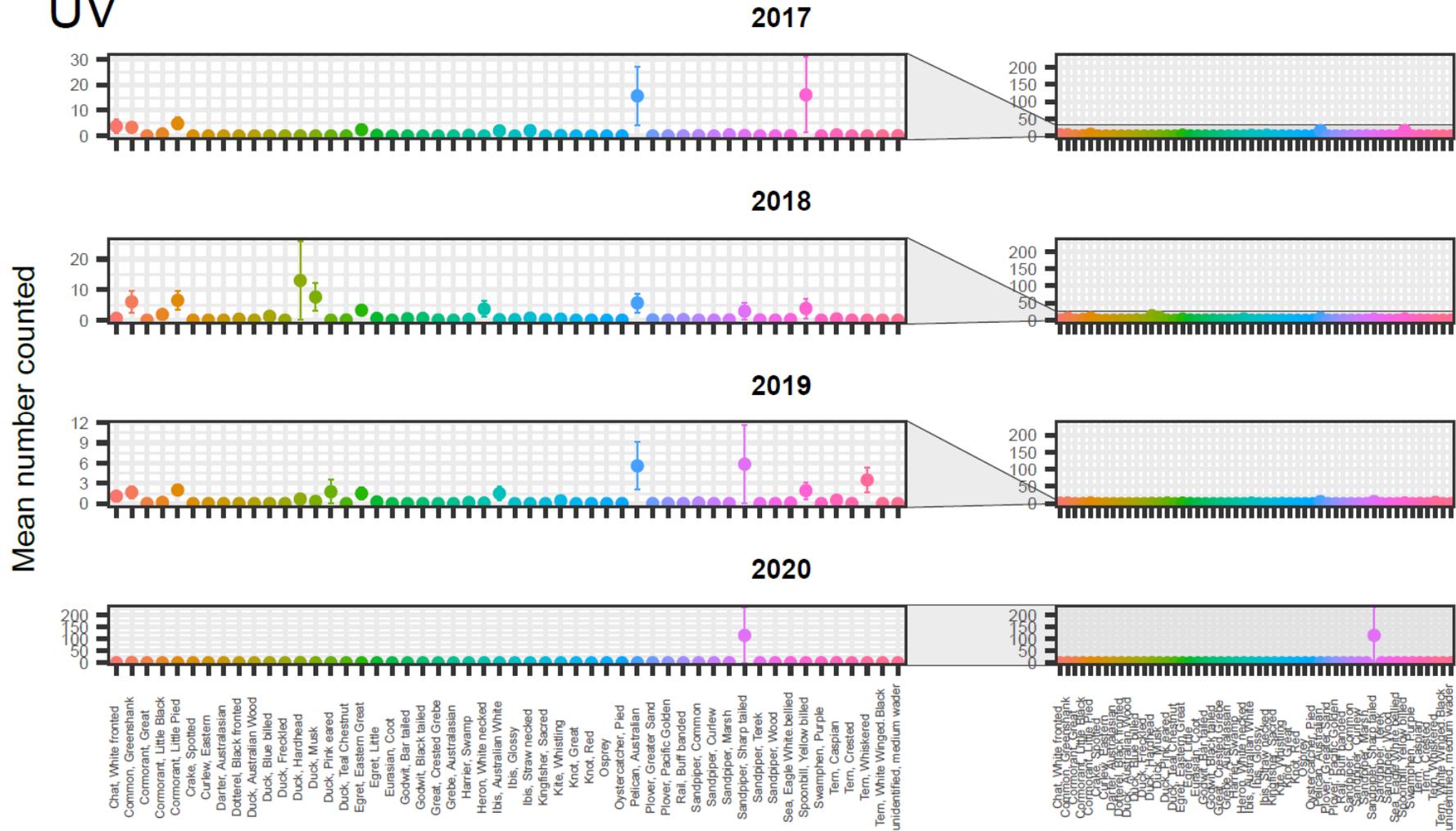


Figure 25. Mean number of waterbirds ( $\pm$  SE) per year in the Upper Vasse between 2017-2020 for species detected less than 15 or fewer times.

UW

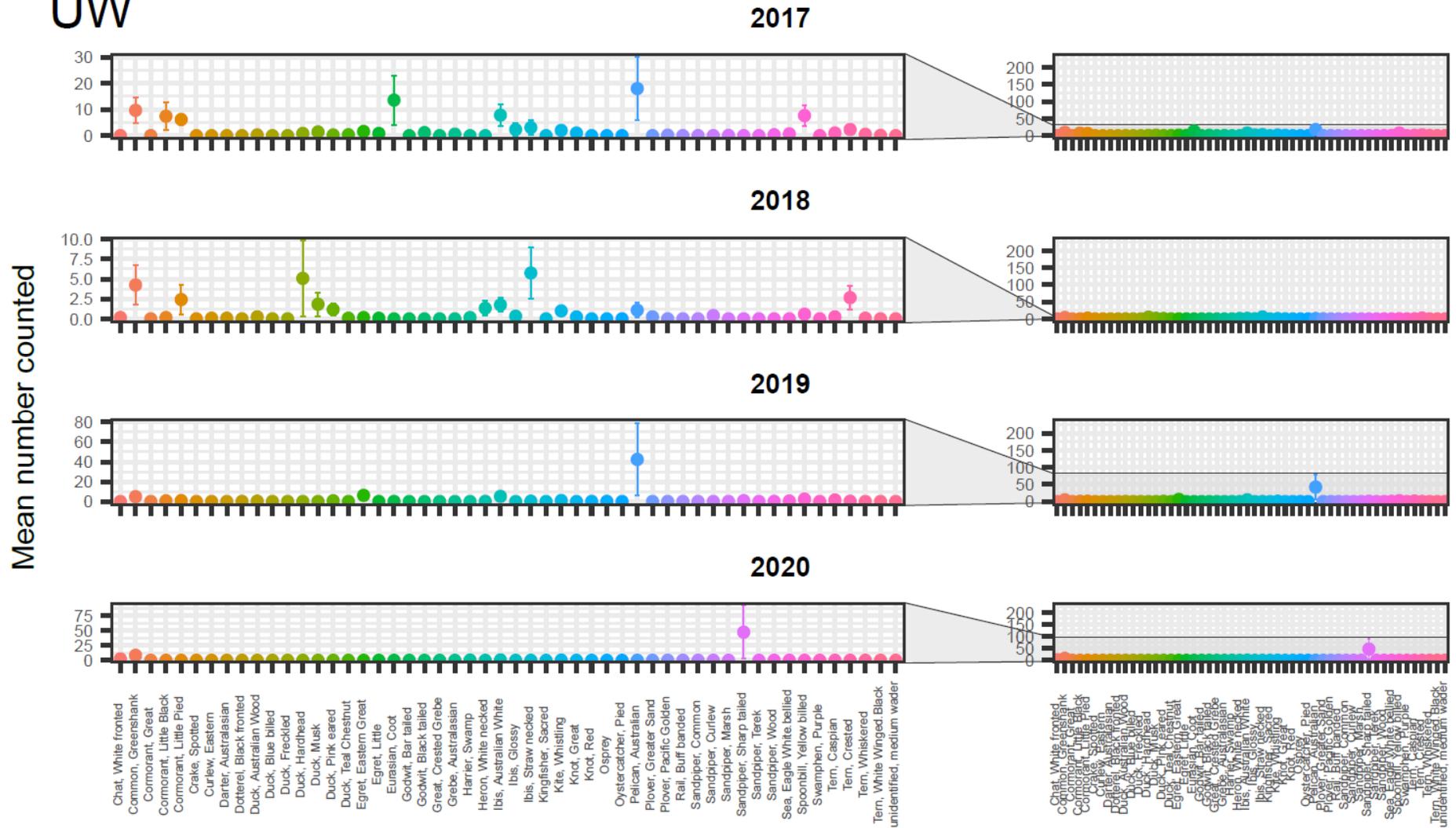


Figure 26. Mean number of waterbirds (± SE) per year in the Upper Wonerup between 2017-2020 for species detected less than 15 or fewer times.

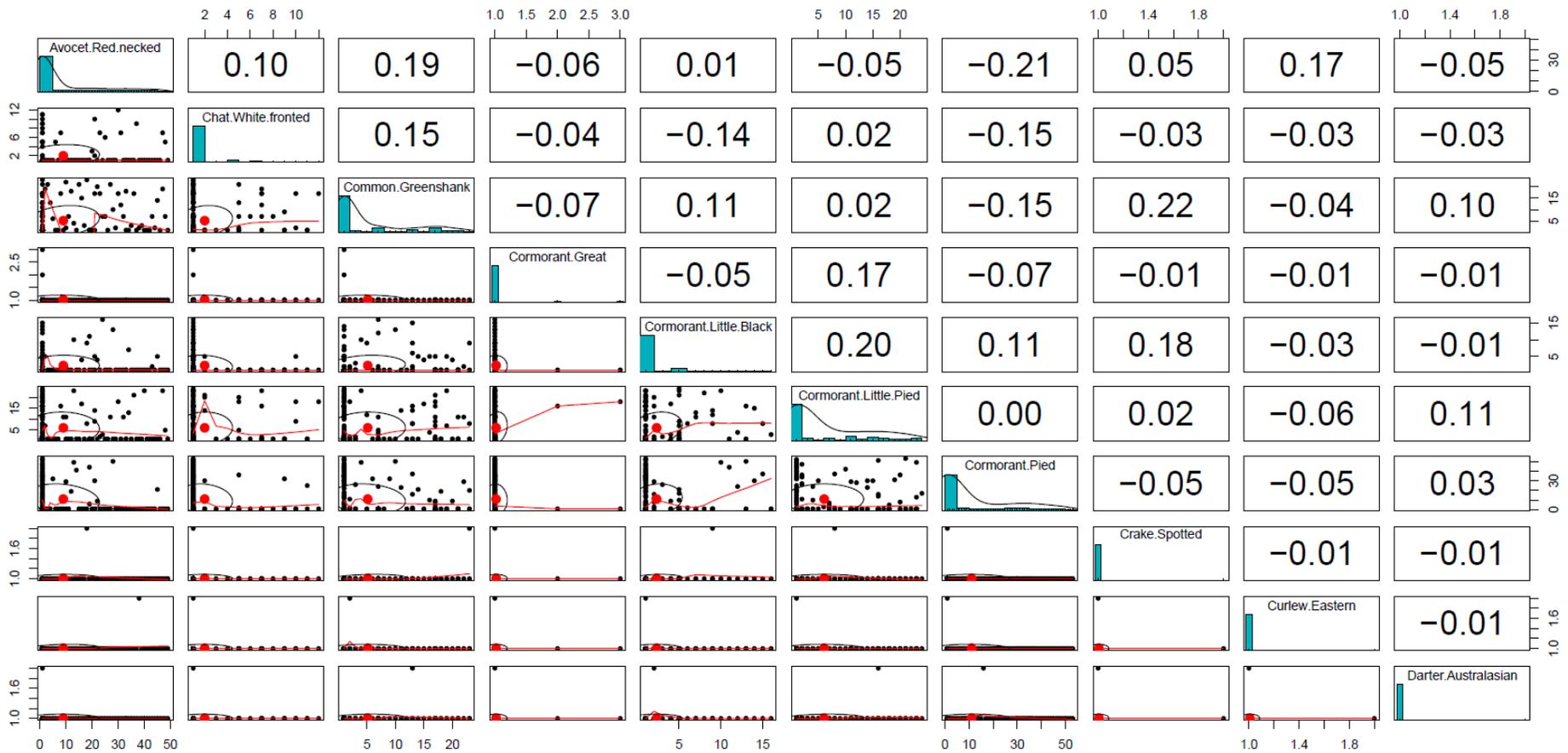


Figure 27. Collinearity plots of waterbird species counts undertaken in the Vasse-Wonnerup wetland between 2017-2020. Only 10 of the 65 species are demonstrated here. Bivariate scatter plots with correlation ellipses are below the diagonal, histograms on the diagonal, and Pearson correlations above the diagonal.

## Appendix B. Abiotic and biotic variables identified as important in predicting waterbirds

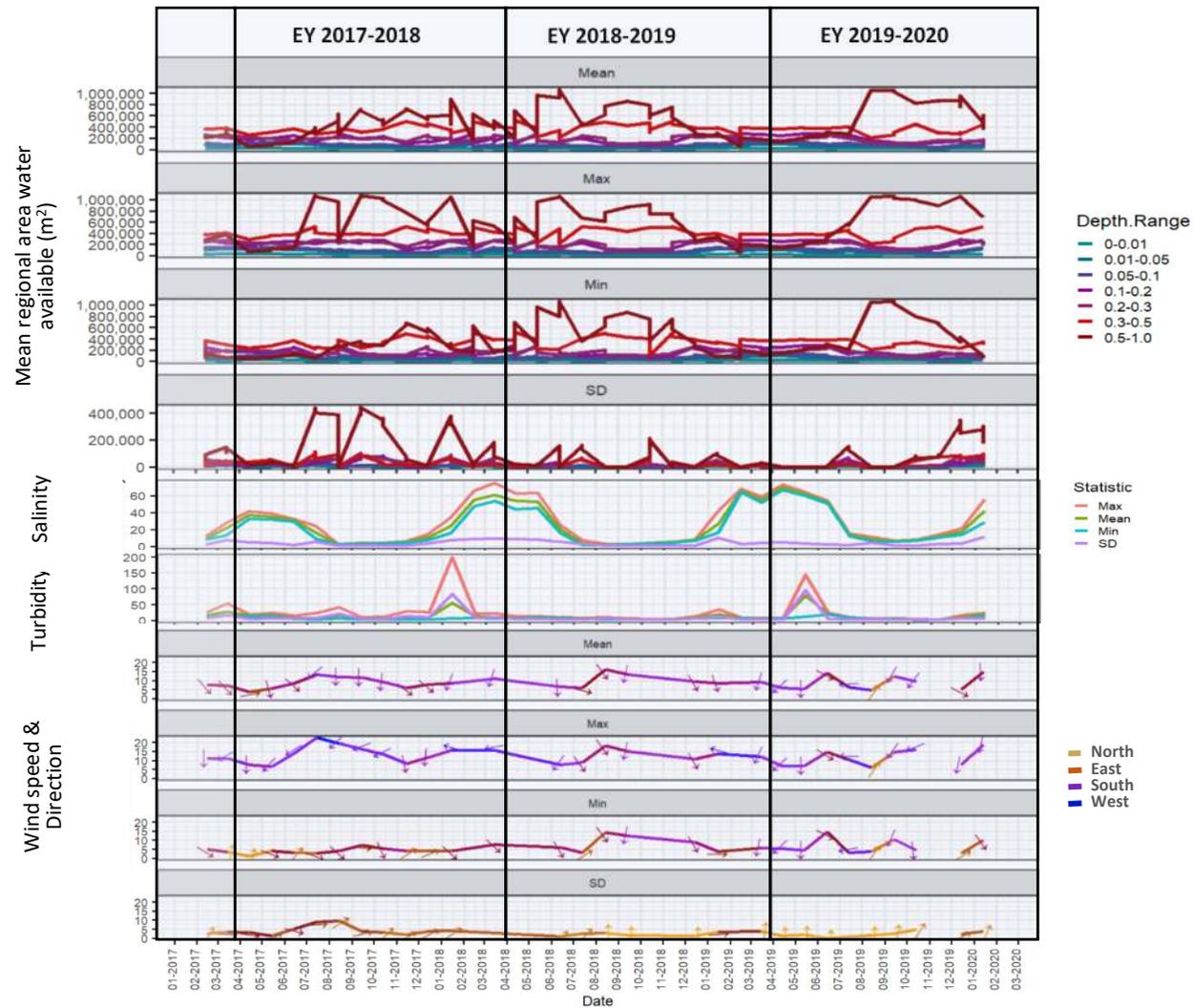


Figure 28. Abiotic variables (average values) that were significantly associated with particular waterbird species over the period of the study. Gate management is illustrated in the bottom panel.

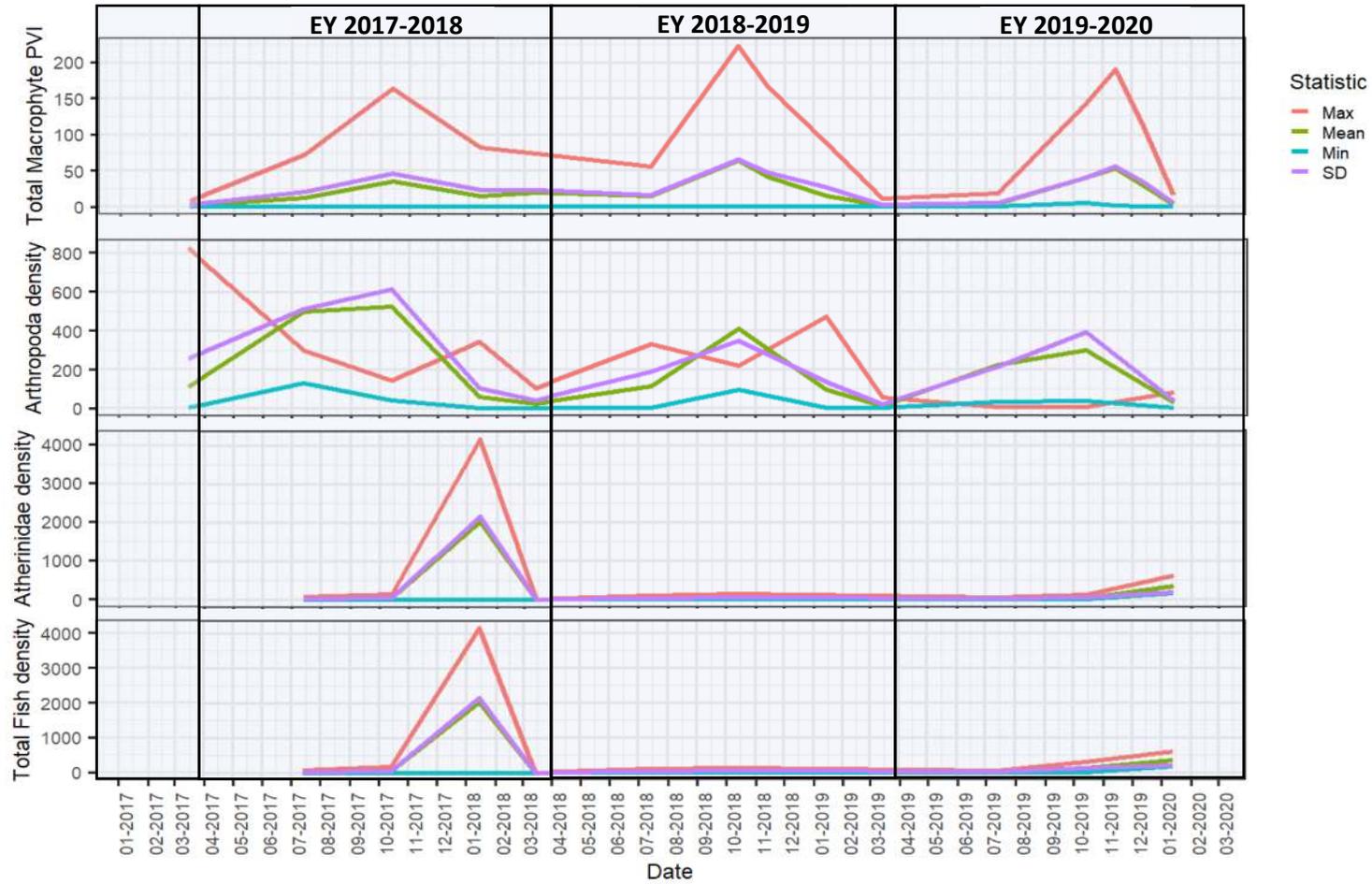


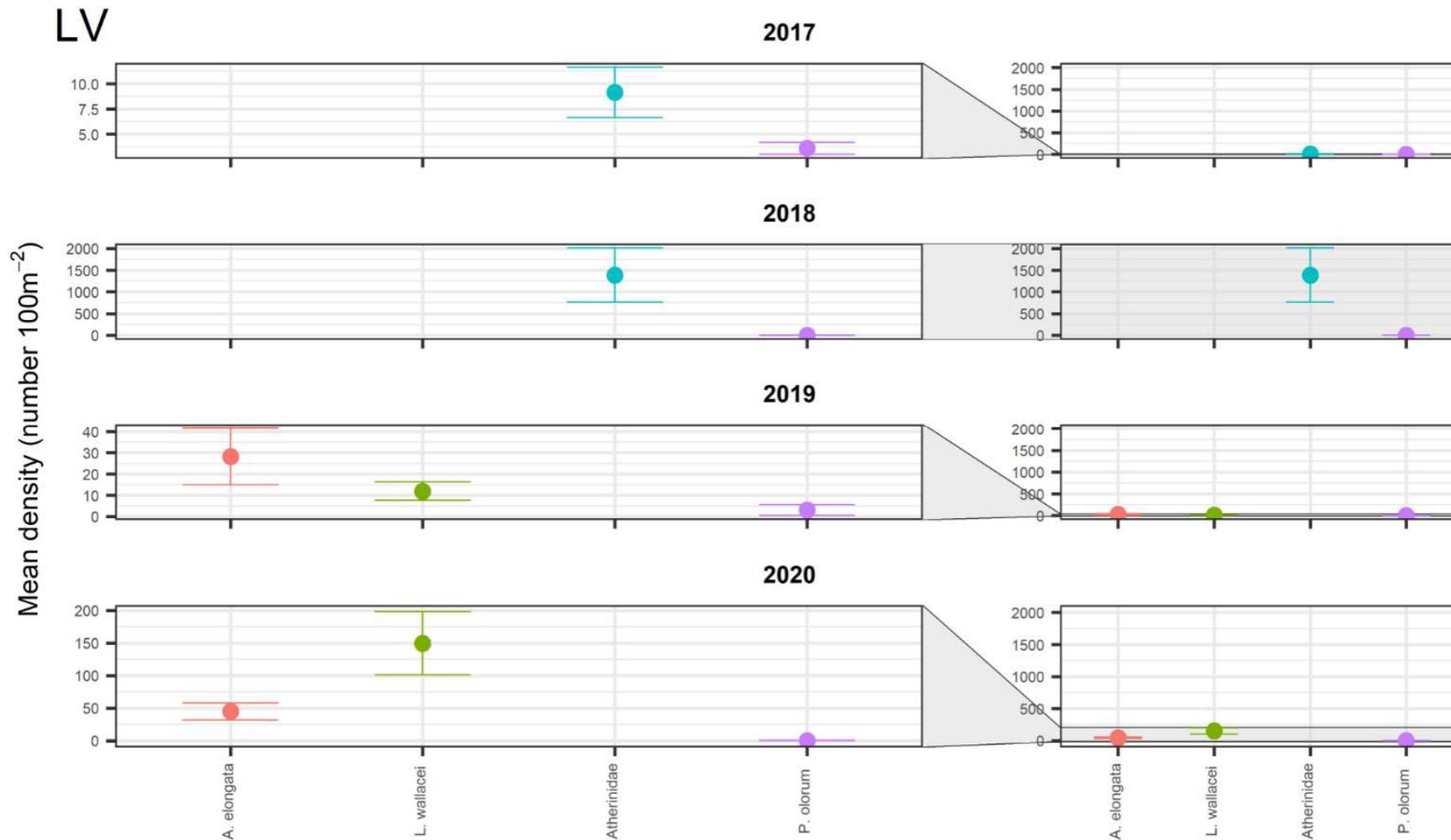
Figure 29. Biotic variables (average values) that were significantly associated with particular waterbird species over the period of the study. Gate management is illustrated in the bottom panel.

## Appendix C. Fish

**Table 6. Total number of fish species densities and sample sizes (replicates by sites by region) in the Vasse-Wonnerup wetland between 2017-2020.**

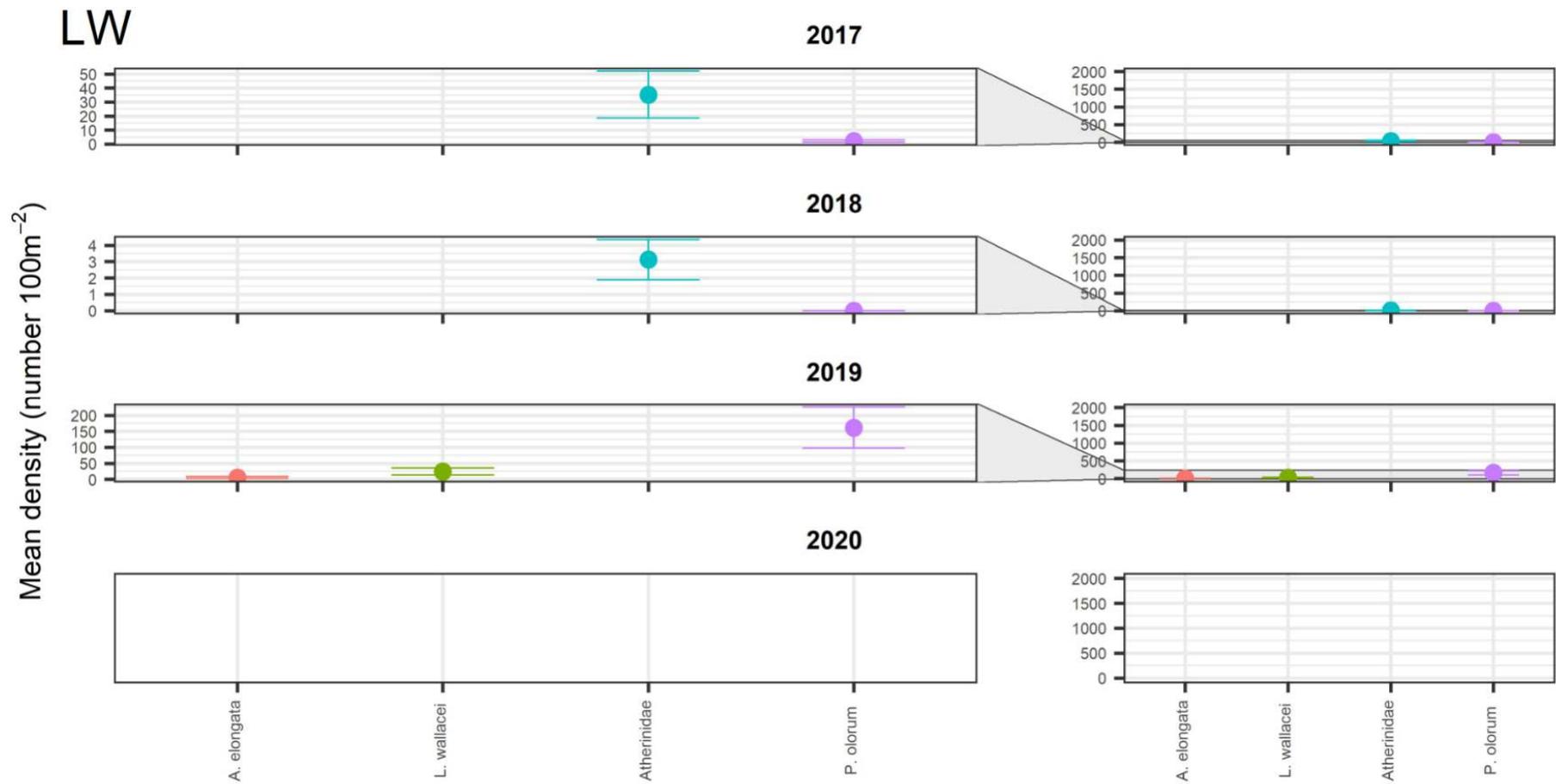
Species	Abundance per 100 m <sup>2</sup>	Sample size	Total density (number km <sup>2</sup> per sampling period across all sampled sites)
<i>A. elongata</i>	1400.86	88	159.19
<i>A. forsteri</i>	10.34	244	0.42
<i>A. vaigiensis</i>	46.55	244	1.91
<i>Atherinidae</i>	35835.34	156	2297.14
<i>C. auratus</i>	3.45	244	0.14
<i>C. mugiloides</i>	28.45	88	3.23
<i>F. lateralis</i>	1.72	244	0.07
<i>G. occidentalis</i>	28.45	244	1.17
<i>L. wallacei</i>	3163.79	88	359.52
<i>P. olorum</i>	3070.69	244	125.85

Note: *A. elongata*, *A. forsteri* and *A. vaigiensis* were included within the taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group. Thus, the total Atherinidae sampled over 2017-2020 is the sum of the abundance for *A. elongata*, *A. forsteri*, *A. vaigiensis*, and Atherinidae presented in the table).



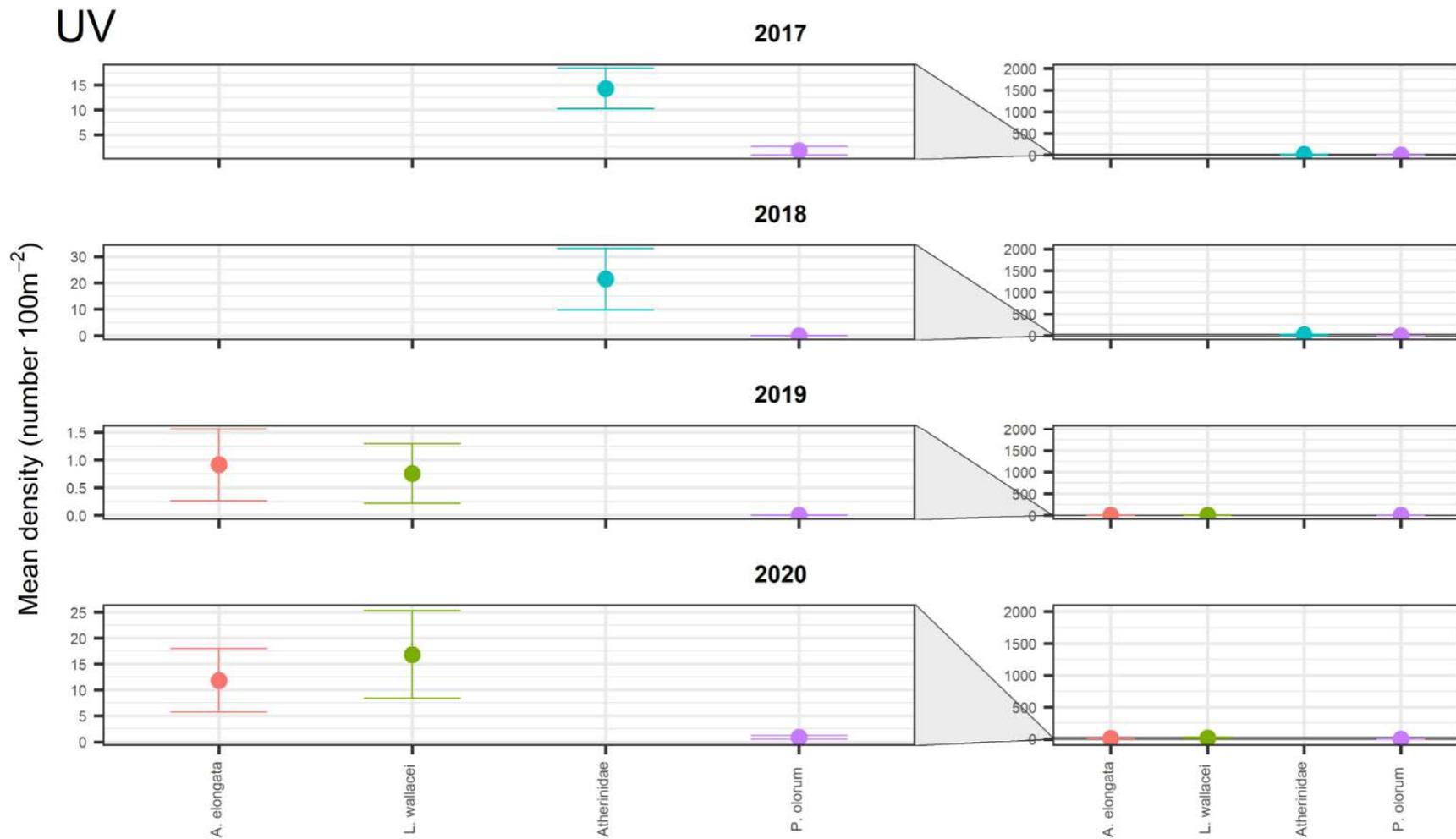
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 30. Mean number of fish per 100m<sup>2</sup> ( $\pm$  SE) per year sampled in the Lower Vasse between 2017-2020 for species encountered more than 100 times during the study.**



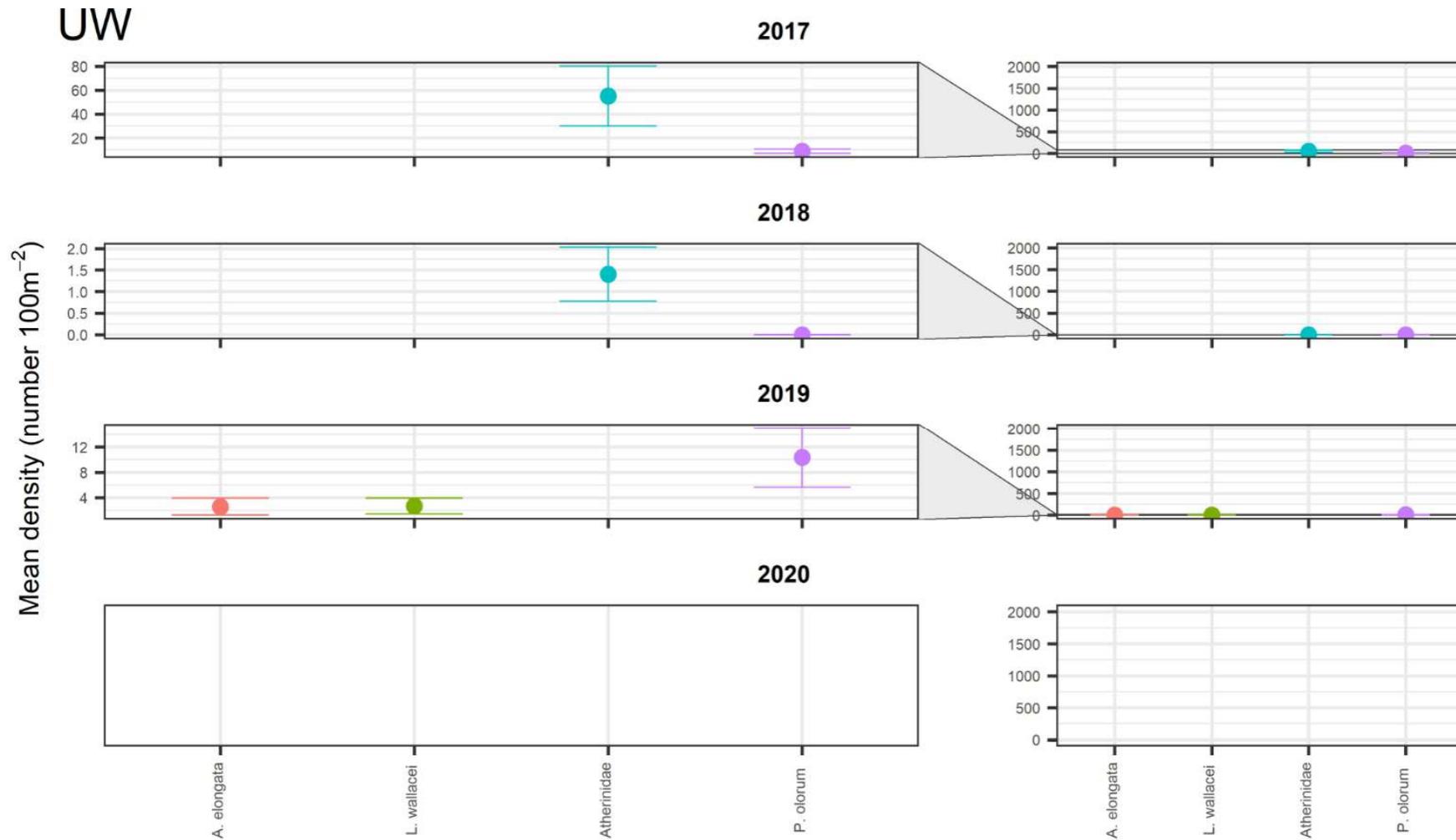
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 31. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Lower Wonnerup between 2017-2020 for species encountered more than 100 times during the study.**



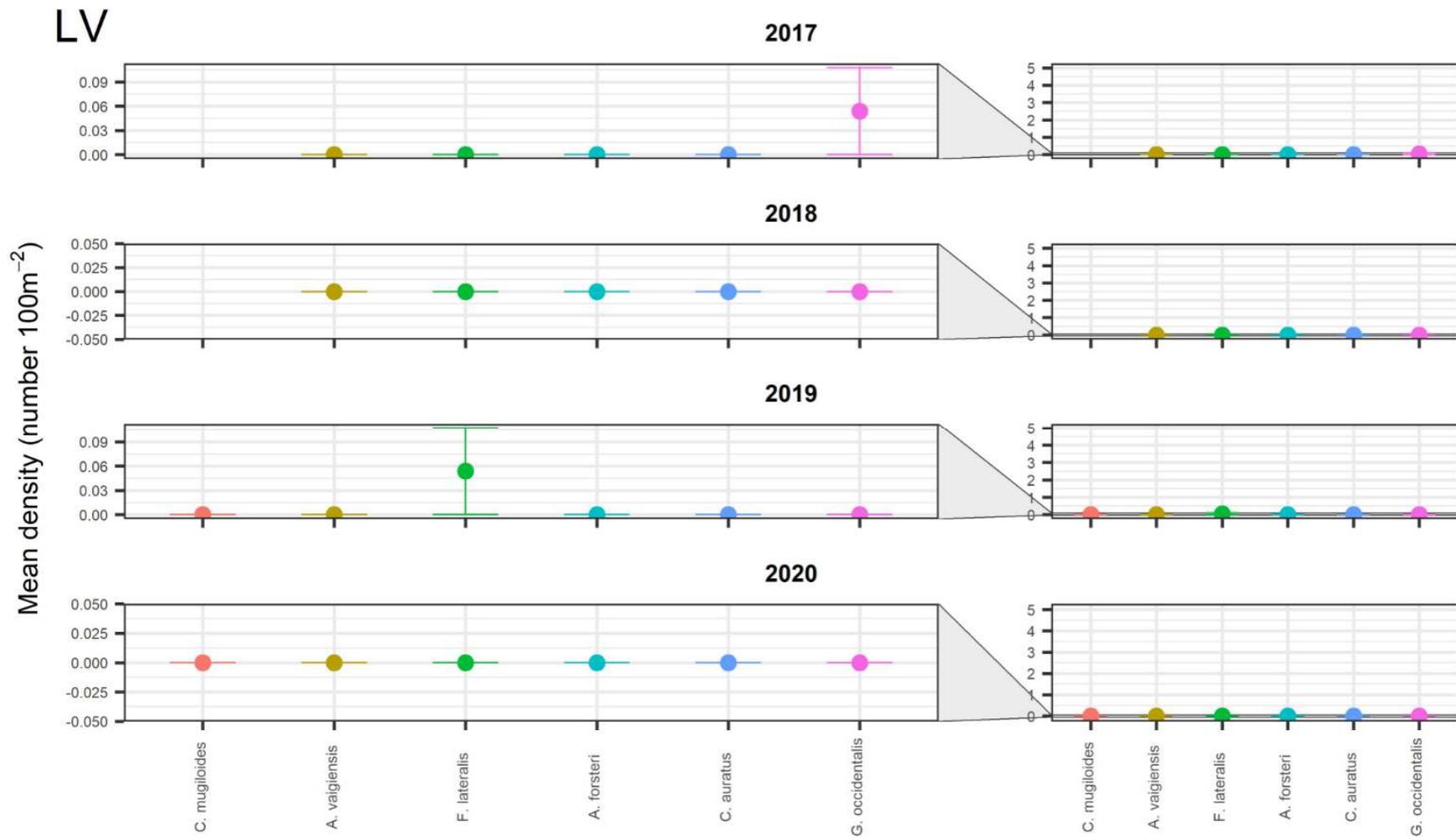
Note: *A. elongata*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 32. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Upper Vasse between 2017-2020 for species encountered more than 100 times during the study.**



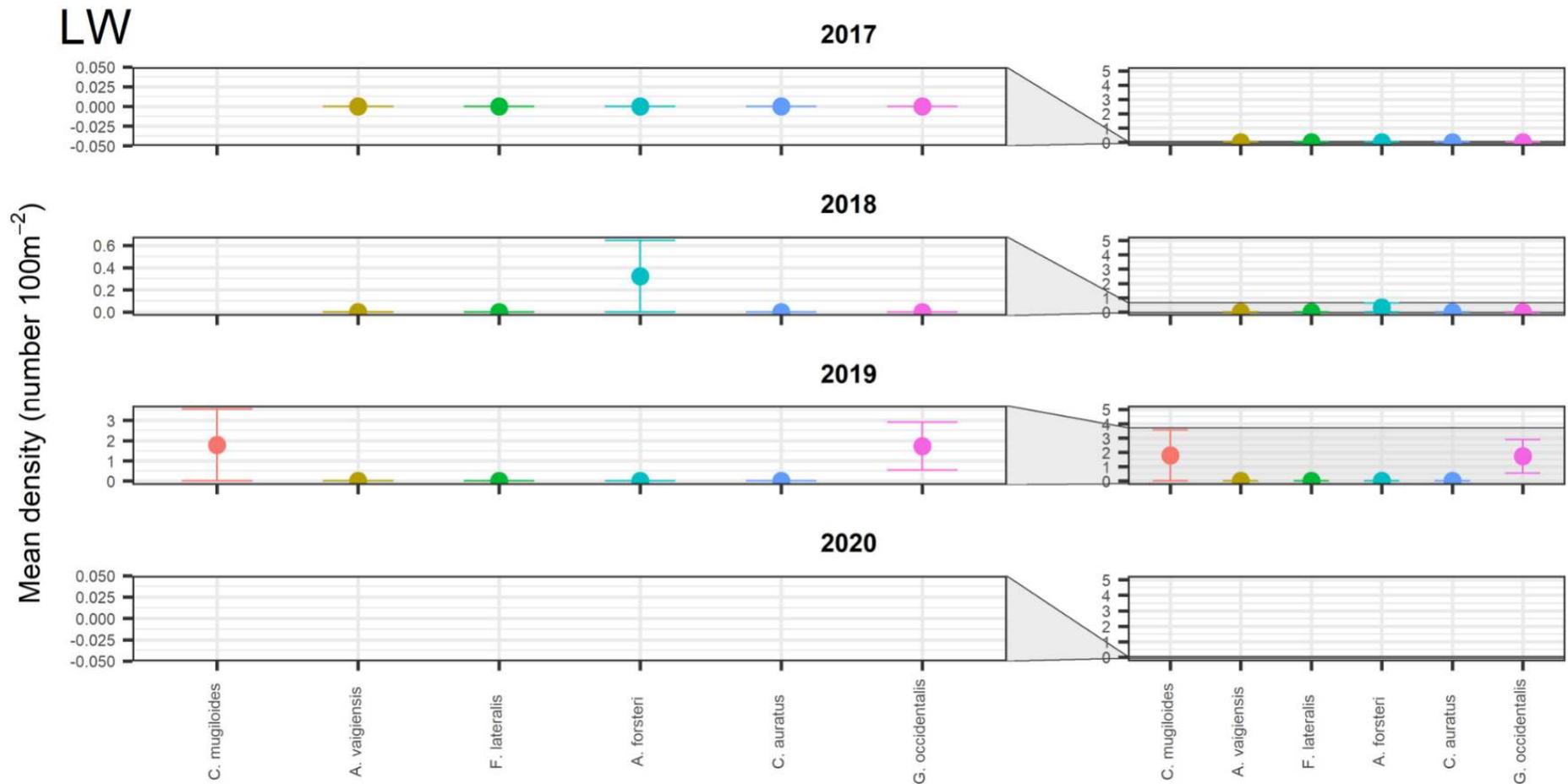
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 33. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Upper Wonnerup between 2017-2020 for species encountered more than 100 times during the study.**



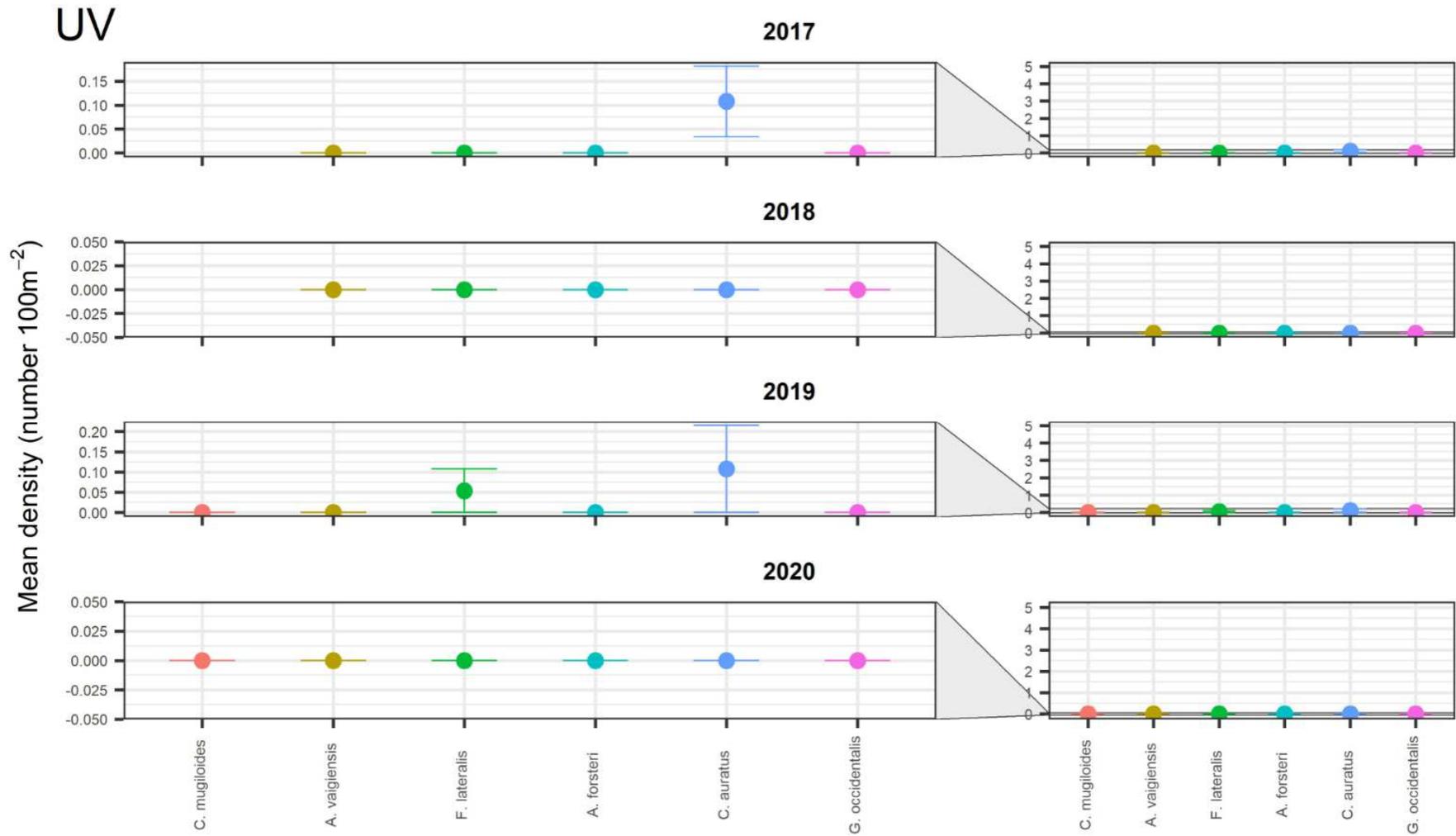
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 34. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Lower Vasse between 2017-2020 for species encountered 100 or fewer times during the study.**



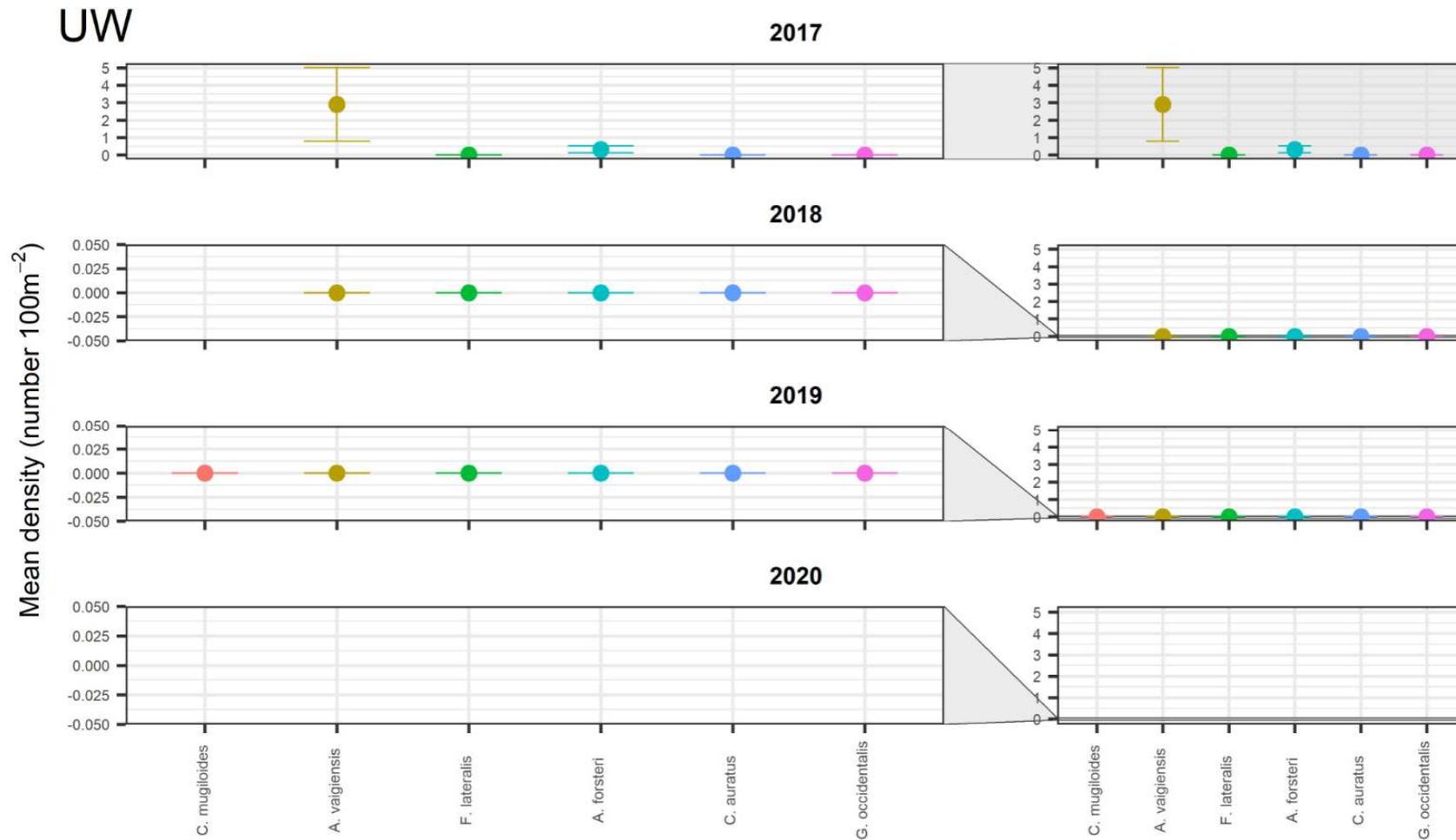
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 35. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Lower Wonnerup between 2017-2020 for species encountered 100 or fewer times during the study.**



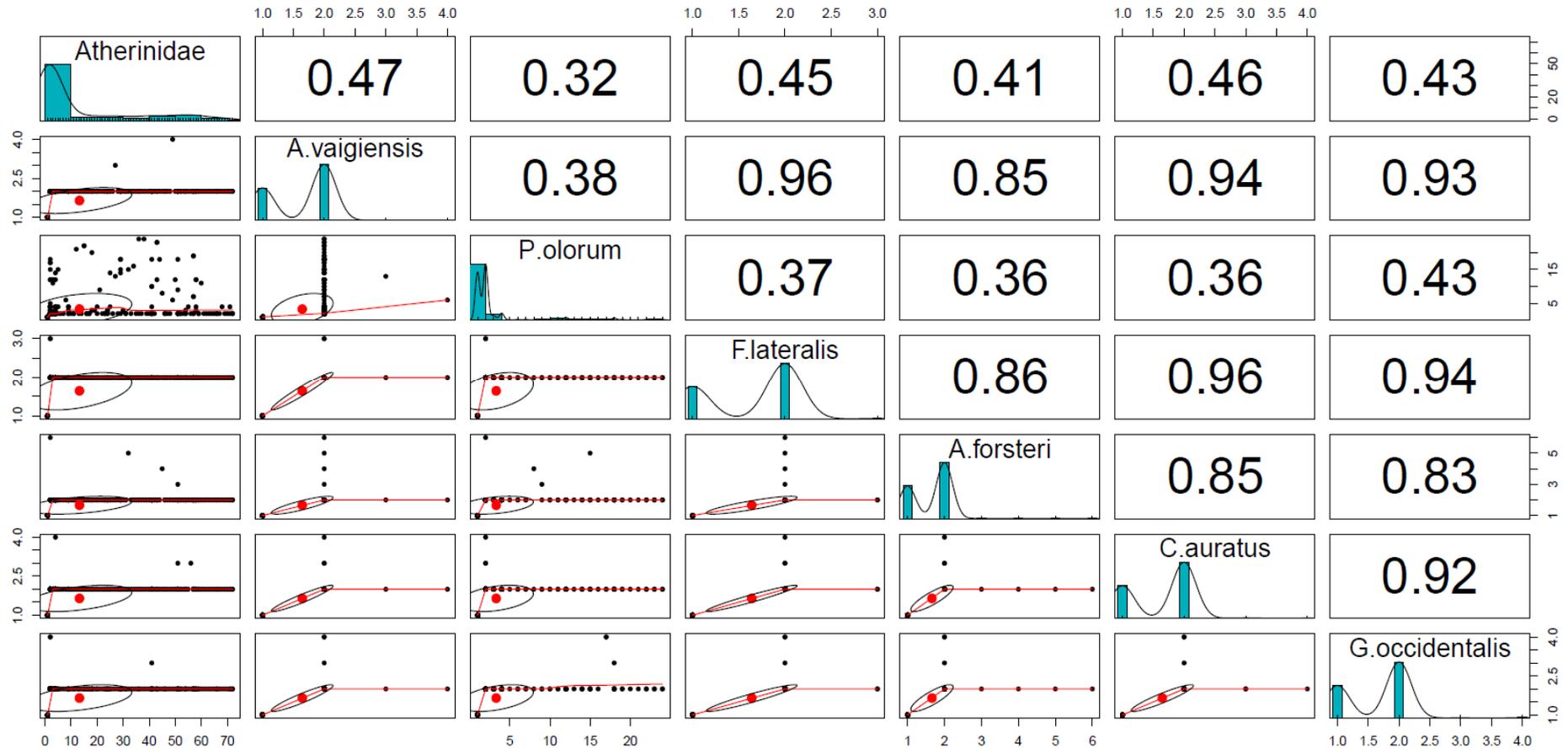
Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 36.** Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Upper Vasse between 2017-2020 for species encountered 100 or fewer times during the study.



Note: *A. elongate*, *A. forsteri* and *A. vaigiensis* were included within their taxonomic grouping as “Atherinidae” between 2017 and autumn 2019, and thereafter were identified by species rather than the taxonomic group.

**Figure 37. Mean number of fish per 100m<sup>2</sup> (± SE) per year sampled in the Upper Wonnerup between 2017-2020 for species encountered 100 or fewer times during the study.**



Note: Atherinidae include all *A. elongate*, *A. forsteri* and *A. vaigiensis* sampled.

Figure 38. Collinearity plots of fish species counts per sample in the Vasse Wonnerup wetland between 2017-2020. Bivariate scatter plots with correlation ellipses are below the diagonal, histograms on the diagonal, and Pearson correlations above the diagonal.

## Appendix D. Invertebrates

**Table 7. Total number of invertebrate species in different life stages and their densities over all sampling sites in the Vasse-Wonnerup wetland between 2017-2020.**

Species	Summed abundance over all 0.1 m <sup>2</sup> samples	Number of samples	Total density (number m <sup>2</sup> per sampling period summed over all sampled sites)
<i>Capitella capitata</i>	162229.4	352	4608.79
<i>Potamopyrgus</i> sp.	24884.03	352	706.93
<i>Procladius</i> sp.	11110.96	352	315.65
<i>Mytilocypris tasmanica chapmani</i>	10248.68	352	291.16
<i>Chironominae</i> sp. 1	8195.34	352	232.82
<i>Austrochiltonia subtenuis</i>	6177.68	352	175.5
Oligochaete spp.	2382.16	352	67.67
<i>Mytilocypris ambiguosa</i>	1870.98	352	53.15
<i>Harpacticoida</i> spp.	1511.09	352	42.93
<i>Chironominae</i> spp.	1333.3	352	37.88
<i>Chironominae occidentalis</i>	1026.61	352	29.17
<i>Berosus</i> sp. 1 - larvae	964.25	352	27.39
<i>Perthia</i> sp.	724.36	352	20.58
<i>Tanypodinae</i> sp. 1	595.49	352	16.92
<i>Alboa worooa</i>	484.31	352	13.76
<i>Chironominae</i> sp. - pupa	475.52	352	13.51
<i>Cyclopoida</i> spp.	462.19	352	13.13
<i>Lepidoptera</i> spp. - larvae	391.08	352	11.11
<i>Berosus</i> sp. 2 adult	355.53	352	10.1
<i>Simplisetia aequisetis</i>	324.4	352	9.22
<i>Culicidae</i> spp. - larvae	319.97	352	9.09
<i>Dytiscidae</i> sp. 1	266.64	352	7.57
<i>Hydrochus</i> sp. 1 - adult	262.21	352	7.45
Nematode spp.	257.77	352	7.32
<i>Ceratopogonidae</i> sp.	217.75	352	6.19
<i>Arthritica semen</i>	204.43	352	5.81
<i>Hirudinea</i> spp.	177.77	352	5.05
Diptera spp. - pupa	173.28	352	4.92
<i>Haliphus</i> sp.	164.4	352	4.67
<i>Calanoida</i> spp.	159.99	352	4.55
<i>Grandierella propodentata</i>	137.78	352	3.91
<i>Canonocypris novazelandae</i>	124.41	352	3.53
<i>Megaporus</i> sp. 1	111.07	352	3.16
<i>Barnardomelita matilda</i>	88.89	352	2.53
Formicidae spp.	88.88	352	2.52
Orthocladiinae sp.	88.86	352	2.52
Ostracoda sp. 1	88.86	352	2.52
<i>Coxiella striatula</i>	79.94	352	2.27

Table continued...

Hydrochus sp. 2 - adult	75.54	352	2.15
Tipulidae sp. - larvae	75.54	352	2.15
Diptera spp. - adult	71.11	352	2.02
Hydrophilidae sp. 1 - larvae	71.08	352	2.02
Hydrophilidae sp. 1	62.22	352	1.77
Coleoptera - pupa	62.21	352	1.77
Daphniidae sp. 1	53.33	352	1.52
<i>Micronecta robusta</i>	53.3	352	1.51
<i>Corophium minor</i>	48.86	352	1.39
Berosus sp. 2 - larvae	44.44	352	1.26
<i>Pseudopolydora kempfi</i>	44.43	352	1.26
Culicidae - adult	44.42	352	1.26
Diacypsis spinosa	31.1	352	0.88
Nymphulinae sp. 1	31.1	352	0.88
<i>Chironominae alternans</i>	26.66	352	0.76
Dytiscidae sp. 2	22.21	352	0.63
Hydrophilidae - adult	22.21	352	0.63
Stratiomyidae sp.	22.21	352	0.63
Zygoptera sp. 1	17.78	352	0.51
Anisoptera sp. 1 - larvae	13.33	352	0.38
Culicidae sp. 1 - larvae	13.33	352	0.38
Enchytraeidae sp.	13.33	352	0.38
<i>Paracorophium excavatum</i>	13.33	352	0.38
Physa sp.	13.33	352	0.38
Trichoptera sp. 2 - larvae	13.33	352	0.38
Hydrophilidae sp. 3 - larvae	13.32	352	0.38
<i>Boccardiella limnicola</i>	8.89	352	0.25
Curculionoidea sp.	8.89	352	0.25
Hydrophilidae sp. 2	8.89	352	0.25
Phyllodoce spp.	8.89	352	0.25
<i>Xenostrobis securis</i>	8.89	352	0.25
Agraptocorixa sp. 1	8.88	352	0.25
Cirolanidae sp.	8.88	352	0.25
Corixidae sp. 1	8.88	352	0.25
<i>Heteromastus filiformis</i>	8.88	352	0.25
Hydrophilidae sp. 1 - larvae 1	8.88	352	0.25
Noteridae	8.88	352	0.25
Notonectidae sp. 1	8.88	352	0.25
<i>Palaemonetes australis</i>	8.88	352	0.25
Anthophila sp.	4.44	352	0.13
<i>Bennelongia australis</i>	4.44	352	0.13
Coccinellidae sp.	4.44	352	0.13
Culicidae sp. 1	4.44	352	0.13
Daphniidae sp. 2	4.44	352	0.13

**Table continued...**

Desdemona ornata	4.44	352	0.13
Hyphydrus elegans - adult	4.44	352	0.13
Necterosoma sp.	4.44	352	0.13
Nemertea	4.44	352	0.13
Nephilidae sp.	4.44	352	0.13
Nymphulinae sp. 2	4.44	352	0.13
Piona sp.	4.44	352	0.13
<i>Pseudogobius olorum</i>	4.44	352	0.13
<i>Scoloplos normalis</i>	4.44	352	0.13
Tabanidae - pupa	4.44	352	0.13
<i>Tanais dulongii</i>	4.44	352	0.13
Tipulidae sp. - adult	4.44	352	0.13

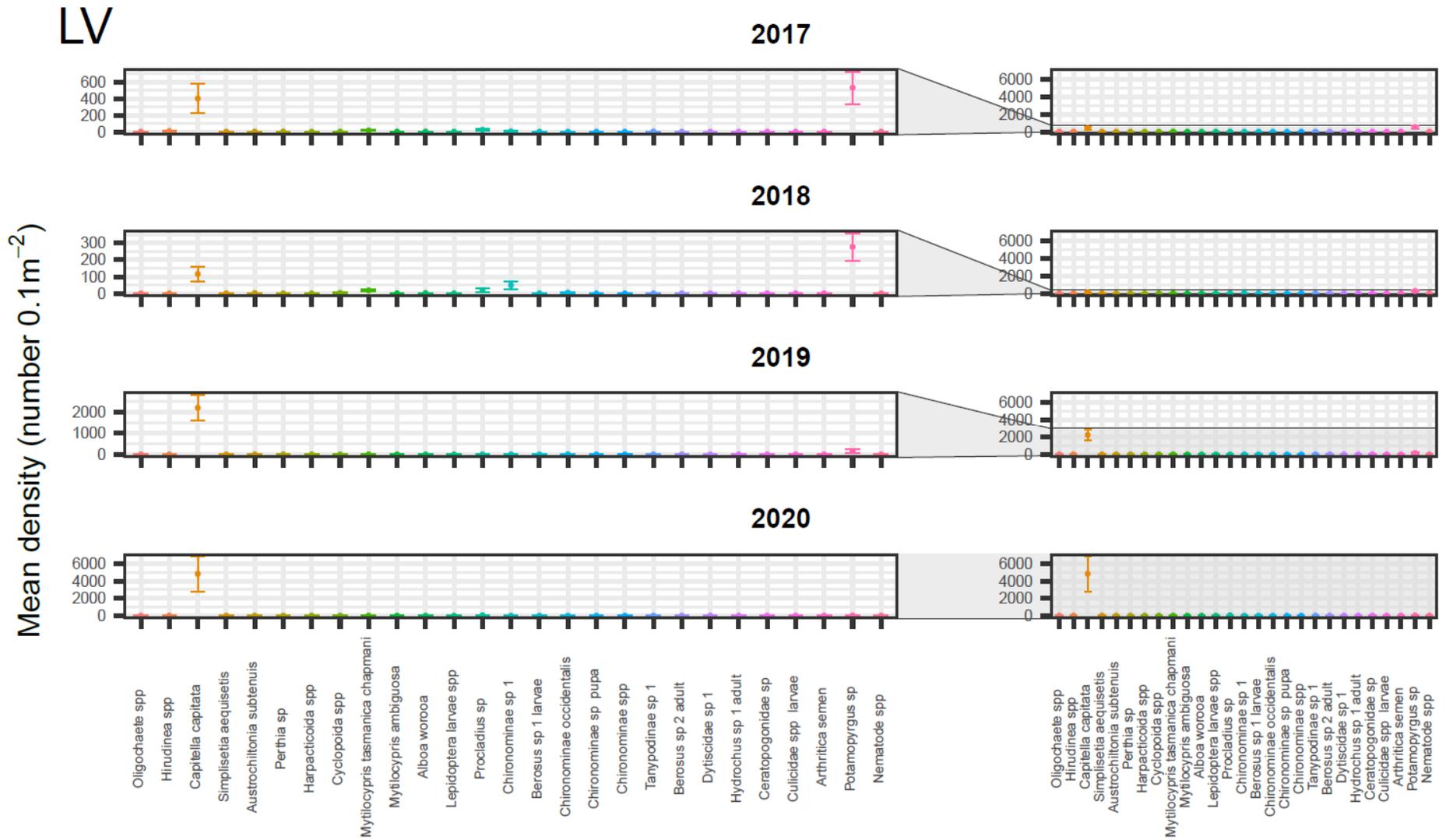


Figure 39. Mean number of fish per  $0.1\text{m}^2$  ( $\pm$  SE) per year sampled in the Lower Vasse between 2017-2020 for species encountered more than 5 times during the study.

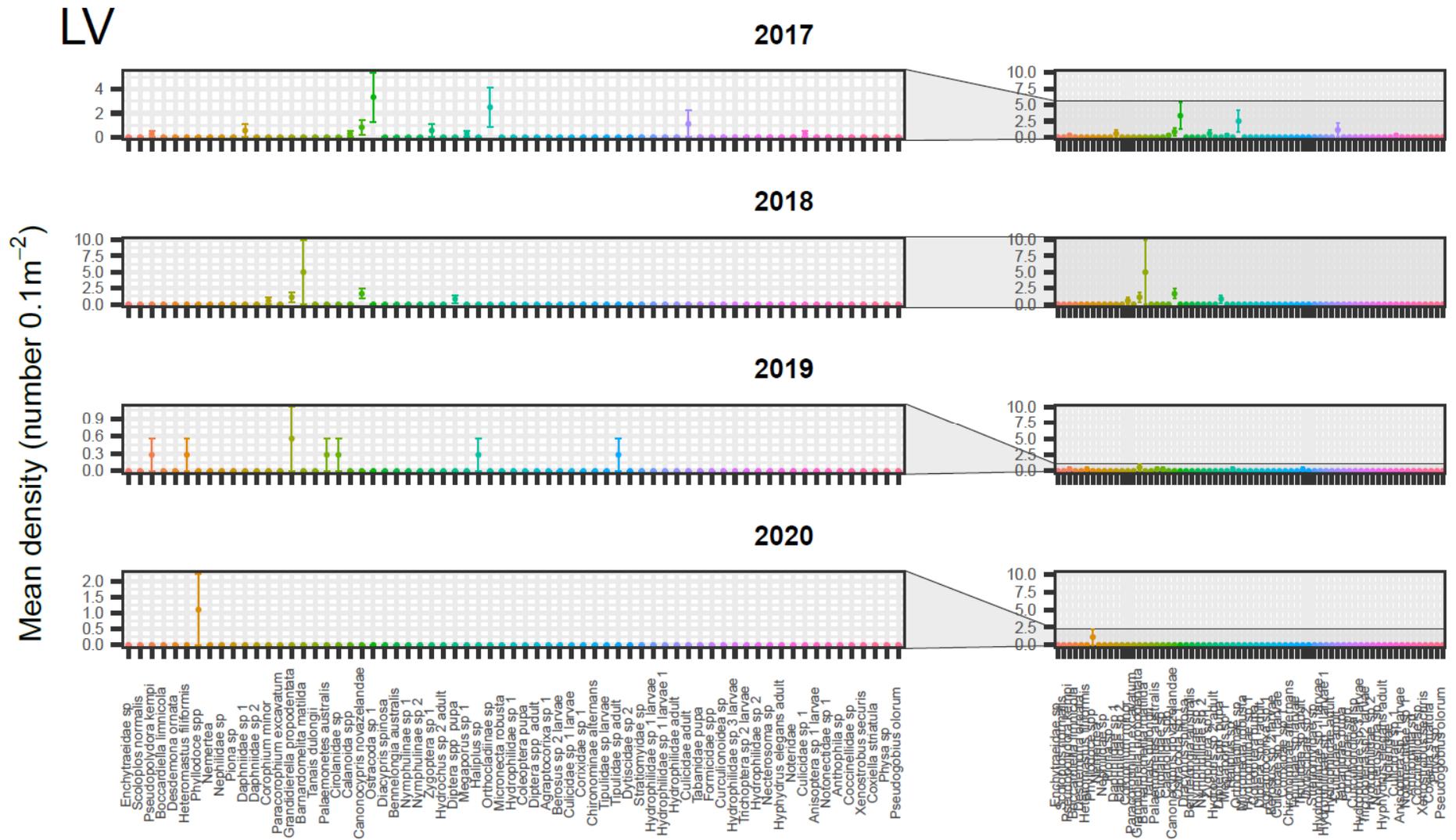


Figure 40. Mean number of fish per 0.1m<sup>2</sup> ( $\pm$  SE) per year sampled in the Lower Vasse between 2017-2020 for species encountered 5 or fewer times during the study.

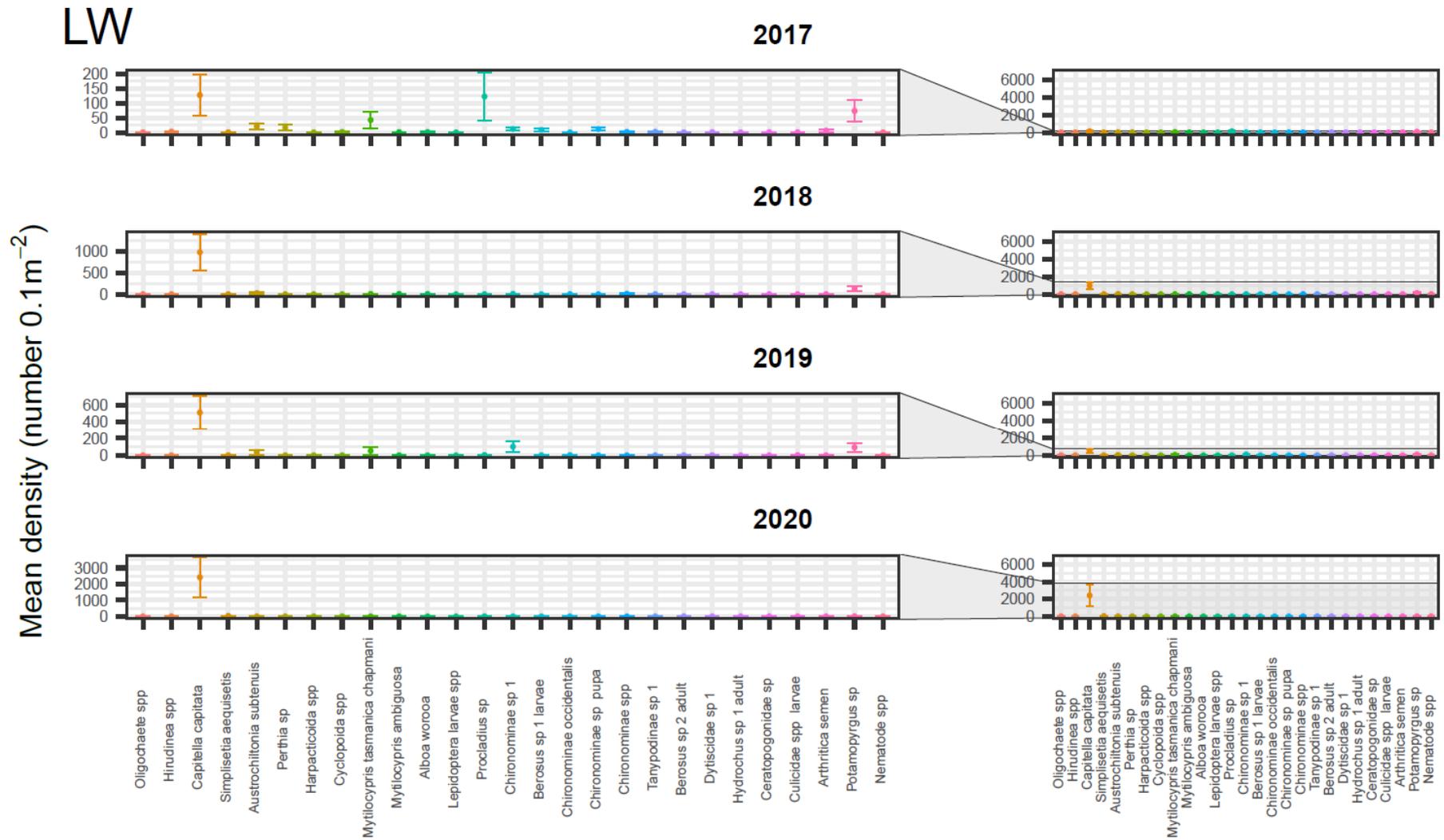


Figure 41. Mean number of fish per  $0.1\text{m}^2$  ( $\pm$  SE) per year sampled in the Lower Wonnerup between 2017-2020 for species encountered more than 5 times during the study.

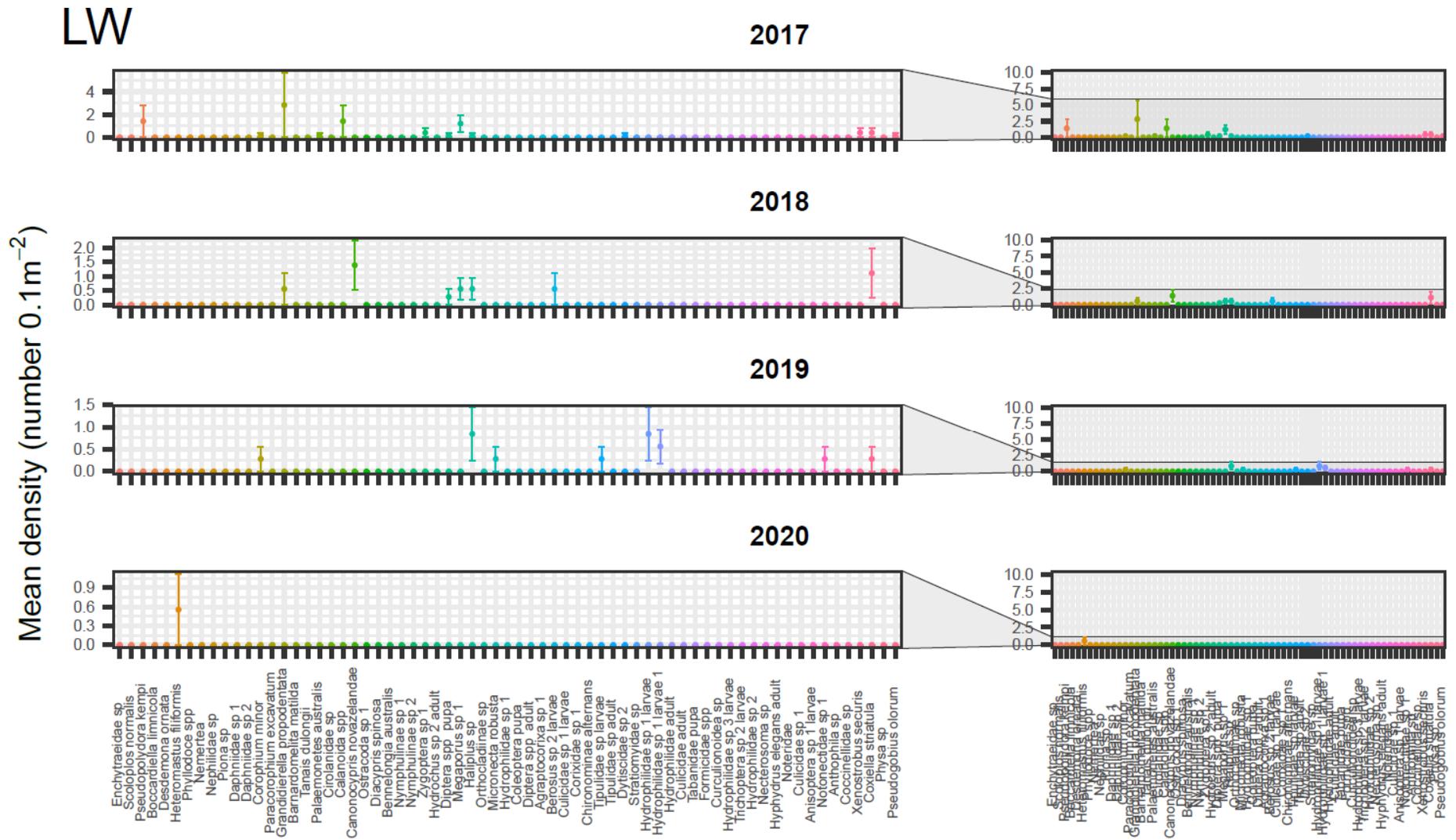


Figure 42. Mean number of fish per 0.1m<sup>2</sup> ( $\pm$  SE) per year sampled in the Lower Wonerup between 2017-2020 for species encountered 5 times or fewer during the study.

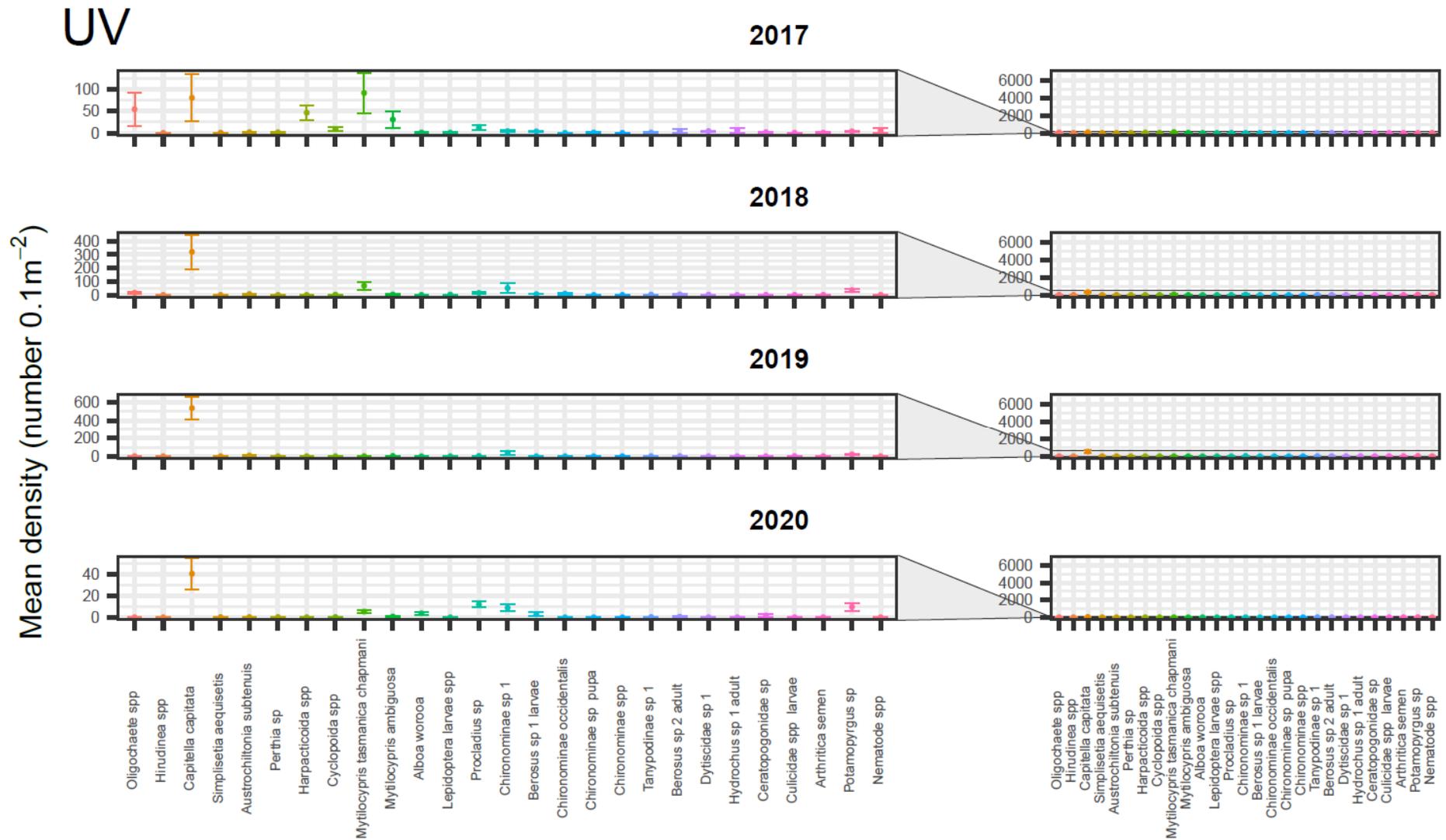


Figure 43. Mean number of fish per 0.1m<sup>2</sup> ( $\pm$  SE) per year sampled in the Upper Vasse between 2017-2020 for species encountered more than 5 times during the study.

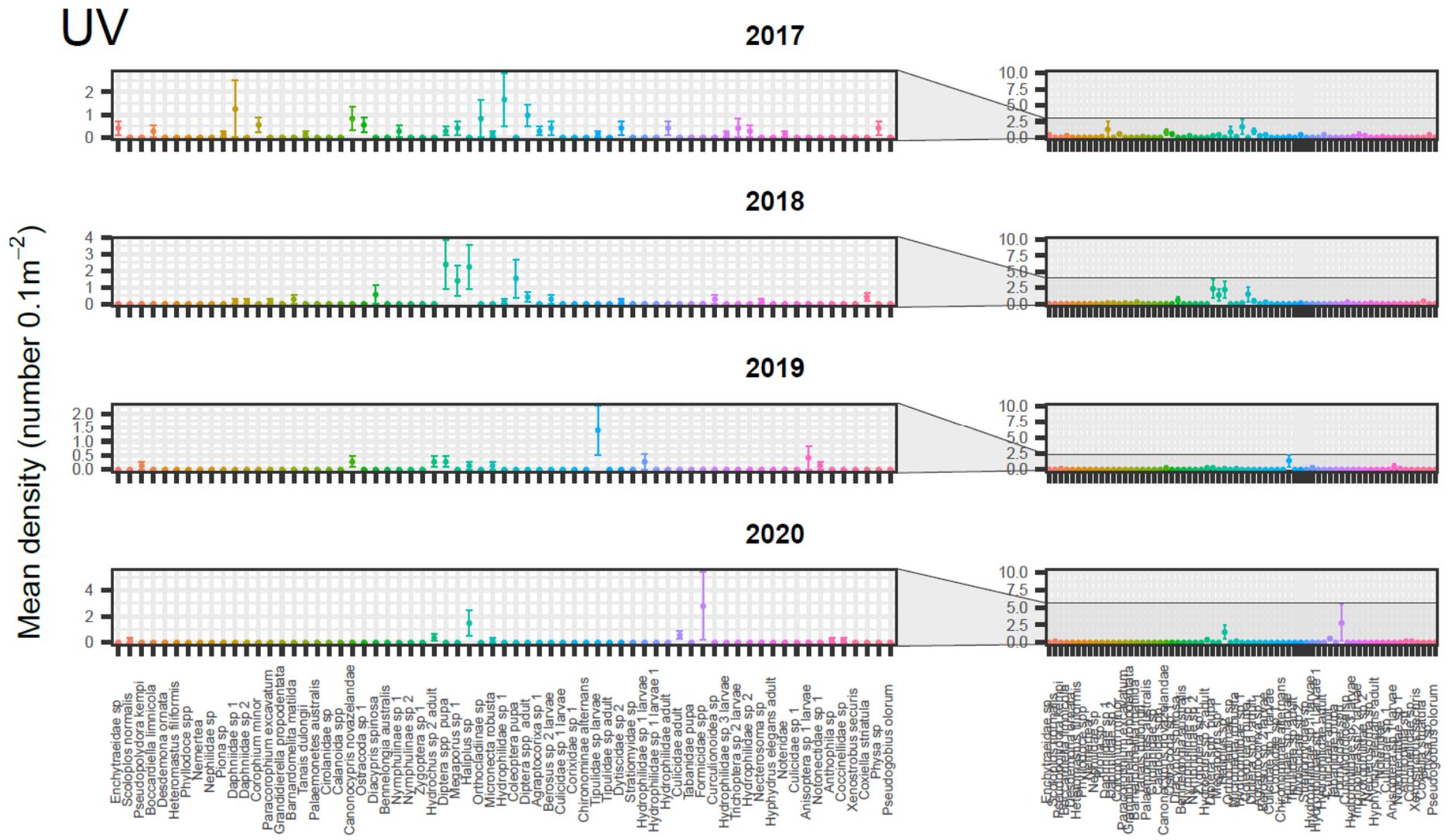


Figure 44. Mean number of fish per 0.1m<sup>2</sup> (± SE) per year sampled in the Upper Vasse between 2017-2020 for species encountered 5 times or fewer during the study.

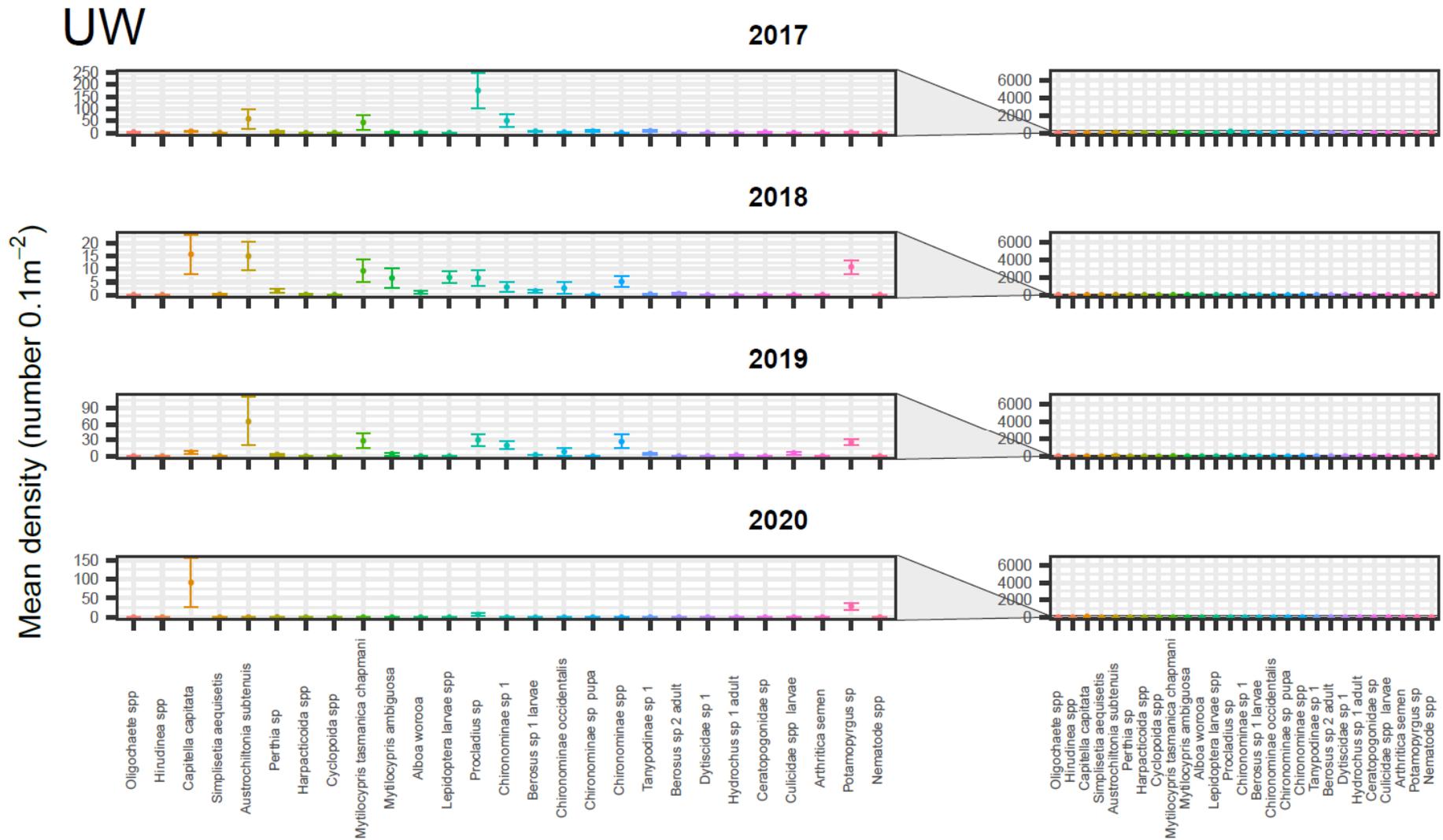


Figure 45. Mean number of fish per  $0.1\text{m}^2$  ( $\pm$  SE) per year sampled in the Upper Wonerup between 2017-2020 for species encountered more than 5 times during the study.

UW

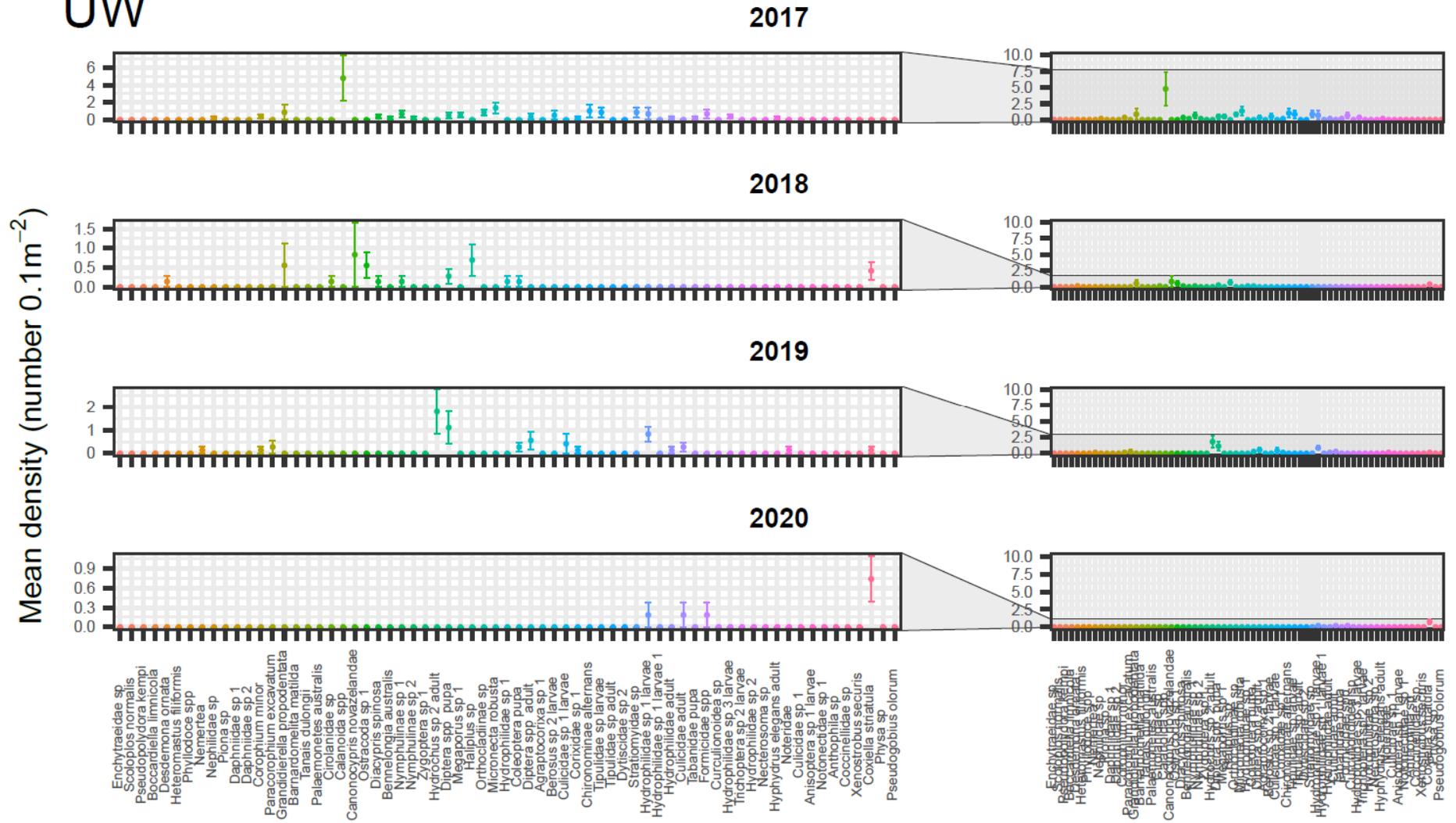


Figure 46. Mean number of fish per 0.1m<sup>2</sup> (± SE) per year sampled in the Upper Wonerup between 2017-2020 for species encountered 5 times or fewer during the study.



Figure 47. Collinearity plots of log invertebrate species counts per sample in the Vasse-Wonnerup wetland between 2017-2020. Bivariate scatter plots with correlation ellipses are below the diagonal, histograms on the diagonal, and Pearson correlations above the diagonal.

## Appendix E. Macrophytes

**Table 8. Total macrophyte percent volume index (PVI) over all samples (replicate by site by region) and sample size in the Vasse-Wonnerup wetland between 2017-2020.**

Macrophyte grouping	Total PVI	Sample size	Total PVI per sampling period across all sampled sites
<i>Ruppia sp.</i>	7310.61	1128	6.48
<i>Lamprothamnium</i>	5323.78	1128	4.72
<i>Ruppia polycarpa</i>	4438.32	1048	4.24
<i>Stuckenia</i>	2887.03	1128	2.56
<i>Ulva sp.</i>	2854.74	1128	2.53
<i>Ruppia megacarpa</i>	2407.10	1048	2.30
<i>Cladophora</i>	2173.11	1128	1.93
<i>Rhizoclonium</i>	1167.59	1128	1.04
<i>Althenia</i>	1133.60	1128	1.00

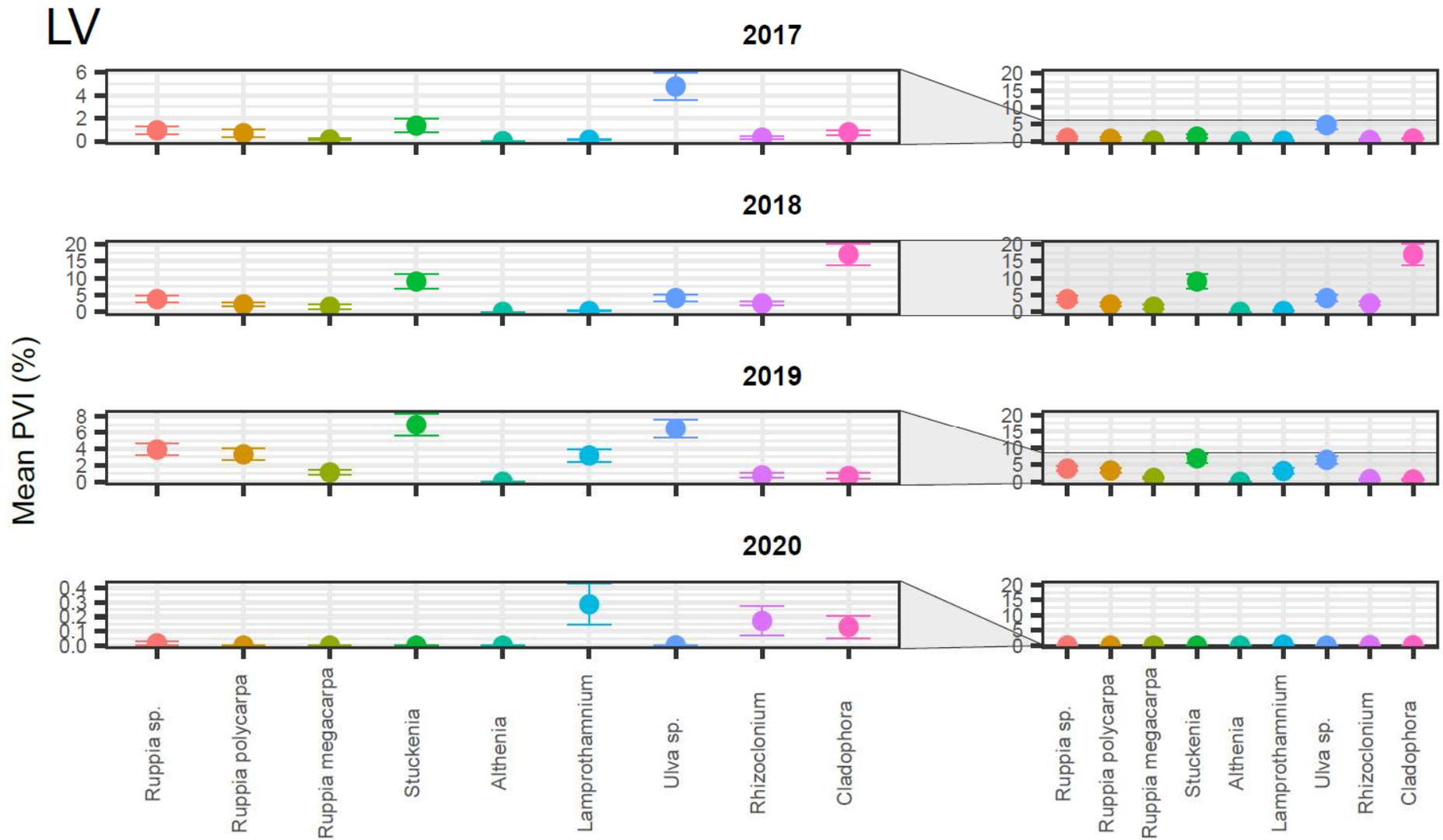


Figure 48. Mean macrophytes percent volume index (PVI) ( $\pm$  SE) per year sampled in the Lower Vasse between 2017-2020.

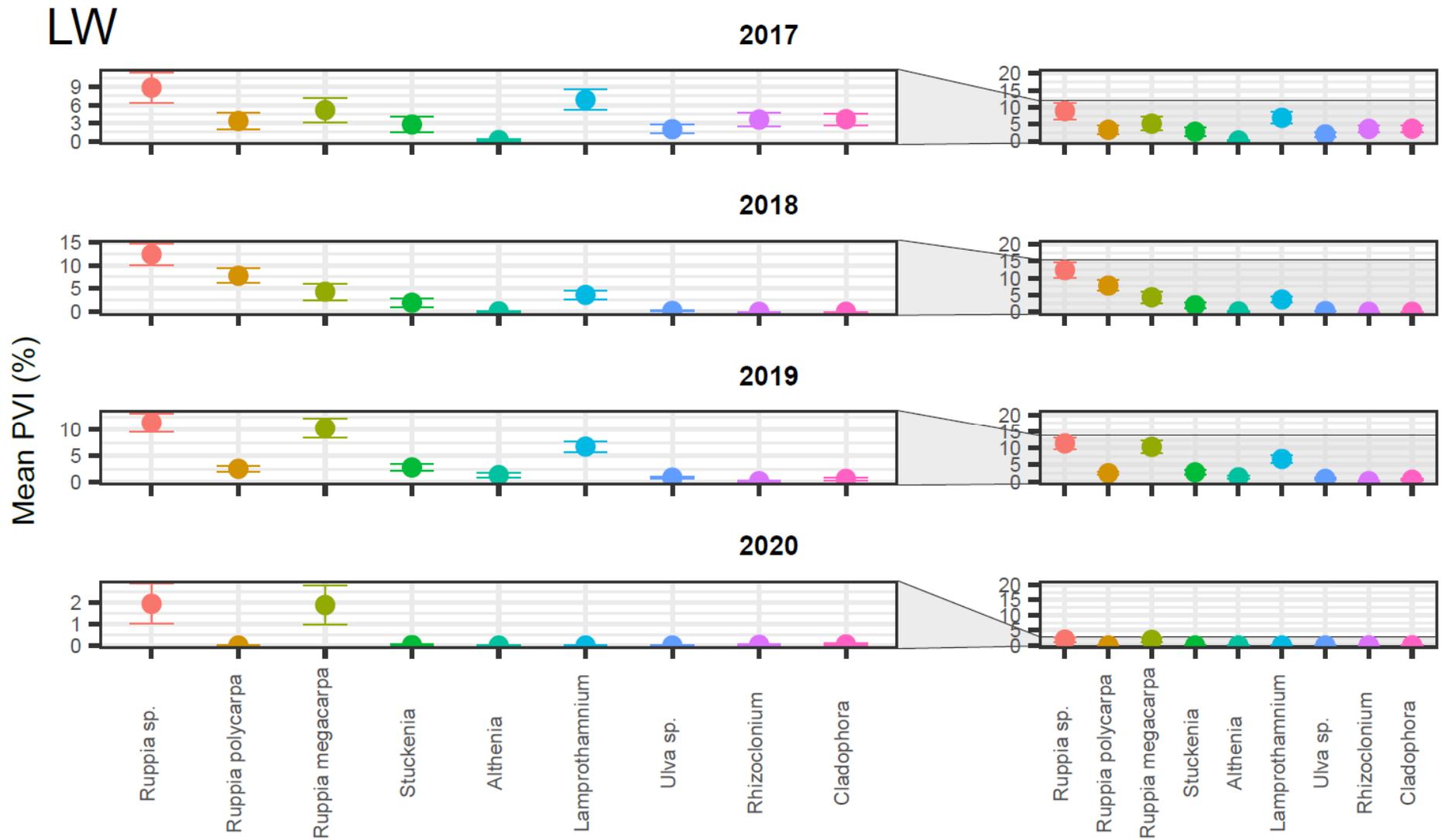


Figure 49. Mean macrophytes percent volume index (PVI) ( $\pm$  SE) per year sampled in the Lower Wonerup between 2017-2020.

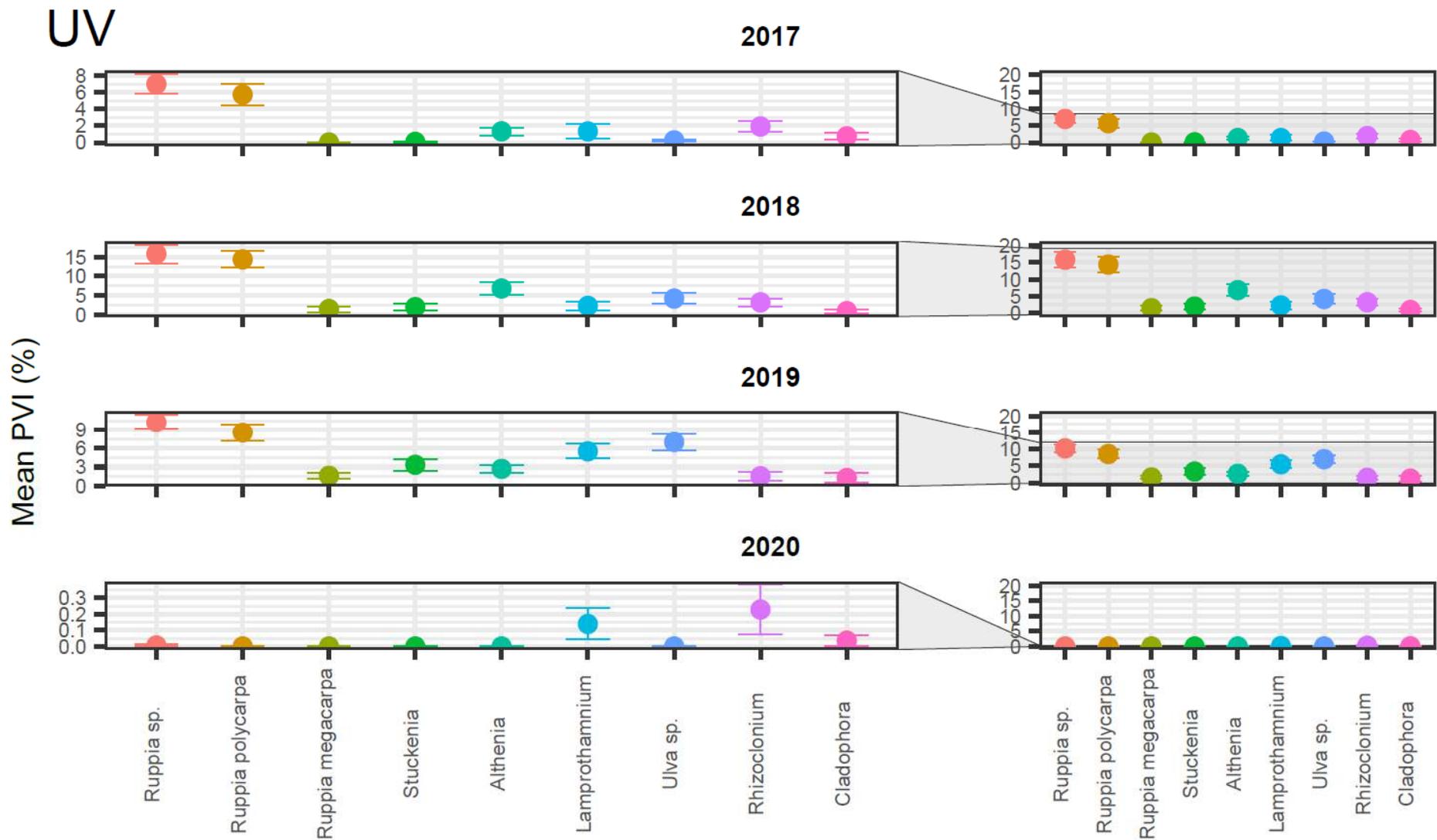


Figure 50. Mean macrophytes percent volume index (PVI) ( $\pm$  SE) per year sampled in the Upper Vasse between 2017-2020.

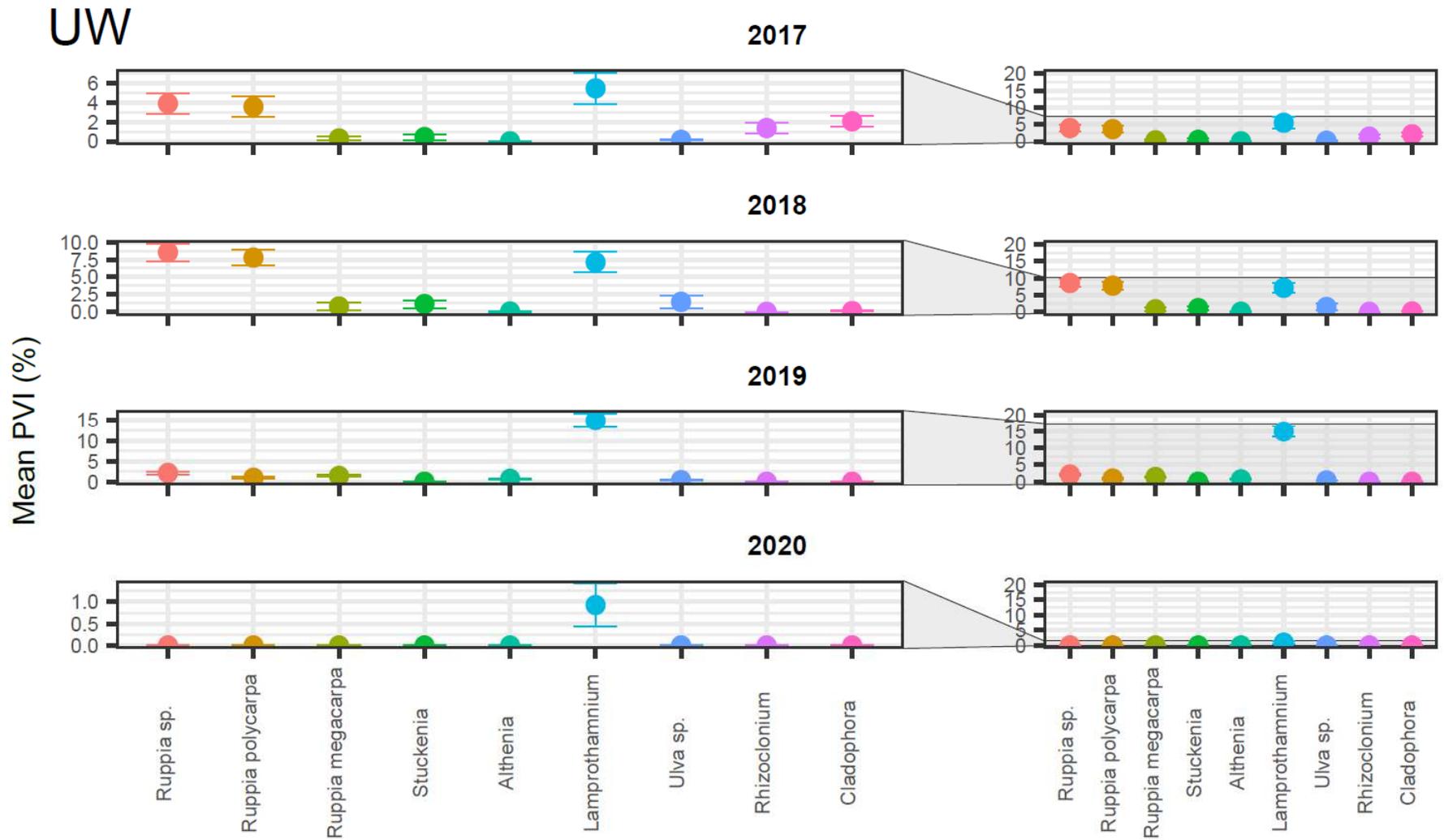


Figure 51. Mean macrophytes percent volume index (PVI) ( $\pm$  SE) per year sampled in the Upper Wonneerup between 2017-2020.

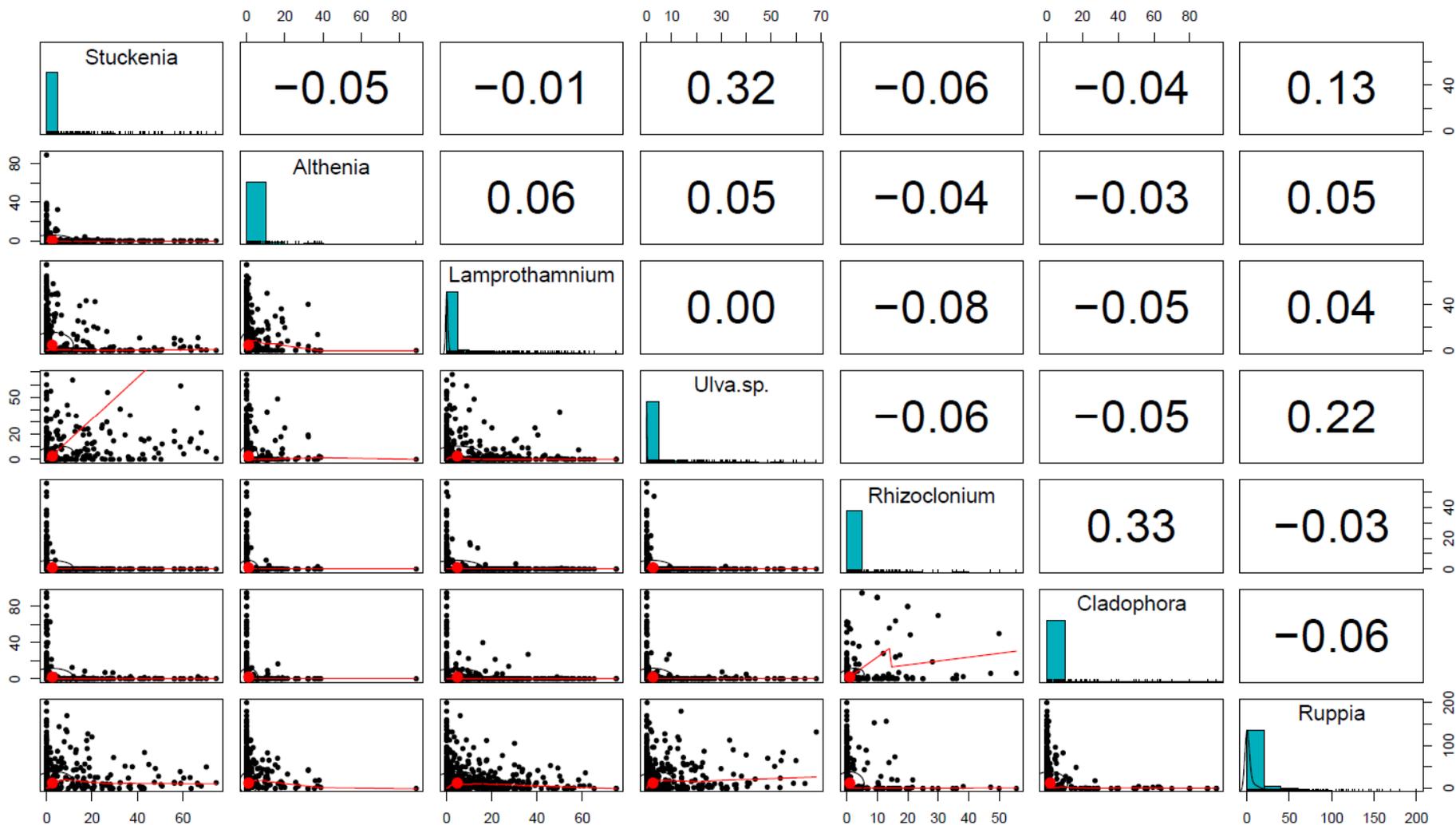


Figure 52. Collinearity plots of invertebrate species counts per sample in the Vasse-Wonnerup wetland between 2017-2020. Bivariate scatter plots with correlation ellipses are below the diagonal, histograms on the diagonal, and Pearson correlations above the diagonal.

Appendix F. Abiotic and biotic variables identified as important in predicting macrophytes, benthic macroinvertebrates or fish that were important for waterbirds

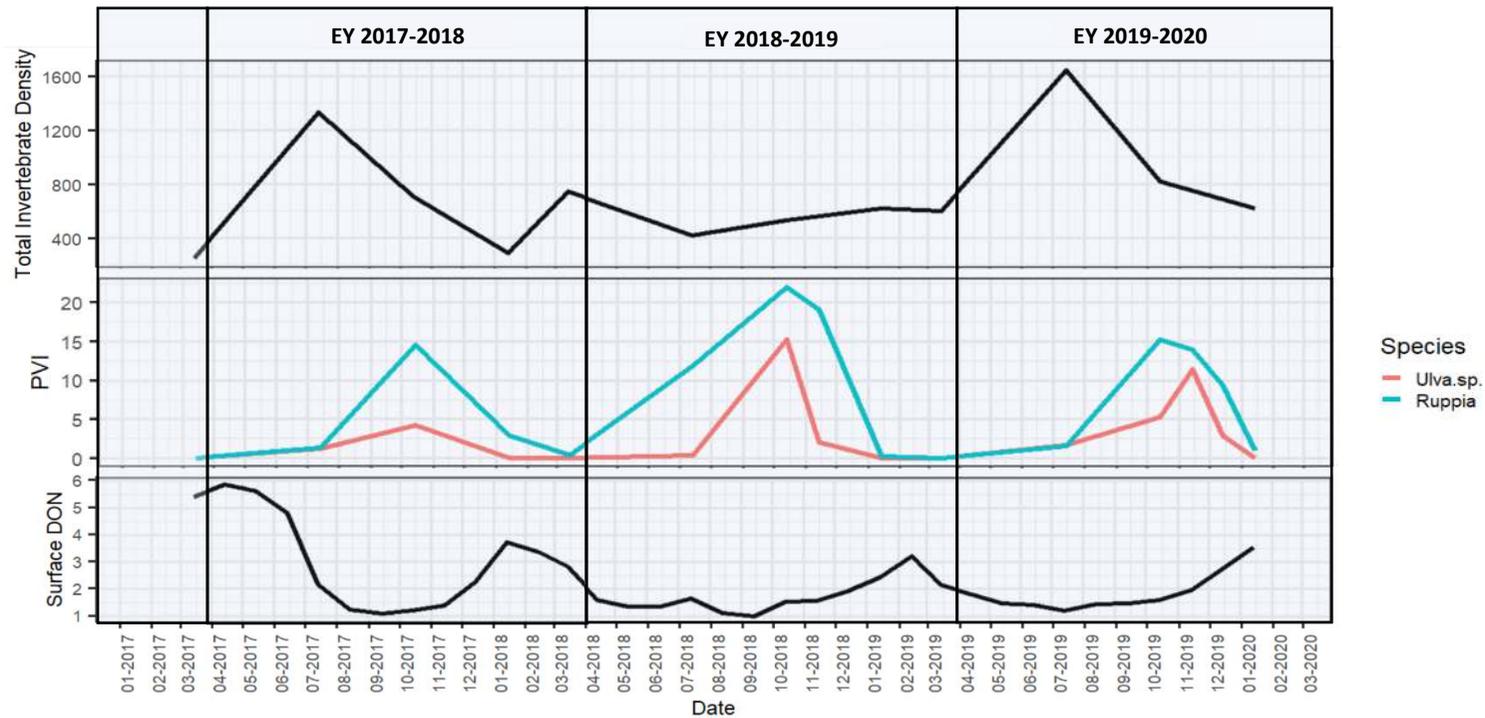


Figure 53. Variables (average values) that were significantly associated with biotic groups, that were in turn associated with waterbirds, over the period of the study. Gate management is illustrated in the bottom panel.

