Assessing the utility of the Melbourne Water fish database for detecting long-term population trends

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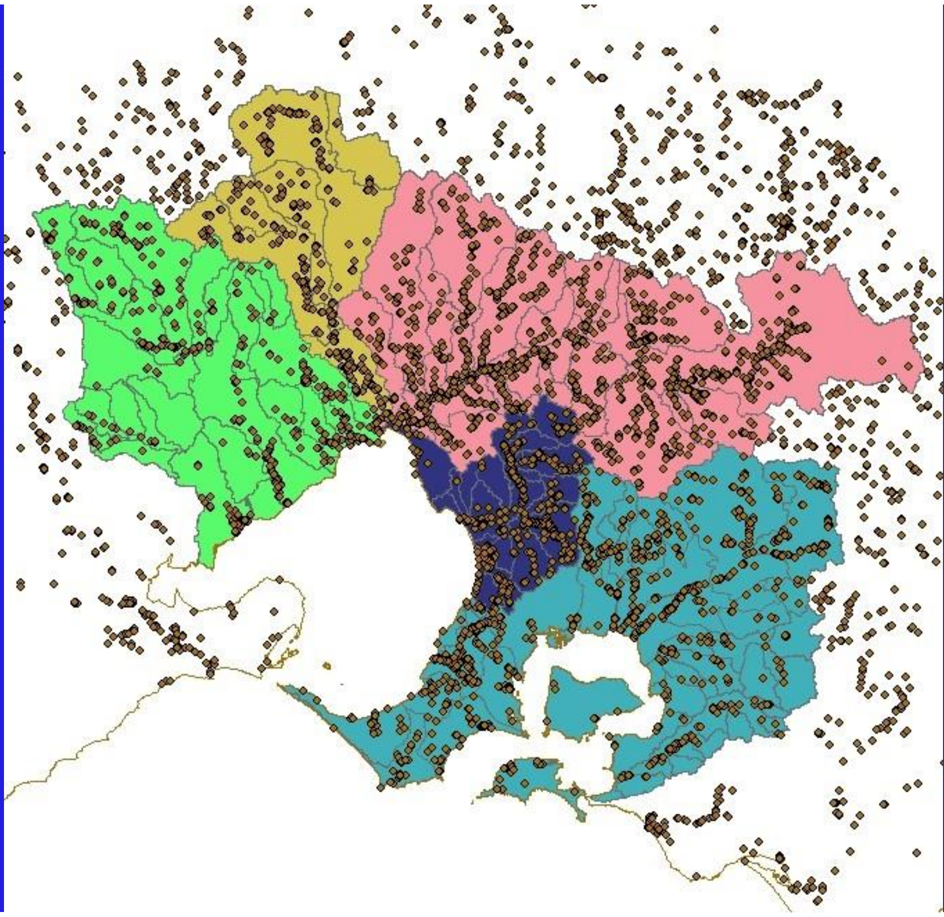

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Executive summary

Melbourne Water have recently collated all known datasets related to fish distribution across the Melbourne Water Management Region (MWMR) into a single database. The dataset included thousands of species observations gathered between 1900 and 2022. To varying degrees these observations were accompanied by fish abundance, length and weight data. Melbourne Water were interested in using the dataset to assess the effectiveness of their waterway management efforts over time, so they engaged Jacobs to investigate the database and see if metrics could be developed for assessing long-term trends in the population health of freshwater fish species. Following consultation between Melbourne Water and Jacobs, three key questions were decided upon to be answered from the data, if possible:

1. How has species richness and nativeness changed over time at the sub-catchment and catchment scale? What could be driving these trajectories?

2. How has populations health changing over time for priority species highlighted in the Monitoring Evaluation Reporting and Improvement (MERI) framework – with particular attention to trends in recent years. What could be driving these trajectories?

3. Is there any evidence of range expansions or contractions over time using all data?

Methods

We first conducted a review of the dataset focussing on the temporal and spatial distribution of species records. Then, to answer the study questions we:

1. Plotted the proportion of native versus exotic species (by species richness) through time;
2. assessed the availability of abundance (catch-per-unit-effort), length, and weight data for developing population health metrics; and
3. developed multi-season occupancy models for a range of species to assess changes in their distribution through time.

Data review

We found that data gathered since 1980 was the most useful for inter-annual analysis as surveys of the region became regular and more widespread across river basins from that time. That said, some river basins received considerably more sampling effort than others. The main findings were that:

- The Yarra River Basin had longest and most consistent annual survey record in the MWMR, and the largest mean number of sites sampled in each year (44), which were fairly well spread across the river basins18 sub-catchments.
- Western Port, which had a near continuous annual survey record and the second largest mean number of sampled sites per year (26), where were fairly well spread across the river basins primary catchments.
- The Werribee, Maribyrnong, and Dandenong basins received far more sparse sampling than the Yarra and Western Port basins. For the purpose of spatial analysis, these catchments may be divided into broader spatial units other than primary or sub-catchments such as lower, mid, and upper catchment zones to help overcome the limited spatial distribution of the data.

Changes in nativeness (based on species richness) through time

- Exotic species have had a long history in the MWMR, with a number of European sports fishes being present from at least the early 1900s.
- Most of the current exotic fauna being in place by 1983, around the time that regular fish monitoring started in most MWMR river basins.
- As fish community data prior to the exotic fish introductions is rare, it is difficult to identify changes before and after the introductions based on species richness metrics.

- There is not apparent decline in the proportion of native species, at the river basin scale, over the study period.
- Nativeness scores were the lowest on average in the Dandenong Creek basin and highest in the Western Port basin. This likely reflects the intensity of development across these catchments and proximity to Melbourne's main urban centres.

A nativeness metric based on species abundance would be a more sensitive and informative approach to assessing change in fish communities over time. Sufficient abundance data may exist in the dataset, over the last two decades in particular, but further interrogation of the dataset is required.

Assessing changes in community health through time

Commonly applied metrics for monitoring fish population health require data detailing the abundance (standardised by fishing effort), length, and weight of fish. This information can be used to infer individual condition, recruitment success, and the trajectory of population size (i.e. increasing or decreasing). Following investigation of the dataset our key findings were:

- Length and weight data for individual fish are generally lacking, so changes in population health through time cannot be assessed using length-weight metrics.
- While species abundance data was regularly recorded in the database, fishing effort was not, so catch-per-unit-effort could not be calculated and changes in relative abundance between years could not be assessed.
- Fish survey data needs to be entered into the Victorian Biodiversity Atlas (VBA; the major source of data in this dataset) in a particular format for it to be uploaded into the VBA system. A lack of clarity about the correct format and perhaps increased difficulty in following it, has meant that this data is not available here, even though it would have been collected as part of most fish surveys.

Incorporating this data into the MW fish database (and VBA) would greatly aid in the assessment of fish community health through time and would thus help inform effective management of those communities. We present options to help overcome the current data entry issues in our recommendations.

Assessing changes in species distribution through time

Our analysis of changes in species occupancy in the Yarra River Basin between 1973 and 2022 indicated three main trends in species range size:

1. The range size of native obligate freshwater species is in equilibrium or is contracting.
2. The range size of native migratory species is in equilibrium or is expanding.
3. The range size of more recently introduced exotic species has been expanding but has reached or is close to reaching equilibrium.

Most of the observed trends were not statistically significant, owing in part to large confidence intervals around occupancy estimates. Such wide confidence intervals are unavoidable given the nature of the dataset which is made up of spatially and temporally sporadic species observations. Regardless, the trends are consistent over almost five decades and meet a-priori expectations, suggesting that they are likely reflective of actual changes in species occupancy.

Conclusions

- We show that while the MW fish database is imperfect, it can be utilised with great effect to retrospectively assess the effectiveness of management efforts.
- Occupancy modelling offers an effective method for utilizing historic presence and absence data for species for which little other data exist. The insight the analysis provides into large-scale and long-term population changes can provide a strong indication of weather river restoration or conservation efforts are working.
- The positive trajectory observed among several of the migratory species is reflective of positive change, but more needs to be done to arrest the decline of non-migratory species in the Yarra River Basin.

- The analysis presented here should be considered preliminary as it only aims to describe trends in occupancy, rather than attempting to determine biotic (e.g. exotic species) and abiotic (e.g. stream flow metrics) drivers of those trends. It is also focussed on a single river basin.

Recommendations

- Years with missing survey data impact significantly on the sensitivity of the occupancy models and confidence in the model estimates, reducing the utility of the dataset. We strongly recommend that a base level of monitoring be maintained by MW to improve the veracity estimates in the future.

- The analysis presented here aims to describe trends in occupancy, rather than attempting to determine biotic (e.g. exotic species) and abiotic (e.g. stream flow metrics) drivers of those trends. We recommend that the analysis be extended so that it incorporates co-variates (e.g. environmental flow releases and exotic species presence/richness) that might explain the observed pattens. This would help to focus management efforts appropriately.

- The analysis presented here uses the Yarra River Basin as a test case as it was the most data rich river basin. We recommend that the analysis be broadened across the whole MWMR so that large-scale trends can be assessed. This would be particularly important for assessing the lowland migratory species, such as Australian grayling, for which river basin scale occupancy trends are more difficult to detect than regional scale trends.

- Data detailing species catch-per-unit-effort, length, and weight are regularly collected as part of fish surveys but are mostly not included in the database. Such data would provide a critical resource for catchment managers trying to monitor the health of populations through time. We recommend taking steps to better facilitate the upload of this data into the MW and VBA databases.

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1. Introduction

Melbourne Water have recently collated all known datasets related to fish distribution across the Melbourne Water Management Region (MWMR). Critically, within the dataset, species observations were accompanied by locations and dates allowing for temporal and spatial analysis. Some length and weight data are also present, which may facilitate an assessment of population health through time, but the amount and consistency of recording requires further investigation. There is the potential to derive metrics describing the health of aquatic species and to describe how they have changed over time. Such analyses would improve inferences from more recent monitoring programs such as the Monitoring Evaluation Reporting and Improvement (MERI) framework (King *et al.* 2020), and may present a way of assessing the efficacy of management initiatives such as the Health Waterways Strategy (Melbourne Water 2018).

Melbourne Water engaged Jacobs to investigate the dataset and summarise what data was present regarding the spatial and temporal distribution of records. Following consultation between Melbourne Water and Jacobs, three key questions were decided upon to be answered from the data, if possible:

1. How has species richness and nativeness changed over time at the sub-catchment and catchment scale? What could be driving these trajectories?

2. How has populations health changing over time for priority species highlighted in the Monitoring Evaluation Reporting and Improvement (MERI) framework – with particular attention to trends in recent years. What could be driving these trajectories?

3. Is there any evidence of range expansions or contractions over time using all data?

The aim of this project was to investigate the Melbourne Water fish dataset, determine whether the key questions could be answered in a meaningful way, and to answer those that can be.

2. Methods

2.1 Dataset exploration

The dataset provided by Melbourne Water was provided as a spatial layer containing point localities for each species detection ArcGIS. This layer was spatially joined to polygon layers of (1) each of the MWMR river basins (i.e. Werribee River, Yarra River, Maribyrnong River, Dandenong Creek, and Western Port basins), and (2) each of the primary and major sub-catchments withing these management areas. We discuss the results of our analysis in terms of these spatial units (i.e. River basin → Primary catchment → Major sub-catchment).

2.2 Nativeness dataset

Species were assigned as being either native or exotic so that the proportion of native versus exotic species could be explored within catchments, between years. As the number of sites surveyed in some years were very low, proportional results could be misleading. For instance, if only two sites were sampled in the Yarra River in 1960 and only native species were found, it would appear that the whole fish community was made up of native species when in reality this is likely an artifact of low replication. To account for this, we filtered out years where there were less than five sites were surveyed within a catchment.

At the sub-catchment level, there was little or no replication of surveys within years. Furthermore, many years had missing data leading to a very inconsistent record through time. As such, we conducted our analysis at the river basin level only.

2.3 Length-Weight dataset

We summarised the recording of length-weight data over the last decade. Ultimately, we found that there was insufficient length and weight records to conduct meaningful analysis of trends through time, so no analysis was undertaken.

2.4 Temporal trends in occupancy estimates

2.4.1 An introduction to site occupancy models

Site-occupancy models detection/non-detection data are commonly called ‘presence/ absence data’, although in reality these data come from two nested, stochastic processes, one determining the true state of the site (present/absent) and the other governing the observation of the site state (detected/not detected). The first process generates the true distribution of a species for which there are two possible states for each site (present/absent). However, the result of this process is not what we actually observe. Inevitably, a species may be undetected at a site even when it is present, thus the detection probability is less than one.

Without extra data, conventional species distribution models cannot tease apart true occurrence from the detection probability. The extra information required to partition detection/non-detection data into these two components (true occurrence and detection) comes from repeat surveys at some or all sites. The pattern of detection and non-detection in temporal replicates provides the information about detection probability that allows one to estimate true occurrence, accounting for imperfect detection.

Site-occupancy models account for the ambiguity about true occurrence that is introduced by imperfect detection (MacKenzie et al., 2002, 2003, 2006). With replicate surveys, detection/non-detection data can be formatted into detection histories, such as 0-0-1 for a site where a target species was only detected on the third of three surveys, or 0-0-0 for another site where the species was never detected during three surveys. Assuming a closed population (i.e. a site remains either occupied or unoccupied for the entire duration of the repeated surveys), the probability of observing each such combination of detection and non-detection events can be expressed as a function of parameters for occurrence probability (occupancy, φ) and detection probability (detectability, p), a likelihood can be built and maximum likelihood estimates obtained.

In the case of this dataset, repeat surveys at a given site were not available within most years. To overcome this, to calculate the detection probability parameter of the occupancy models, we grouped the data into 3-year time periods, where each year represented a replicate. Not all rivers were surveyed in each year, but occupancy models are robust to missing dependent data (Mackenzie et al. 2006).

2.4.2 Occupancy model dataset

We converted species catch data into a presence (detection) and absence (non-detection) matrix for six priority species outlined in the MERI, and a further three species that were added out of general interest:

- Common galaxias (*Galaxias maculatus*)
- Eastern gambusia (*Gambusia holbrooki*) – not a priority species
- Oriental weatherloach (*Misgurnus anguillicaudatus*) – not a priority species
- Ornate galaxias (*Galaxias ornatus*)
- Pouched lamprey (*Geotria australis*) – not a priority species
- River blackfish (*Gadopsis marmoratus*)
- Short-finned eel (*Anguilla australis*)
- Southern pygmy perch (*Nannoperca australis*)
- Tupong (*Pseudaphritis urvillii*)

Our initial exploration of the dataset revealed that the most spatially and temporally consistent dataset was that from the Yarra River for the period between 1973 and 2022. Records were sparse before that time in all catchments. Also, the other river basins within the MWMR received less sampling effort than the Yarra, limiting the ability to conduct analysis at finer spatial scales over a continuous period of time. As such, we focussed our analysis on the Yarra River Basin in this case. A single systematic sampling program has not been conducted over the entire study period within the MWMR, and occupancy data came from a wide range of targeted and non-targeted sampling efforts using a variety of methods (but mainly backpack electrofisher). As such, the data was spatially and temporally variable. To deal with this, we aggregated sites by the major sub-catchment they fell into (e.g. Merri Creek catchment). Presence for a given sample year and site was indicated if a species was encountered anywhere within that sub-catchment. To model occupancy dynamics in the Yarra River Basin, we used detection histories for 18 sites (sub-catchments) for the 1973–2022 period. We did not include data for years between 1976 and 1983 as few surveys were conducted during that time, but the dataset otherwise included consistent annual records.

Suitable data existed for each of the priority species outlined in the MERI framework that predominantly reside in freshwater habitat. However, too few observations of Yarra pygmy perch (*Nannoperca obscura*) and Short-headed lamprey (*Mordacia mordax*) to run the analysis and Australian grayling (*Prototroctes maraena*) were only recorded at one 'site', the Yarra River mainstream, limiting the utility of the analysis in this instance. Furthermore, there were very few observations of the estuarine species Estuary perch (*Macquaria colonorum*), Black bream (*Acanthopagrus butcheri*), Pale mangrove goby (*Mugilogobius platynotus*), and Glass goby (*Gobiopterus semivestitus*) to run the occupancy models. For instance, they have not been detected in the last decade. While these species may migrate in and out of the lower reaches of river systems, they are predominantly found in estuarine habitat where there is a general lack of sampling effort.

2.5 Occupancy estimates

We created multi-season occupancy models in the program PRESENCE (ver. 2.13.39; Hines 2006) informed by the detection histories of our target species in the Yarra River Basin. The models can be considered as simultaneous generalized linear models (GLMs) of the sequential detection data, applied to each component of the model, with binomial errors (logistic link). Presence fits the GLM models using maximum likelihood methods. Models best supported by data are selected via an information theoretic framework using Akaike's Information Criterion (AIC) scores, normalised across each candidate set to produce AIC weights (AICw; Burnham and Anderson 2002).

Occupancy, local extinction and invasion at each site were estimated from models using a first-order Markov process wherein the probability of occupancy at a site in time period 2 is contingent on occupancy in time period 1, and:

φ_1 (occupancy) = probability a site is occupied in time period 1

ε_t (extinction) = probability a site becomes unoccupied between time periods t and t

γ_t (invasion) = probability a site becomes occupied between time periods t and t

$p_{t,j}$ (probability of detection) = probability that the target species is detected at a site in year j of time period t (given presence)

We ran competing models (16 in total) with different parameterizations to estimate occupancy, invasion, and extinction of target species within the Yarra River Basin, following MacKenzie et al. (2003). We then assessed the weight of evidence that (1) occupancy was constant or varied among time periods, (2) probability of detection was constant or varied among years, and (3) probability of extinction and invasion were either constant or varied between time periods. In this way, we tested for differences in occupancy among time periods, detectability among time periods, and differences in invasion and extinction rates among time periods.

For this analysis, occupancy (φ_t) refers to a point in time that was a 3-year time period (as discussed above). Model construction focused on our main objective, which was test the null hypothesis that occupancy did not vary between time periods.

3. Results

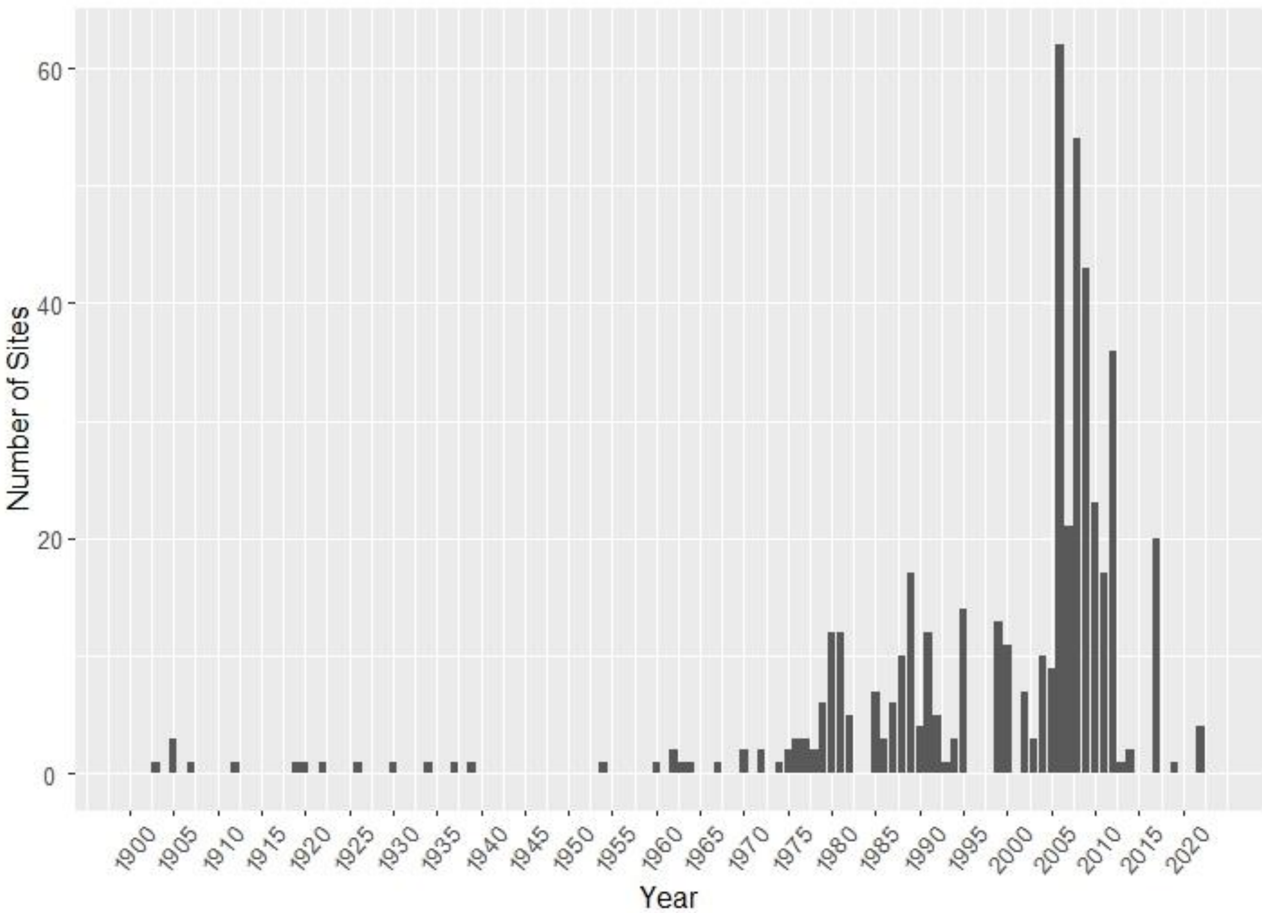
3.1 Dataset

The complete dataset was made up of 4087 species detections made between 1900 and 2022. Sampling intensity varied annually within each of the river basins, with some receiving considerably more sampling attention than others. Generally speaking, the number of surveys began to rise in the 1970s, with more substantial numbers (≥5) of regular surveys occurring in the each of the MWMA river basins in most years from 1980 onward. Given these general patterns, we focus on the period between 1980 and 2022 here when describing the availability of data for quantitative temporal analysis across the MWMR river basins.

3.1.1 Werribee River Basin

In the Werribee River basin, between 1980 and 2022, there were eight years where no survey data was collated (1984, 1996, 1997, 1998, 2001, 2015, 2018, 2021) (Figure 1). For years where surveys were conducted, the mean annual number of sampling sites was 14 (range 1-62), although during only 21 of 34 years (62%) were five of more sites surveyed in a given year.

Figure 1 Number of sites sampled by year in the Werribee catchment



We further assessed survey statistics at the primary catchment level. We found that the six primary catchments of Werribee River Basin were sampled on 7 (16 %) to 21 (49 %) years across the 1980 and 2022 period (Table 1). During survey years, the mean number of sites surveyed range from 3 to 7 (30 at most). We considered this coverage to be unsuitable for an assessment of occupancy trends at this spatial scale, so we did not conduct the assessment at the finer sub-catchment level.

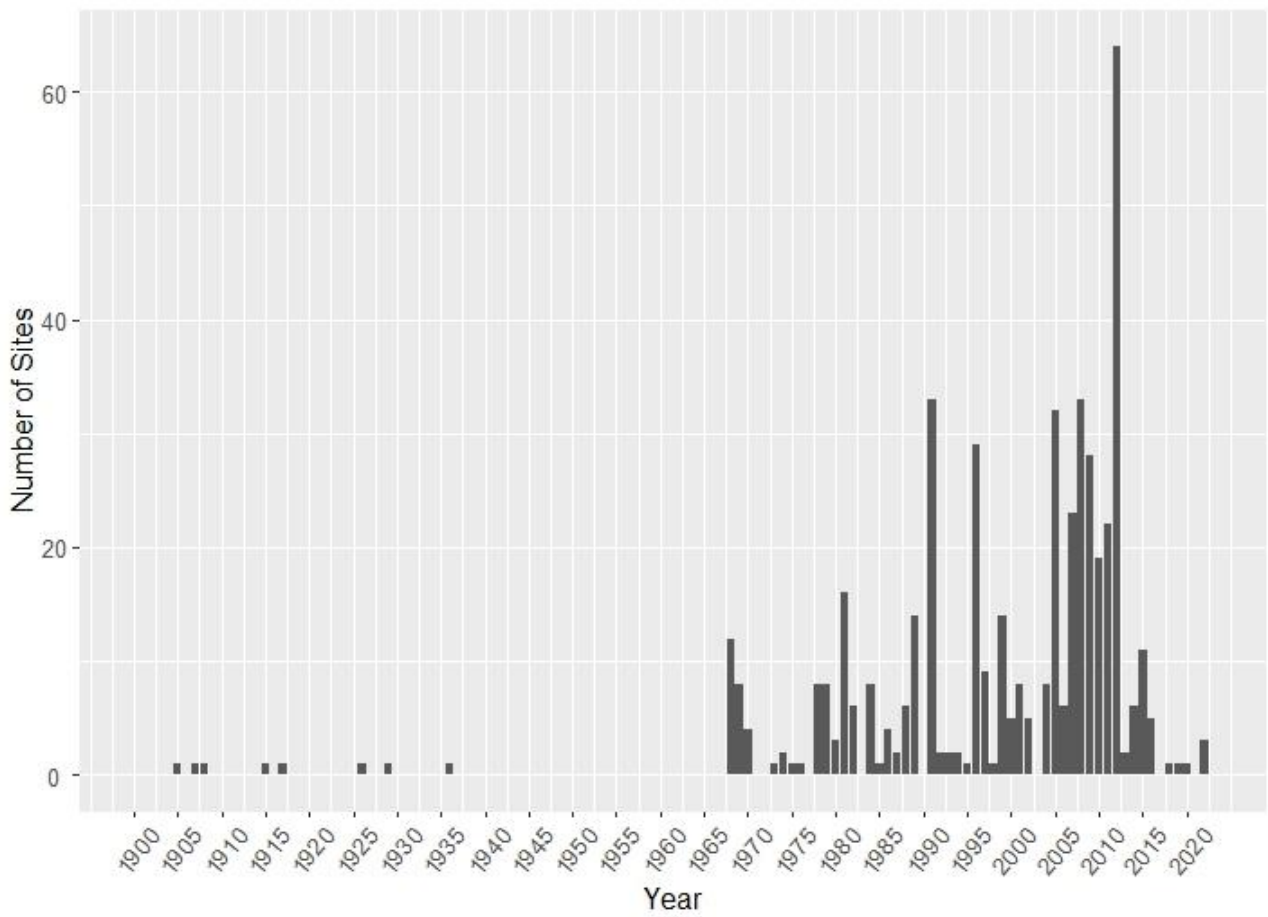
Table 1 Summary statistics for the number of sites sampled across the major sub-catchments of the Werribee catchment between 1980-2022

Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Kororoit Creek	16 (37)	4	1-11
Laverton Creek	8 (19)	3	1-6
Little River	16 (37)	7	1-30
Skeleton Creek	7 (16)	3	1-6
Werribee River	21 (49)	7	1-30
Werribee River Upper	17 (40)	5	1-14

3.1.2 Maribyrnong River Basin

In the Maribyrnong River Basin, between 1980 and 2022, there were four years where no survey data was collated (1983, 1990, 2003, 2017) (Figure 2). For years where surveys were conducted, the mean annual number of sampling sites was 11 (range 1-64), although during only 21 of 42 years (55%) were five of more sites surveyed in a given year.

Figure 2 Number of sites sampled by year in the Maribyrnong catchment



We further assessed survey statistics at the major sub-catchment level as that was the only spatial level available in our GIS dataset below river basin. We found that the six major sub-catchments of Maribyrnong catchment were sampled on 6 (14 %) to 28 (65 %) years across the 1980 and 2022 period (Table 2). During survey years, the mean number of sites surveyed range from 1 to 5 (32 at most). We considered this coverage to be unsuitable for an assessment of trends at this spatial scale.

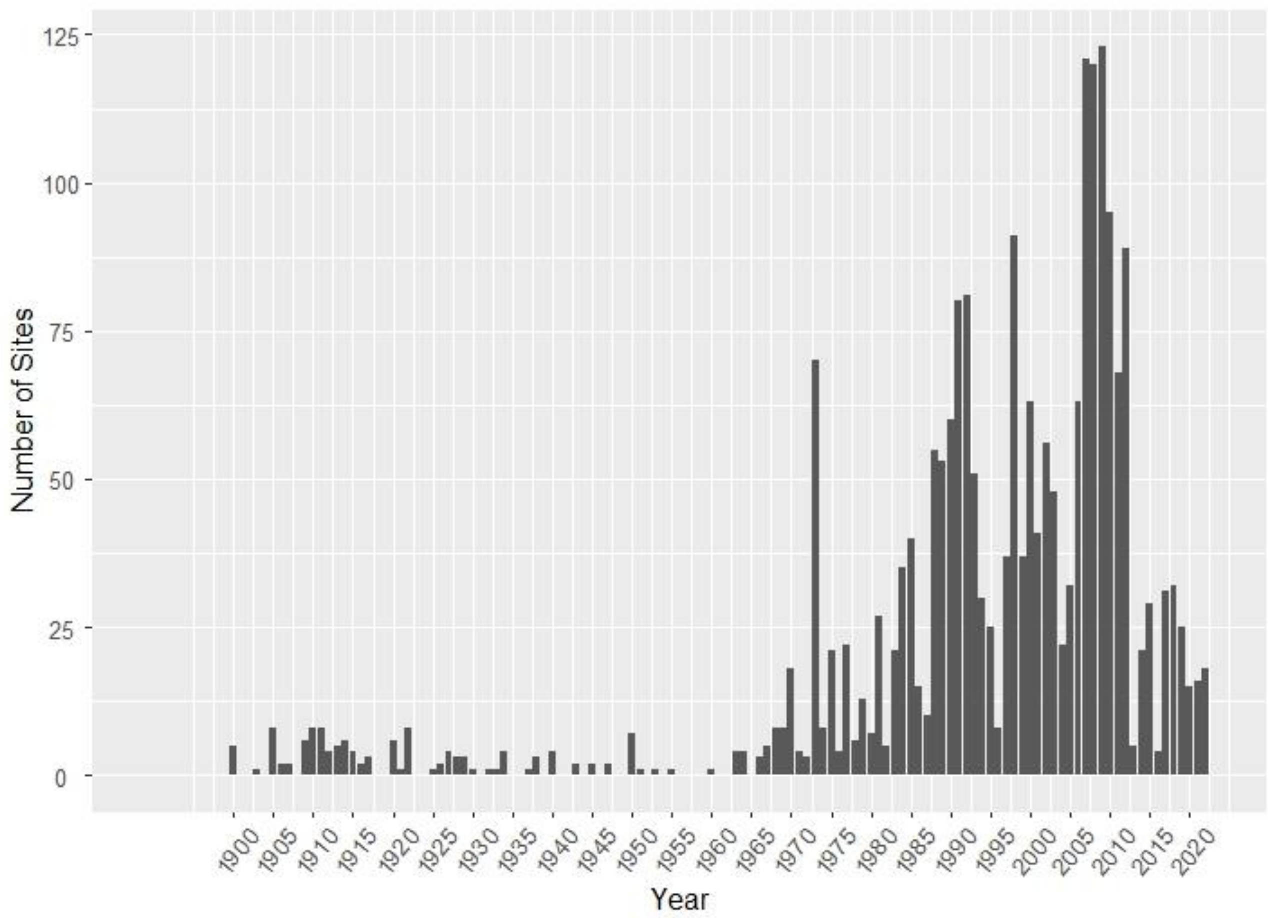
Table 2 Summary statistics for the number of sites sampled across the major sub-catchments of the Maribyrnong catchment between 1980-2022

Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Boyd Creek	6 (14)	1	1-2
Deep Creek	28 (65)	5	1-32
Emu Creek	9 (21)	4	1-11
Jacksons Creek	19 (44)	5	1-16
Maribyrnong River	25 (58)	5	1-11
Riddells Creek	10 (23)	2	1-9

3.1.3 Yarra River Basin

In the Yarra River Basin, between 1980 and 2022, there was a continuous record of survey data (Figure 3). The mean annual number of sampling sites was 44 (range 4-123), and five of more sites were surveyed in 40 of 42 years (95%). This was by far the most well sampled river basin.

Figure 3 Number of sites sampled by year in the Yarra catchment



We further assessed survey statistics at the major sub-catchment level given there was a consistent record and considerable number of sites sampled annually within the catchment. We found that the 18 major sub-catchments of Yarra River Basin were sampled on 1 (2 %) to 43 (100 %) years across the 1980 and 2022 period (Table 3). During survey years, the mean number of sites surveyed range from 1 to 15 (61 at most). We considered this coverage to be reasonable and considered the dataset suitable for an assessment of trends at this spatial scale.

Table 3 Summary statistics for the number of sites sampled across the major sub-catchments of the Yarra River Basin between 1980-2022

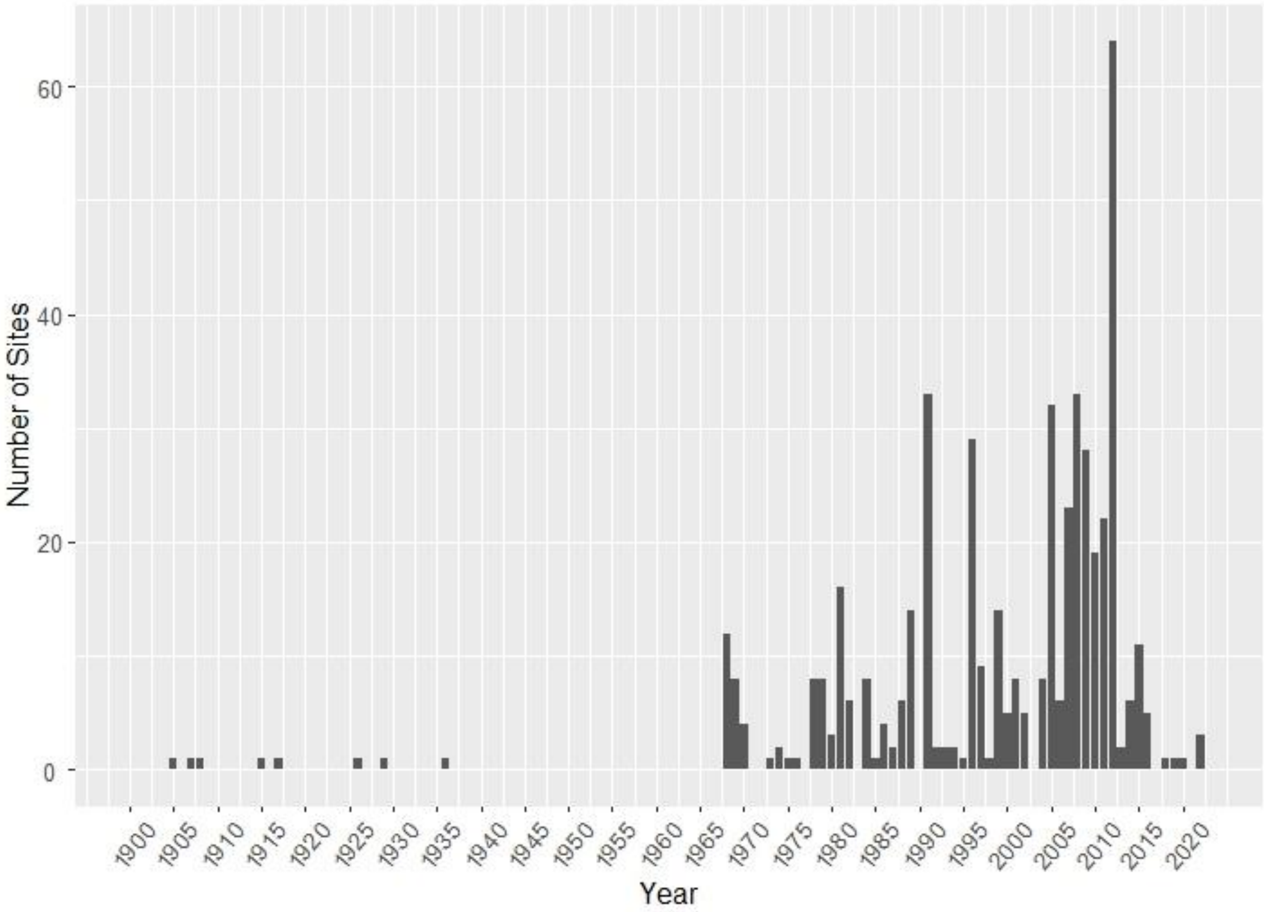
Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Brushy Creek	10 (23)	1	1-2
Darebin Creek	15 (35)	3	1-9
Diamond Creek	23 (53)	8	1-29
Gardiners Creek	11 (26)	3	1-10
Hoddles Creek	9 (21)	5	1-16
Koonung Creek	10 (23)	4	1-10
Little Yarra River	15 (35)	4	1-12

Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Merri Creek	29 (67)	4	1-16
Moonee Ponds Creek	9 (21)	3	1-8
Mullum Mullum Creek	21 (49)	5	1-22
Olinda Creek	21 (49)	5	1-23
Paul Creek	1 (2)	1	1-1
Plenty River	33 (77)	5	1-37
Steels Creek	6 (14)	4	1-12
Watsons Creek	11 (26)	3	1-12
Watts River	30 (70)	5	1-14
Woori Yallock Creek	20 (47)	6	1-25
Yarra River Main Stream	43 (100)	15	1-61

3.1.4 Dandenong Creek Basin

In the Dandenong Creek Basin, between 1980 and 2022, there were three years where no survey data was collated (1984, 2014, 2021) (Figure 4). For years where surveys were conducted, the mean annual number of sampling sites was 14 (range 1-78), although during only 15 of 39 years (38%) were five of more sites surveyed in a given year.

Figure 4 Number of sites sampled by year in the Dandenong catchment



We further assessed survey statistics at the primary catchment level. We found that the four primary catchments of Dandenong Creek Basin were sampled on 7 (16 %) to 35 (81 %) years across the 1980 and 2022 period (Table 4). During survey years, the mean number of sites surveyed range from 2 to 13 (64 at most). We considered the coverage to be unsuitable for an assessment of trends at this spatial scale as most surveys were conducted in the Patterson River, while very little data was available for the other primary catchments. Given this result, we did not conduct the analysis at the finer sub-catchment scale.

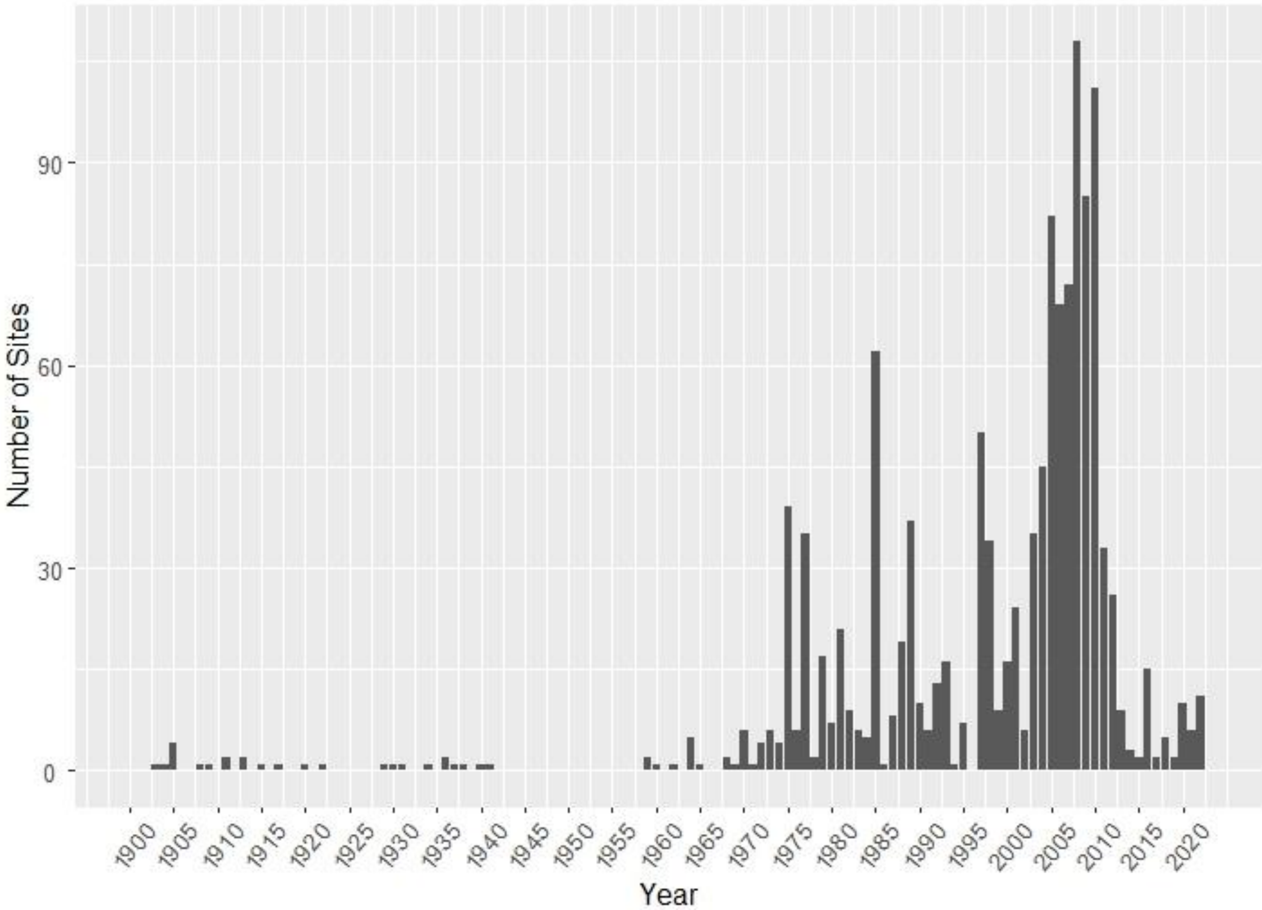
Table 4 Summary statistics for the number of sites sampled across the primary sub-catchments of the Western Port catchment between 1980-2022

Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Elster Creek	7 (16)	2	1-4
Kananook Creek	15 (35)	4	1-12
Mordialloc Creek	14 (33)	3	1-9
Patterson River	35 (81)	13	1-64

3.1.5 Western Port Basin

In the Western Port Basin, between 1980 and 2022, there was only one year (1996) where no survey data was collated (Figure 5). For years where surveys were conducted, the mean annual number of sampling sites was 26 (range 1-108), and greater than five of sites were surveyed in 34 of 42 years (83%).

Figure 5 Number of sites sampled by year in the Western Port catchment



The primary catchments of the Western Port Basin were sampled between 12 (21%) and 33 (79%) of years across the 1980 and 2022 period (Table 5). During survey years, the mean number of sites surveyed range from 3 to 9. We considered the dataset reasonably suitable for an assessment of occupancy trends at the primary catchment scale.

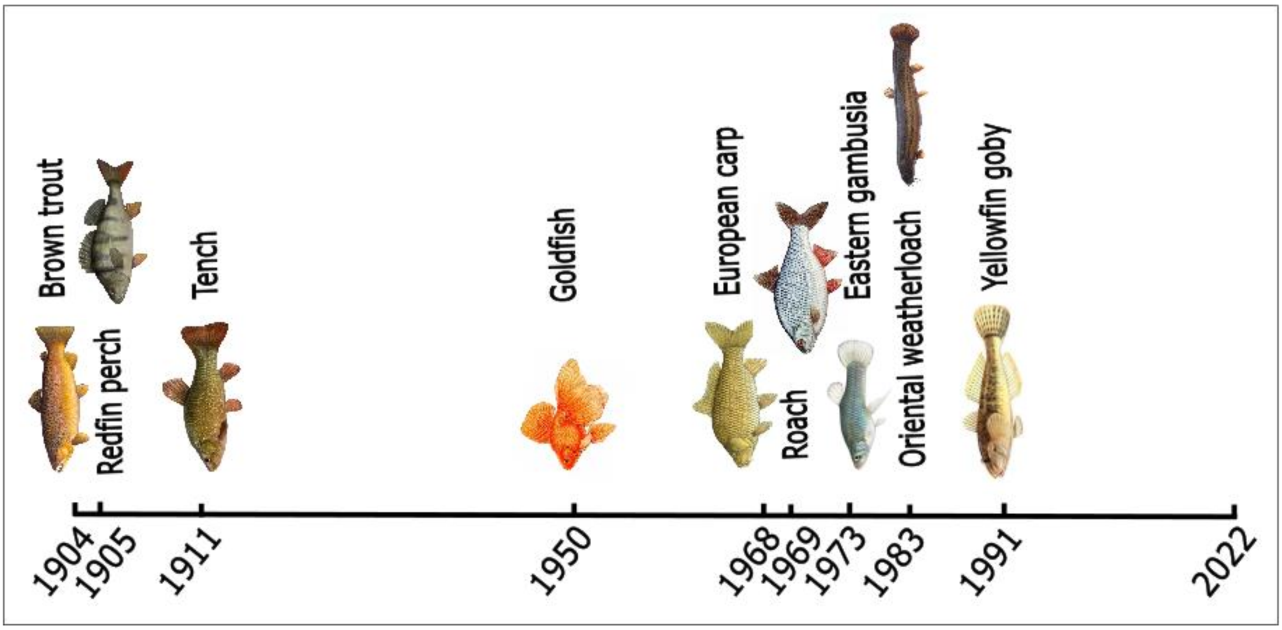
Table 5 Summary statistics for the number of sites sampled across the primary sub-catchments of the Western Port catchment between 1980-2022

Primary sub-catchment	Number of years with data / % of analysis period	No. sites (mean) sampled during survey years	No. sites (range) sampled during survey years
Bass River	12 (28)	5	1-24
Bunyip River	33 (77)	9	1-30
Cardinia Creek	20 (47)	7	1-25
Deep Creek	11 (26)	4	1-12
Lang Lang River	15 (35)	5	1-23
Toomuc Creek	9 (21)	3	1-6
Yallock Creek	12 (28)	6	1-24

3.2 History of exotic fish introductions in the Melbourne Water Management Region

The fish database provides a historic overview of exotic fish invasions in the MWMR. As the introduction of exotic species constitutes a significant change in the fish community structure and function, we summarise the timeline of these invasions here (Figure 6). The first three exotic species detected in the region were Brown trout (*Salmo trutta*), Redfin perch (*Perca fluviatilis*), and Tench (*Tinca tinca*). Each of these species are European sports fish that were purposefully stocked by Europeans in southeastern Australia during the mid to late1800s. Goldfish, a popular aquarium fish, were the next exotic species to appear in 1950, likely being released from aquaria or escaping from ponds. European carp (*Cyprinus carpio*) were first detected in the MWMR in 1968. The species was introduced to dams and reservoirs across Victoria, from the mid-1800s, but significant increases in their range were observed from the 1960s, which is consistent with this observation. Roach (*Rutilus rutilus*) were stocked in Australian waters, again in the mid-1800s, but appear to have invaded the MWMR much later, being first detected in 1969. Eastern gambusia (*Gambusia holbrooki*) were the next to appear in 1973. The species was first introduced to NSW from the USA in 1920 for the purpose of mosquito control and it has since spread across most of Australia. Oriental weatherloach (*Misgurndus anguillicaudatus*) were first detected in 1980 in the ACT, and then in the Yarra River in 1983. It is likely that they were released from aquaria. Yellowfin goby (*Acanthogobius flavimanus*) were the most recent arrival, being first detected in the Yarra, Werribee, and Maribyrnong river basins in 1991. It suspected that the species was accidently release in Australia, from Asia, in ballast water, and was originally detected in NSW in the 1970s.

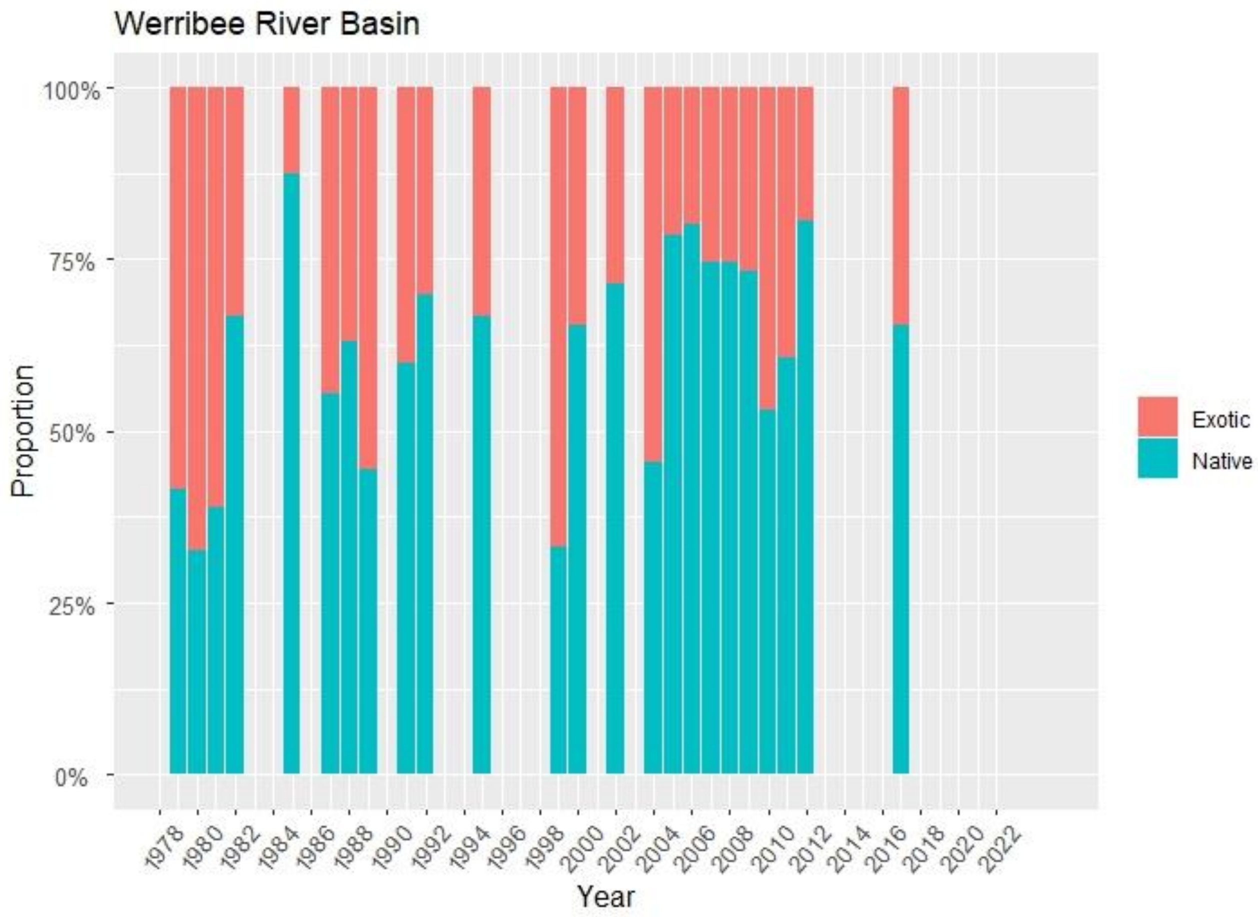
Figure 6 Years in which exotic fish species were first detected in the Melbourne Water Management Region



3.3 Nativeness scores through time in the Melbourne Water Management Region

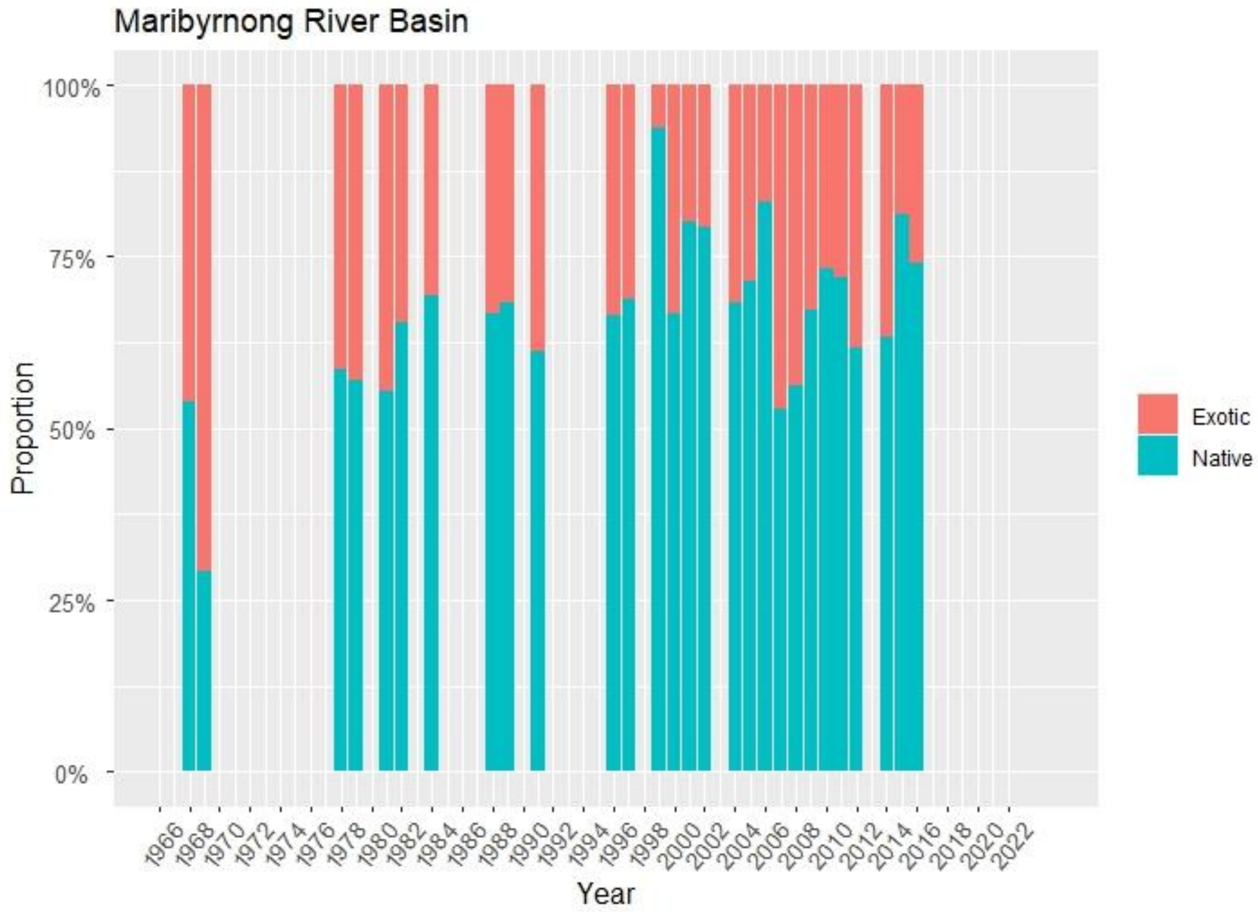
Broad sampling (more than five sites) of the Werribee River Basin began in 1979 and the nativeness plot indicates that the sampled fish community was dominated >50% by exotic species at that point in time (Figure 7). The proportion of native versus exotic species varied through time ranging from 12.5% to 70%. They made up more than half of the sampled fish community in six individual years. A lack of broad sampling in the catchment over the last decade has meant that recent conditions could not be assessed.

Figure 7 Proportion of native and exotic species, based on species richness, found the Werribee River by year. Only years where more than five sites were sampled within the catchment as this was considered minimum sampling coverage needed for interpretable results.



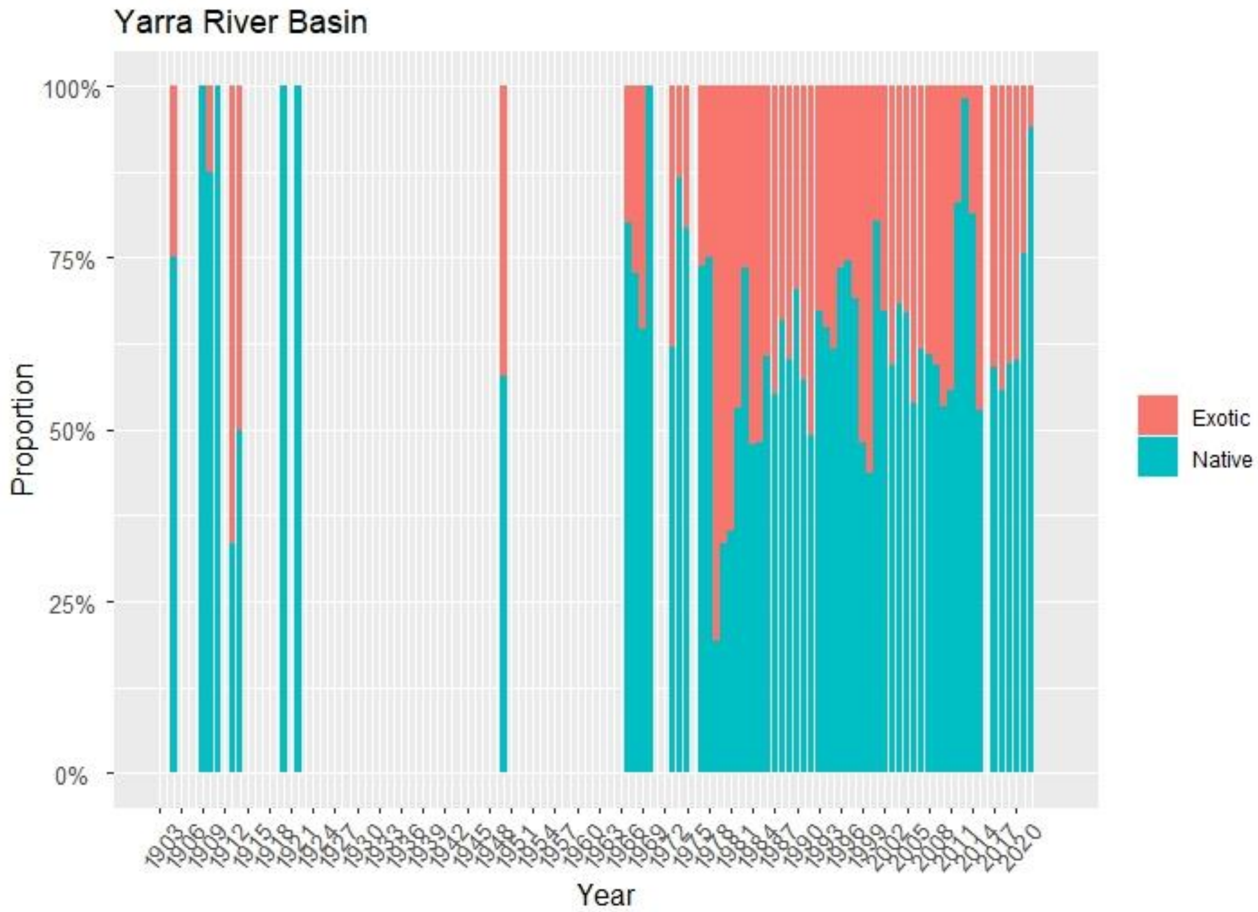
Broad sampling of the Maribyrnong River Basin began in 1968. There was a higher proportion of exotic species than native ones in only one year, 1969 (Figure 8). Otherwise, the proportion of exotic species ranged between 6% and 48%. A lack of broad sampling in the catchment over the seven years has meant that recent conditions could not be assessed.

Figure 8 Proportion of native and exotic species, based on species richness, found the Maribyrnong River by year. Only years where more than five sites were sampled within the catchment as this was considered minimum sampling coverage needed for interpretable results.



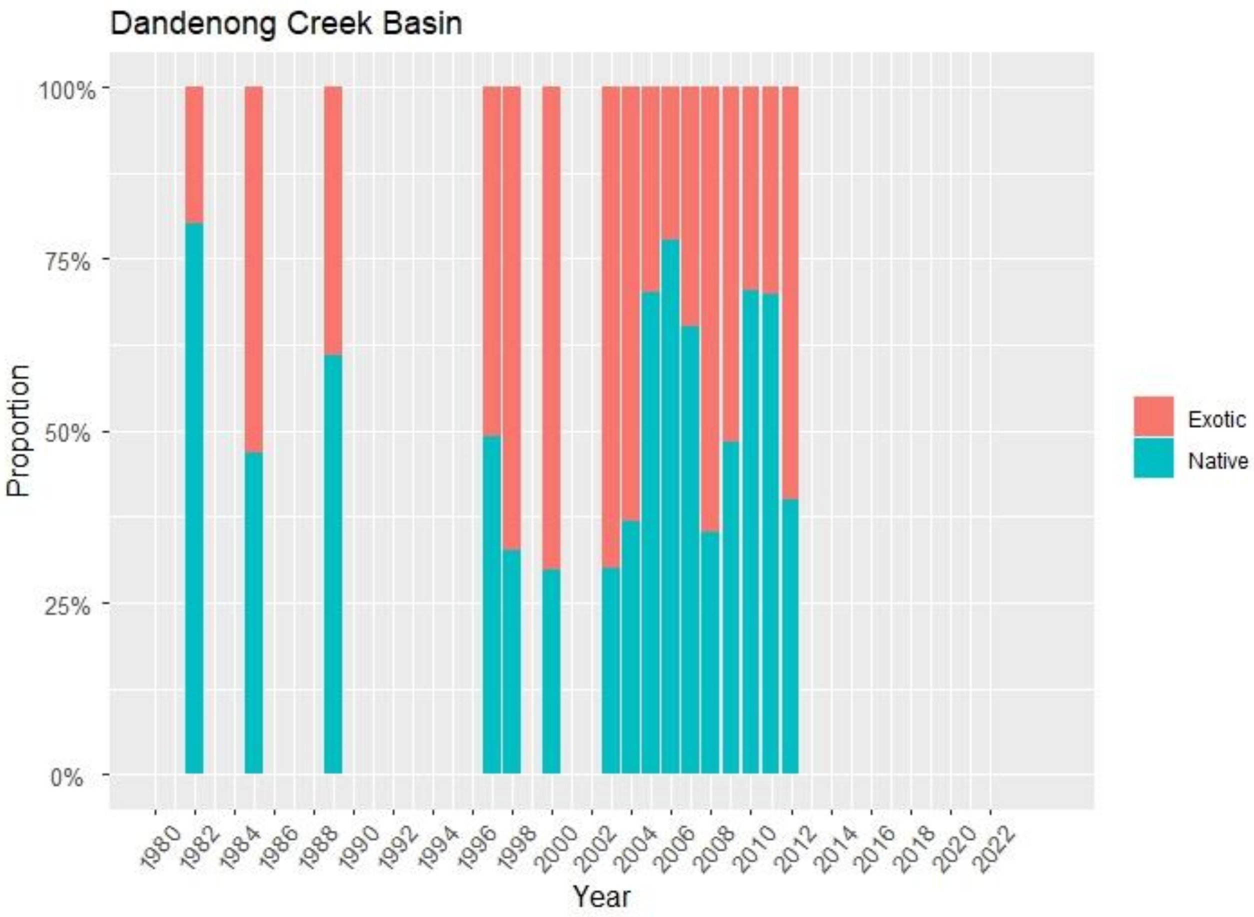
Some broad sampling of the Yarra River Basin occurred in the early 1900s, but regular annual sampling began in 1964. Apart from 1910 and 1911 where greater than or equal to 50% of the sampled fish community was made up of exotic species, native species clearly dominated the community up until 1975, after which exotic species made up a notably larger proportion (Figure 9). The proportion of exotics ranged between 2.5% and 78% but was typically less than 30%.

Figure 9 Proportion of native and exotic species, based on species richness, found the Yarra River by year. Only years where more than five sites were sampled within the catchment as this was considered minimum sampling coverage needed for interpretable results.



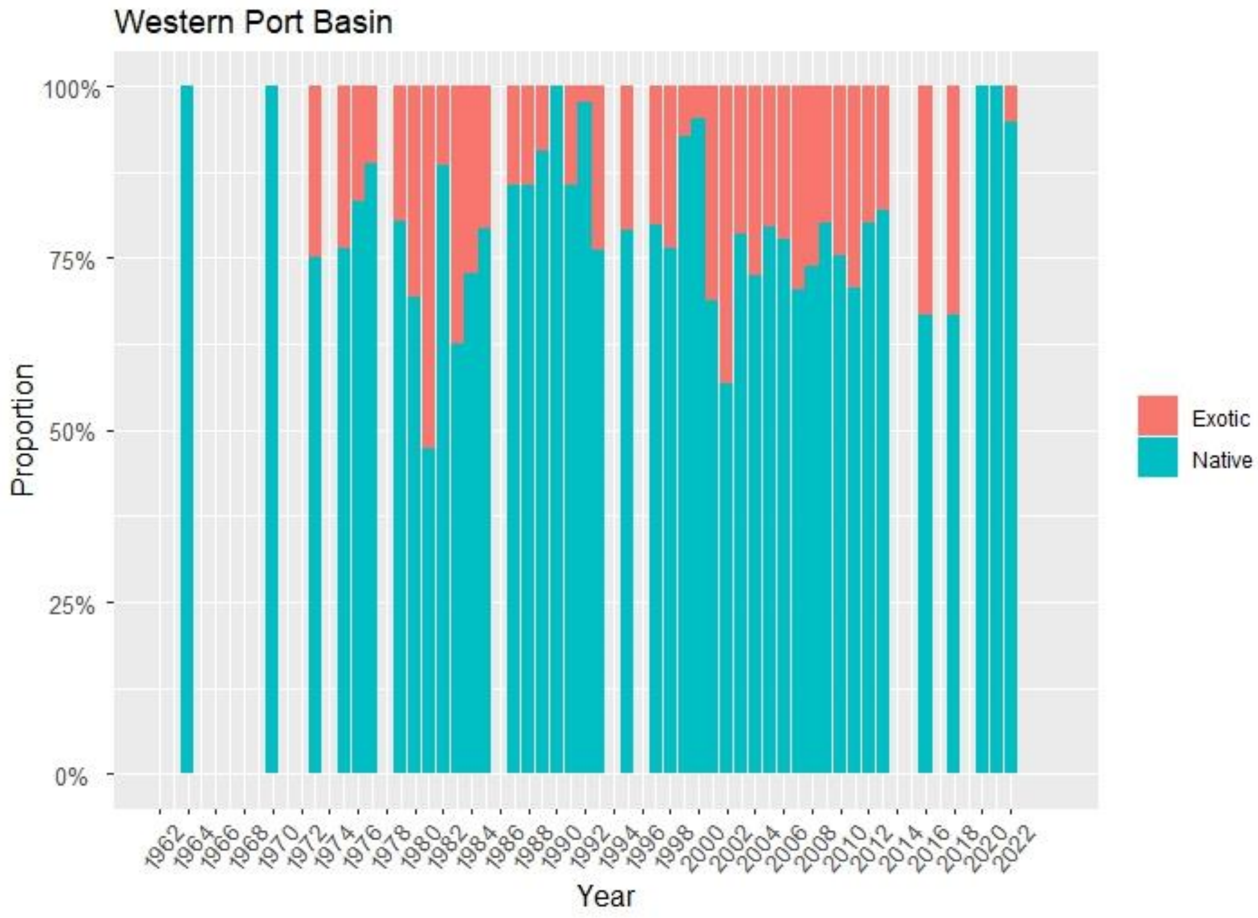
Broad sampling of the Dandenong Creek Basin began in 1982 but has been intermittent between then and present day. There has typically been a high proportion of exotic species, with more than half of the community being made up of exotic species in nine of the 16 years for which there was data (Figure 10). A lack of broad sampling in the catchment over the last decade has meant that recent nativeness values could not be assessed. Broadly speaking, it appears that Dandenong Creek fish community has a higher proportion of exotic species than the other MWMR basins.

Figure 10 Proportion of native and exotic species, based on species richness, found the Dandenong Creek by year. Only years where more than five sites were sampled within the catchment as this was considered minimum sampling coverage needed for interpretable results.



Broad sampling of the Western Port Basin began in 1964, but more regular annual sampling (of 5 of more sites) began in 1973. There was a higher proportion of exotic species than native ones in only one year, 1980. Otherwise, the proportion of exotic species ranged between 0% and 42%, being mostly less than 25% (Figure 11). The Western Port catchment has consistently had lowest proportion of exotic fish species in the community of any of the MWMR catchments.

Figure 11 Proportion of native and exotic species, based on species richness, found the Western Port catchment by year. Only years where more than five sites were sampled within the catchment as this was considered minimum sampling coverage needed for interpretable results.



3.4 Temporal trends in occupancy estimates in the Yarra River Basin

We fitted multi-season occupancy models to nine species based on presence/absence data collected across the Yarra River Basin. We discuss the main results herein.

Most importantly in interpreting the results in the context of this study, models that allow occupancy to varied between time periods are denoted by φ(YEAR), while models where occupancy was fixed across all 14 time periods (i.e. the null hypothesis) are denoted by $\varphi_{1\text{-}14}$(.). When comparing models, we consider any model that is within 2 AIC units of the top supported model as having substantial support (evidence) (see Burnham and Anderson 2004), so we discuss each of them.

Further to this, detection probability would be expected to vary between time periods in most cases in this analysis given time periods were 3-years long and the study period was around 50 years long. Over such time periods variance in the environment (e.g. flows) and populations (e.g. species abundance) could vary greatly, impacting on detectability. We interpret the probability of detection results with this in mind.

Finally, to the best of our knowledge, robust methods for assessing model fit and accounting for zero inflation and overdispersion in a dataset, are yet to be developed for multi-season models and they are certainly not available in the PRESENCE package. As such, model fit statistics are not reported here. Rather, we somewhat accounted for these potential issues by analysing a data rich dataset (i.e. Yarra River Basin between 1973-2022).

3.4.1 River blackfish

River Blackfish were detected in 12 of the 18 sampled sites (67 %) between 1973 and 2022. We modelled occupancy, colonization, extinction, and detection as varying with YEAR or held constant.

The best-supported models (Models 1 and 2, $AIC_{W} = 0.4048$, Table 6), with 41% weight of evidence each, estimated initial occupancy ($\varphi_{1973\text{-}1975}$) and assumed that it was constant through all 14 time periods. Similarly, invasion (γ) and extinction (ε) parameters were given initial estimates for model 1 and 2 respectively, and assumed to be constant through time. However, reciprocal equations were used to calculate yearly changes in detectability (*p*).

These results suggest that probability of occupancy (estimate = 0.50), colonization rates (estimate = 0.10), and extinction rates (estimate = 0.10) did not vary among years, although detectability did differ. The extinction probability estimate was high relative to most of the species we examined, but was equal to the invasion probability. The observed temporal changes in detectability may be an ecological signal (e.g. the species is in lower abundance during a given time period and therefore harder to detect) or an artifact of sampling (e.g. river levels were up during sampling in a given period and thus detectability was lower). Regardless, these changes followed no particular pattern suggesting they represent natural variability rather than evidence of say, chronic decline in species abundance.

This variability aside, the estimated detection probability across surveys and years was $p = 0.68$ (SE = 0.05). This indicates that, on average, fishing surveys detected River blackfish when they were present at Yarra River sites about 2/3 of the time, and thus missed detecting them about 1/3 of the time. Naïve estimates of occurrence of River Blackfish in the Yarra River are therefore slightly negatively biased.

Table 6 Occupancy models of River blackfish in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

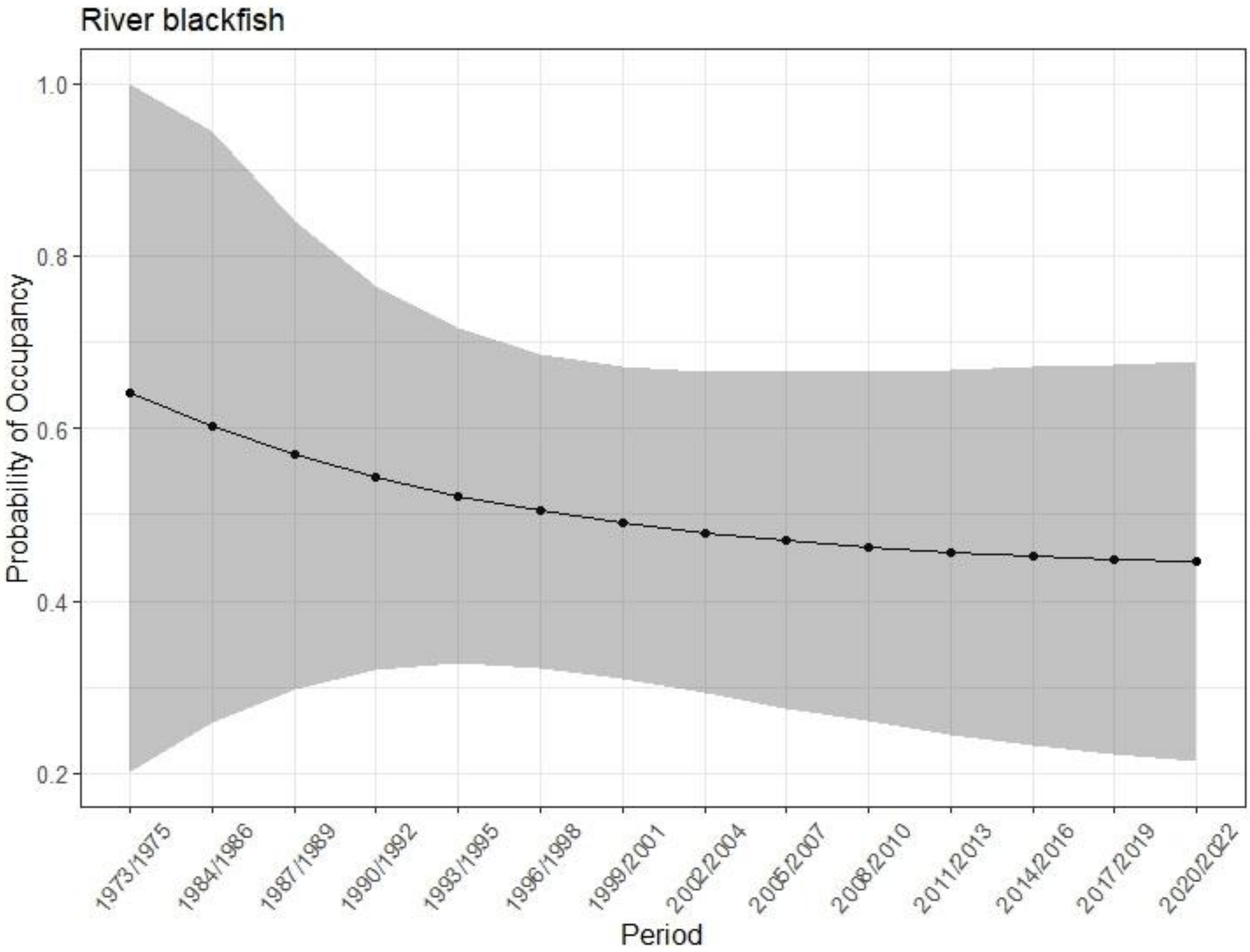
ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML	#para	2LL
1	$\varphi_{1\text{-}14}(.),\gamma(.),p(\text{YEAR})$	317.03	0.00	0.4048	1.0000	16	285.03
2	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(\text{YEAR})$	317.03	0.00	0.4048	1.0000	16	285.03
3	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(\text{YEAR})$	318.55	1.52	0.1893	0.4677	17	284.55

Model 3 received reasonable support ($AIC_{W} = 0.1893$, Table 6), with 19% weight of evidence and being only 1.5 AIC units different from models 1 and 2. In this model, occupancy and detectability were allowed to vary through time in a Markovian pattern (i.e. the probability that a unit is occupied time period 2 depends upon the unit's occupancy state in time period 1, and so on) while invasion and extinction were fixed as their initial estimate.

The results of this model indicate a consistent pattern of decline in the probability of site occupancy for River blackfish through time (Figure 12). Mean occupancy estimates in the final 2020-2022 period were 31% lower than in the initial 1973-1975 period (95 % $CI\varphi_{1973\text{-}1975} = 0.64 \pm 0.20–1.0$; 95 % $CI\varphi_{2020\text{-}2022} = 0.45 \pm 0.21–0.68$). However, these differences were not statistically significant as 95 % confidence intervals overlap.

Figure 12 Model-averaged estimates of annual probability of site occupancy (φ_t) for River blackfish



3.4.2 Ornate galaxias

Ornate galaxias were detected in 10 of the 18 sampled sites (56 %) between 1973 and 2022. The best-supported model describing Common galaxias occupancy (Models 1, $AIC_{W=}$ 0.7302, Table 7), with 73% weight of evidence, assumed occupancy (estimate = 0.49) and invasion (estimate = 0.01) parameters were constant through time, while detectability varied between years. Inter-annual changes in detectability followed no particular pattern between time periods.

These results strongly suggest that occupancy has not changed significantly through time. Detectability has changed, but as it follows no trend it is likely due to natural variability within the between study periods.

Table 7 Occupancy models of Ornate galaxias in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	$\varphi_{1\text{-}14}(.),\gamma(.),p(\text{YEAR})$	272.95	0.00	0.7302	1.0000	16	240.95
2	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(\text{YEAR})$	275.12	2.17	0.2467	0.3379	17	241.12
3	$\varphi(\text{YEAR}),\gamma(\text{YEAR}),\varepsilon(\text{YEAR}),p(.)$	282.81	9.86	0.0053	0.0072	4	274.81
4	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(.)$	282.81	9.86	0.0053	0.0072	4	274.81
5	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(\text{YEAR})$	283.36	10.41	0.0040	0.0055	3	277.36
6	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(.)$	283.36	10.41	0.0040	0.0055	3	277.36

3.4.3 Southern pygmy perch

Southern pygmy perch were detected in 12 of the 18 sampled sites (67 %) between 1973 and 2022. The best-supported model describing Southern pygmy perch occupancy (Model 1, $AIC_{W} =$ 0.2478, Table 8), with 25% weight of evidence, assumed fixed extinction, and invasion parameters between time periods, but variable occupancy and detectability between periods.

The results suggest that occupancy has varied through time along with detectability, while extinction (estimate = 0.03) and invasion (estimate = 0.00) estimates have remained similar. While the extinction probability estimate was low, it exceeded the invasion estimate that was zero. Figure 13 shows linear decrease in occupancy between the earliest and latest time periods (95 % $CI\varphi_{1973\text{-}1975}$ = 0.70 ± 0.42–0.98; 95 % $CI\varphi_{2020\text{-}2022}$ = 0.45 ± 0.17–0.72), marking a 36% decrease in mean occupancy estimates.

On the other hand, changes in detectability did not show a consistent pattern. The estimated detection probability across surveys and years was $p = 0.34$ (SE = 0.04). This indicates that, on average, fishing surveys detected Southern pygmy perch when they were present at Yarra River sites about 1/3 of the time, and thus missed detecting them about 2/3 of the time. This would suggest a substantial negative bias in naïve estimates of Southern pygmy perch occurrence in the Yarra River.

These results suggest that occupancy has declined through time. On the whole, detectability is fairly low for the species and varies between time periods, but it does not correlate with changes in occupancy suggesting that they are influenced by different factors.

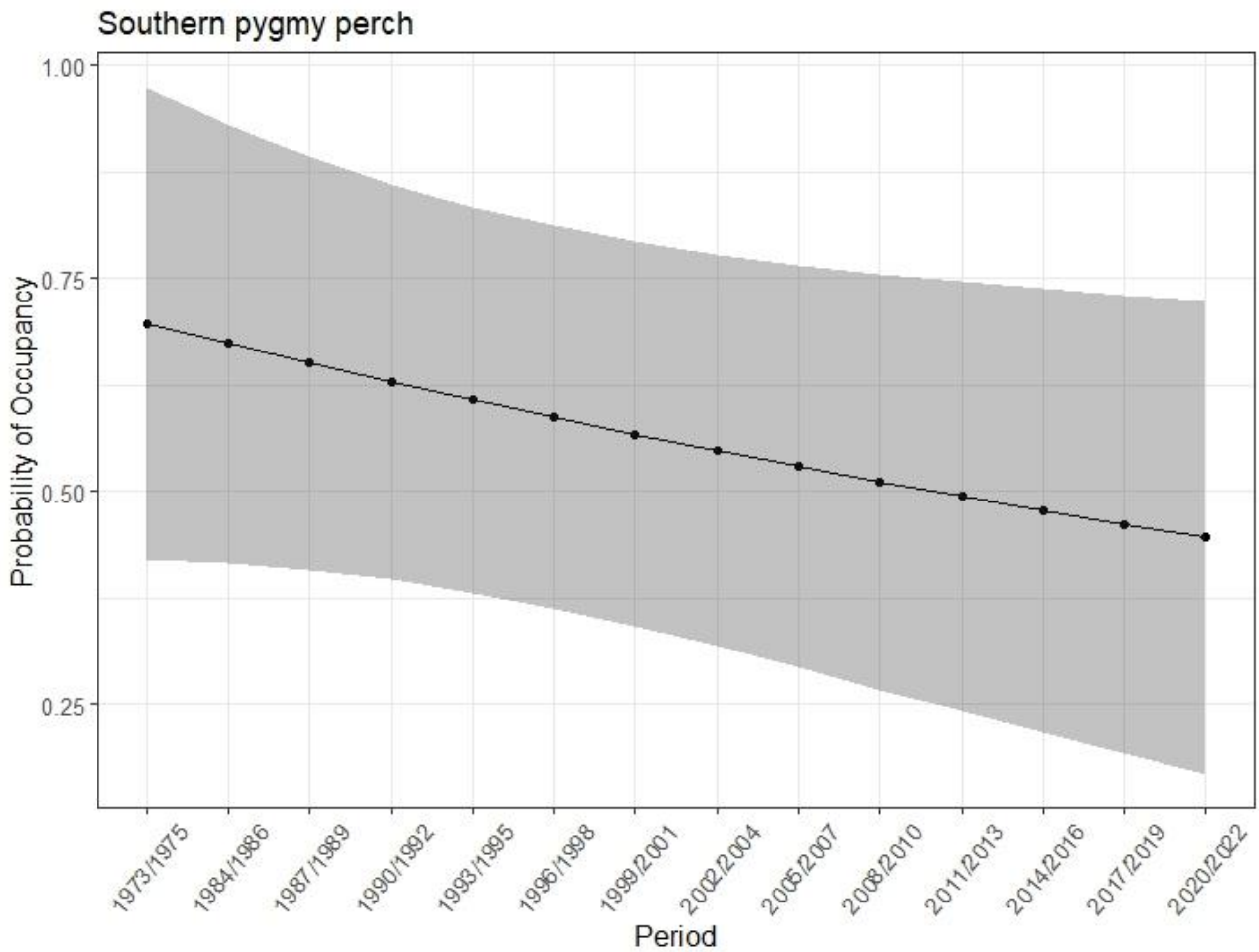
Models 2 and 3, which each assumed occupancy was constant between time periods, also received good support, likely due to the wide confidence intervals around the occupancy estimates. They present some evidence in favour of the null hypothesis.

Table 8 Occupancy models of Southern pygmy perch in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	φ(YEAR),γ(.),ε(.),*p*(YEAR)	283.72	0.00	0.2478	1.0000	17	249.72
2	φ_{1-14}(.),γ(.),*p*(YEAR)	283.90	0.18	0.2265	0.9139	16	251.90
3	φ_{1-14}(.),ε(.),*p*(YEAR)	283.90	0.18	0.2265	0.9139	16	251.90
4	φ(YEAR),γ(.),ε(.),*p*(.)	285.72	2.00	0.1003	0.3679	4	277.72
5	φ_{1-14}(.),γ(.),*p*(.)	286.63	2.91	0.0637	0.2334	3	280.63
6	φ_{1-14}(.),ε(.),*p*(.)	286.63	2.91	0.0637	0.2334	3	280.63

Figure 13 Model-averaged estimates of annual probability of site occupancy (φ_t) for Southern pygmy perch



3.4.4 Common galaxias

Common galaxias were detected in 16 of the 18 sampled sites (89 %) between 1973 and 2022. The best-supported model describing Common galaxias occupancy (Models 1, AIC_{W} = 0.9084, Table 9), with 91% weight of evidence, allowed occupancy and invasion parameters to vary between time periods, while extinction and detectability remained constant.

The results suggest that probability of occupancy and invasion of new sites has varied through time, while probability of extinction (estimate = 0.31) and detectability (estimate = 0.67) have remained similar. It's important to note that the extinction probability estimate is the highest of the species examined here. Figure 14 shows a large, statistically significant increase in occupancy between periods before 1992 and the 1993-1995 period (e.g. 95 % $CI\varphi_{1990\text{-}1992}$ = 0.19 ± 0.03–0.34; 95 % $CI\varphi_{2020\text{-}2022}$ = 0.68 ± 0.34–1.00), marking a 73% increase in mean occupancy estimates. Invasion parameter estimates exhibited a similar jump. A second significant increase in occupancy occurred between the 2011-2013 and 2014-2016 periods (95 % $CI\varphi_{2011\text{-}2013}$ = 0.49 ± 0.28–0.69; 95 % $CI\varphi_{2014\text{-}2016}$ = 0.85 ± 0.80–0.90), which was accompanied by a jump in invasion estimates. However, this was immediately followed by a significant decline (95 % $CI\varphi_{2014\text{-}2016}$ = 0.85 ± 0.80–0.90; 95 % $CI\varphi_{2017\text{-}2019}$ = 0.59 ± 0.46–0.71) that continued through to the 2020-2022 period (95 % $CI\varphi_{2020\text{-}2022}$ = 0.41 ± 0.25–0.57), with invasion estimates being near zero.

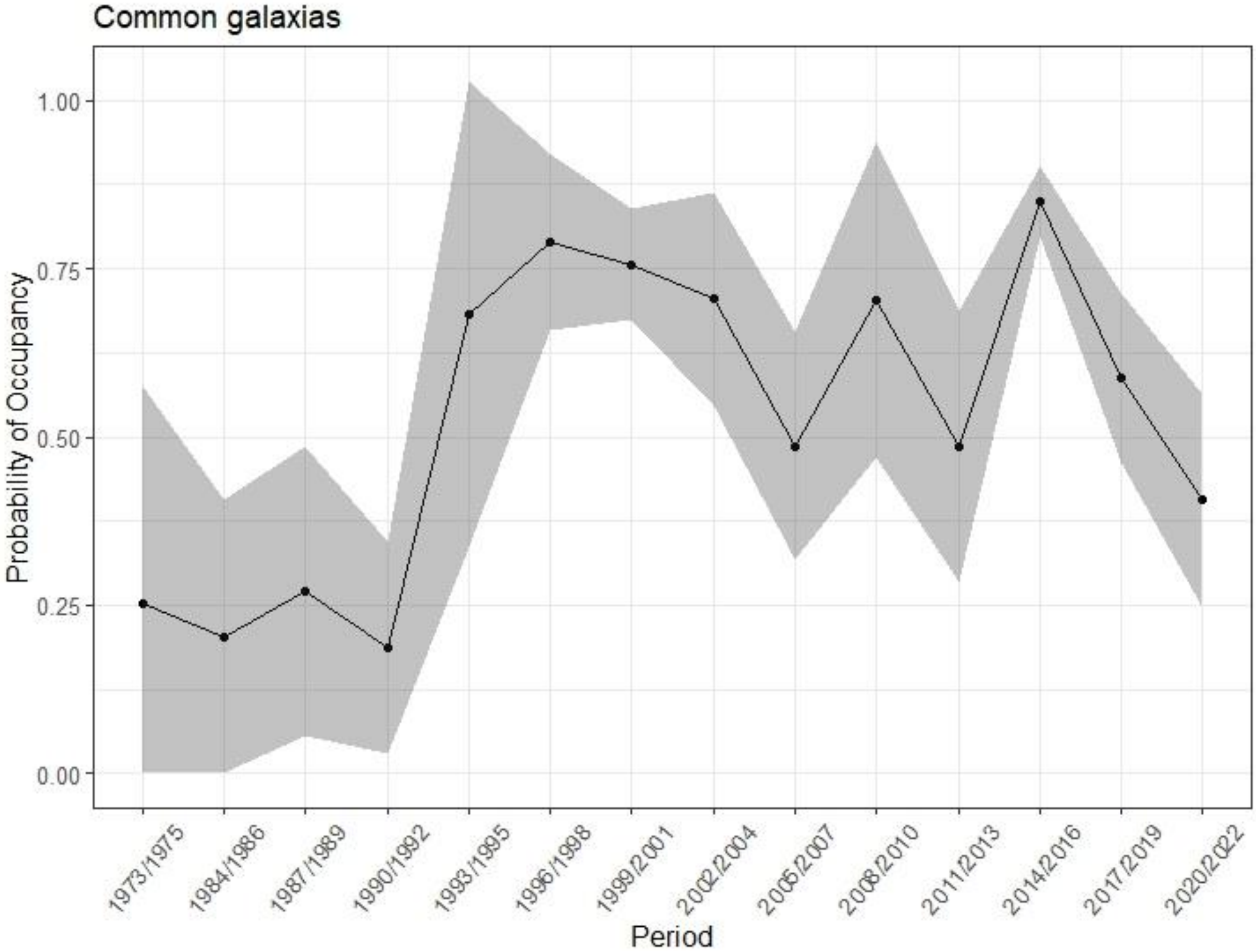
These results strongly suggest that occupancy has changed at various points in time, largely increasing, and this is due to an increase in invasion of new sites rather than any variability in detectability. However, there is evidence of more recent decline due to extinction from various sub-catchments.

Table 9 Occupancy models of Common galaxias in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	φ(YEAR),γ(YEAR),ε(.),*p*(.)	371.90	0.00	0.9084	1.0000	16	339.90
2	φ(YEAR),γ(YEAR),ε(YEAR),*p*(.)	378.33	6.43	0.0352	0.0402	28	322.33
3	$\varphi_{1\text{-}14}$(.),γ(YEAR),*p*(.)	379.42	7.52	0.0212	0.0233	15	349.42
4	$\varphi_{1\text{-}14}$(.),ε(YEAR),*p*(.)	379.49	7.59	0.0204	0.0225	15	349.49
5	φ(YEAR),γ(.),ε(.),*p*(.)	379.95	8.05	0.0162	0.0179	4	371.95
6	$\varphi_{1\text{-}14}$(.),γ(.),*p*(YEAR)	380.85	8.95	0.0103	0.0114	16	348.85
7	$\varphi_{1\text{-}14}$(.),ε(.),*p*(.)	382.17	10.27	0.0053	0.0059	3	376.17
8	$\varphi_{1\text{-}14}$(.),γ(.),*p*(.)	382.17	10.27	0.0053	0.0059	3	376.17
9	$\varphi_{1\text{-}14}$(.),ε(.),*p*(YEAR)	382.17	10.27	0.0053	0.0059	3	376.17
10	φ(YEAR),γ(.),ε(.),*p*(YEAR)	382.33	10.43	0.0049	0.0054	17	348.33

Figure 14 Model-averaged estimates of annual probability of site occupancy (φ_t) for Common galaxias



3.4.5 Short-finned eel

Short-finned eel were detected in all of the 18 sampled sites (100 %) between 1973 and 2022. The best-supported models (Models 1 and 2, AIC_{W} = 0.3954, Table 10), with 40% weight of evidence each, assumed that occupancy was constant through time. Similarly, invasion and extinction parameters were given initial estimates for model 1 and 2 respectively, and assumed to be constant through time. However, reciprocal equations were used to calculate yearly changes in detectability (*p*).

These results suggest that probability of occupancy (estimate = 1.00), colonization rates (estimate = 0.00), and extinction rates (estimate = 0.00) did not vary among years, although detectability did differ. Inter-annual changes in detectability followed no clear pattern. The estimated detection probability across surveys and years was p = 0.77 (SE = 0.03). This indicates that, on average, fishing surveys detected Short-finned eel when they were present at Yarra River sites about 3/4 of the time, and thus missed detecting them about 1/4 of the time. Naïve estimates of occurrence of Short-finned eel in the Yarra River therefore only have a slight negative bias.

Together these results indicate that short-finned eel have been widely distributed throughout the Yarra River Basin throughout the time period investigated here, and there has been no significant change in occupancy.

Table 10 Occupancy models of Short-finned eel in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (p) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

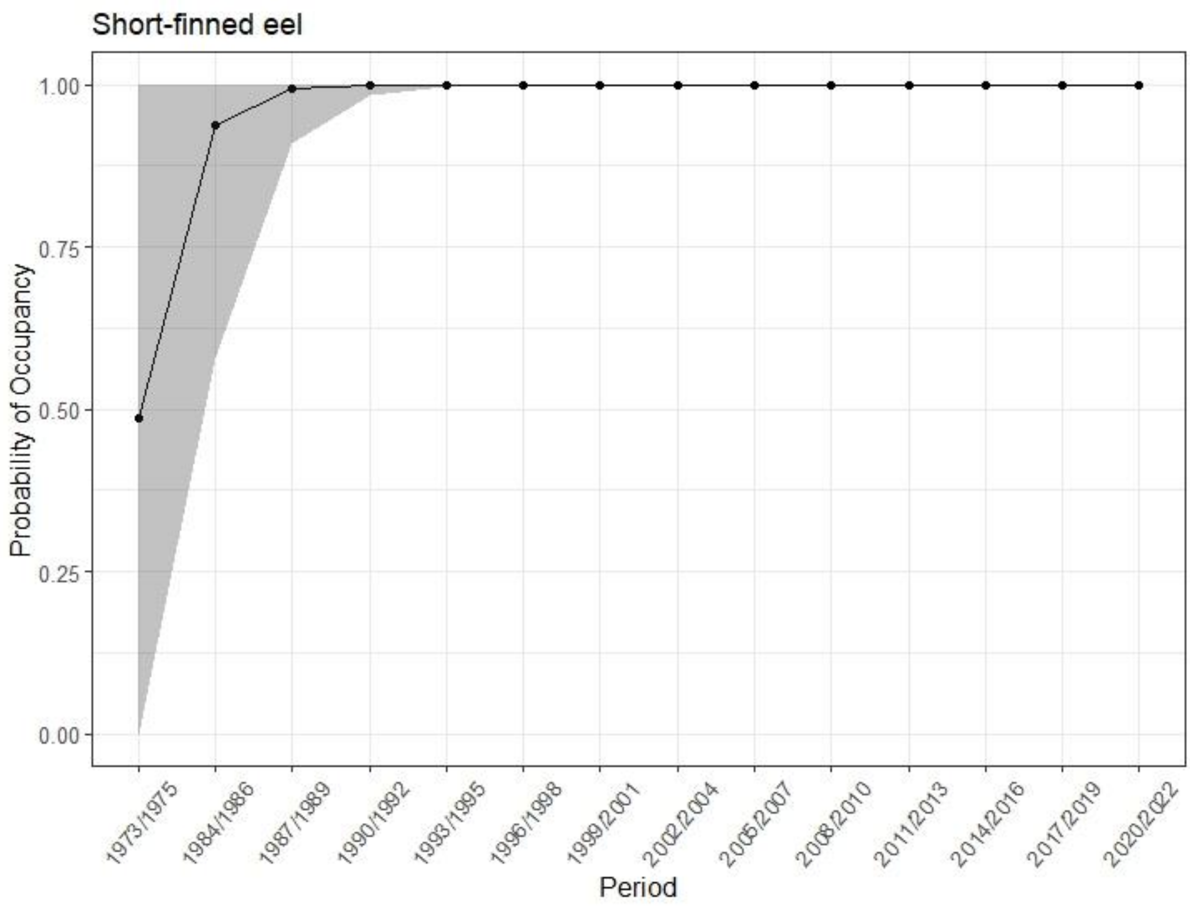
ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	$\varphi_{1\text{-}14}(.),\gamma(.),p(\text{YEAR})$	360.28	0.00	0.3954	1.0000	16	328.28
2	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(\text{YEAR})$	360.28	0.00	0.3954	1.0000	16	328.28
3	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(\text{YEAR})$	361.56	1.28	0.2085	0.5273	17	327.56

Model 3 provided evidence for the alternative hypothesis. The model received reasonable support (AIC_{W} = 0.2085, Table 10), with 21 % weight of evidence and being only 1.28 AIC units different from models 1 and 2. In this model, occupancy and detectability varied through time, while invasion and extinction were fixed as their initial estimate.

The results of this model indicate a large increase in the probability of site occupancy for Short-finned eel between the initial 1973-1975 time period (95 % $CI\varphi_{1973\text{-}1975}$ = 0.49 ± 0.00–1.00) and each of the remaining time periods which had occupancy estimates at or close to 1.00 (Figure 15). The very wide confidence intervals around the 1973-1975 estimates, mean that the change is not statistically significant, although any period in which the species was not near full occupancy of the Yarra River's sub-catchments is notable.

Figure 15 Model-averaged estimates of annual probability of site occupancy (φ_t) for Short-finned eel.



3.4.6 Tupong

Tupong were detected in 8 of the 18 sampled sites (44 %) between 1973 and 2022. The best-supported models (Models 1 and 2, AIC_{W} = 0.3216, Table 11), with 32% weight of evidence each, assumed that occupancy (estimate = 0.24) and detectability (estimate = 0.30) was constant through time. Similarly, invasion (estimate = 0.00) and extinction (estimate = 0.01) parameters were assumed to be constant through time in model 1 and 2 respectively.

The estimated detection probability across surveys and years was p = 0.30 (SE = 0.09). This indicates that, on average, fishing surveys detected Tupong when they were present at Yarra River sites about 1/3 of the time, and thus missed detecting them about 2/3 of the time. Naïve estimates of occurrence of Tupong in the Yarra River therefore have a fairly strong negative bias.

These results suggest that Tupong populations in the Yarra River Basin are in equilibrium and occupancy has not changed significantly through time.

Table 11 Occupancy models of Tupong in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

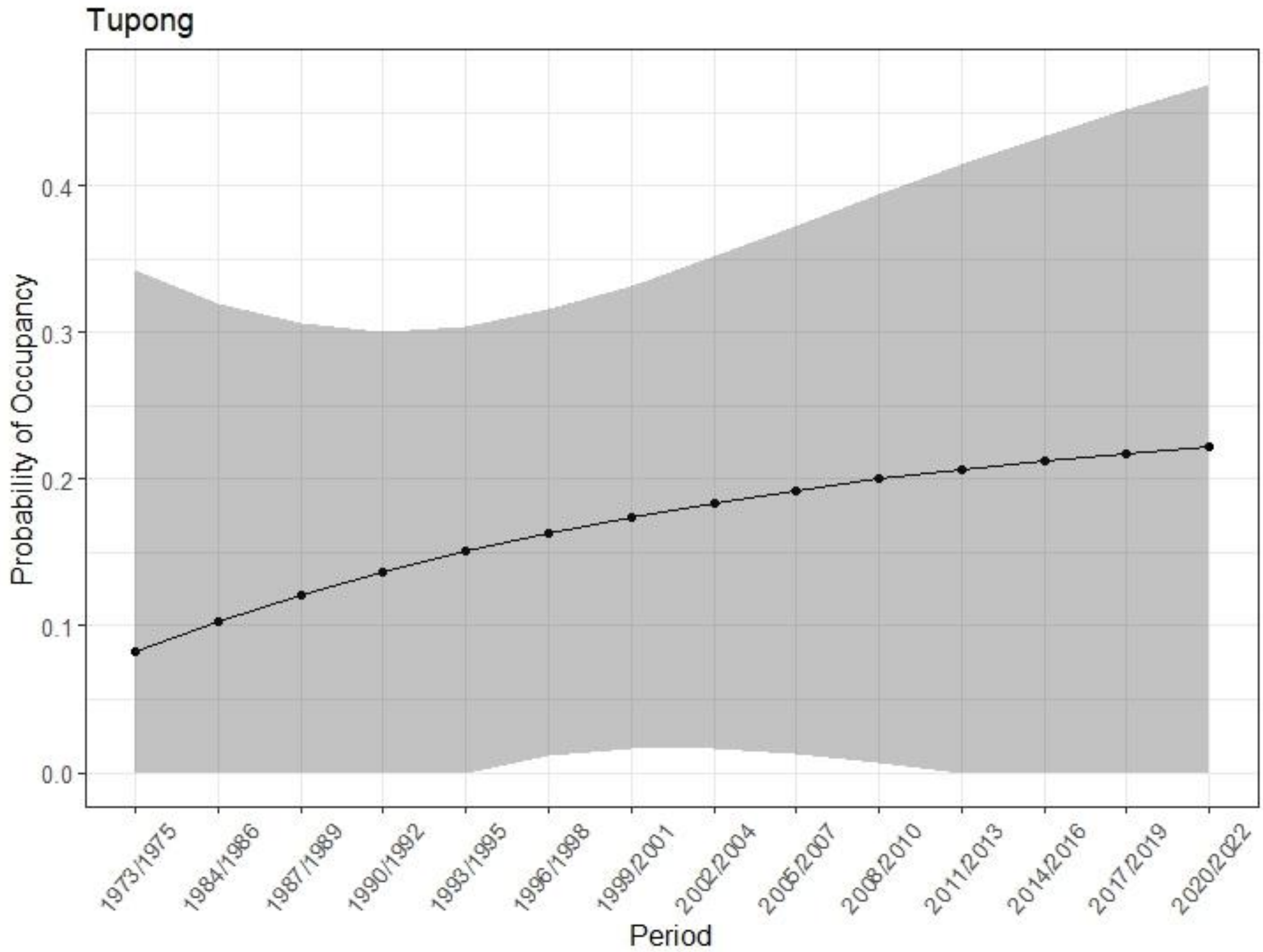
ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	$\varphi_{1-14}(.),\gamma(.),p(.)$	114.95	0.00	0.3216	1.0000	3	108.95
2	$\varphi_{1-14}(.),\varepsilon(.),p(.)$	114.95	0.00	0.3216	1.0000	3	108.95
3	$\varphi(YEAR),\gamma(.),\varepsilon(.),p(.)$	116.09	1.14	0.1818	0.5655	4	108.09
4	$\varphi_{1-14}(.),\gamma(.),p(YEAR)$	117.90	2.95	0.0736	0.2288	16	85.90
5	$\varphi_{1-14}(.),\varepsilon(.),p(YEAR)$	117.90	2.95	0.0736	0.2288	16	85.90
6	$\varphi(YEAR),\gamma(.),\varepsilon(.),p(YEAR)$	119.89	4.94	0.0272	0.0846	17	85.89

Model 3 provided evidence for the alternative hypothesis, with occupancy varying through time, while invasion and extinction and detectability remained constant. The model received reasonable support (AIC_{W} = 0.1818, Table 11), with 18 % weight of evidence and being only 1.14 AIC units different from models 1 and 2.

The results of this model indicate a steady increase in the probability of site occupancy for Tupong between the initial 1973-1975 time period (estimate = 0.08) and the final 2020-2022 period (estimate = 0.22) (Figure 16). The very wide confidence intervals around the estimates mean that the change is not statistically significant, although the pattern is consistent over the nearly 50-year time period.

Figure 16 Model-averaged estimates of annual probability of site occupancy (φ_t) for Tupong.



3.4.7 Pouched lamprey

Pouched lamprey were detected in 4 of the 18 sampled sites (22 %) between 1973 and 2022. The best-supported models describing Pouched lamprey occupancy (Models 1 and 2, $AIC_{W} = 0.3430$, Table 11), with 34% weight of evidence, assumed occupancy (estimate = 0.19), invasion (estimate = 0.00), and extinction (estimate = 0.00) parameters were constant through time, while detectability varied between years.

Inter-annual changes in detectability followed no particular pattern between time periods. The estimated detection probability across surveys and years was $p = 0.28$ (SE = 0.06). This indicates that, on average, fishing surveys detected Pouched lamprey when they were present at Yarra River sites about 1/4 of the time, and thus missed detecting them about 3/4 of the time. Naïve estimates of occurrence of Pouched lamprey in the Yarra River Basin therefore have a strong negative bias.

Overall, these results suggest that Pouched lamprey have only ever occupied a small area of the Yarra River Basin during the study period and their occupancy has not changed significantly through time.

Table 12 Occupancy models of Pouched lamprey in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	$\varphi_{1\text{-}14}(.),\gamma(.),p(\text{YEAR})$	115.01	0.00	0.3430	1.0000	16	83.01
2	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(\text{YEAR})$	115.01	0.00	0.3430	1.0000	16	83.01
3	$\varphi_{1\text{-}14}(.),\varepsilon(.),p(.)$	117.82	2.81	0.0919	0.2454	3	111.82
4	$\varphi_{1\text{-}14}(.),\gamma(.),p(.)$	117.82	2.81	0.0919	0.2454	3	111.82
5	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(.)$	119.33	4.32	0.0432	0.1153	4	111.33
6	$\varphi(\text{YEAR}),\gamma(.),\varepsilon(.),p(\text{YEAR})$	120.52	5.51	0.0238	0.0636	17	86.52

3.4.8 Eastern gambusia

Eastern gambusia were detected in 17 of the 18 sampled sites (94 %) between 1973 and 2022. The best-supported model describing Eastern gambusia occupancy (Model 1, $AIC_{W} = 0.3482$, Table 17), with 35% weight of evidence, assumed fixed extinction, and invasion parameters between time periods, but variable occupancy and detectability between periods.

The results suggest that probability of occupancy has varied through time along with detectability, while probability of extinction (estimate = 0.05) and invasion (estimate = 0.30) have remained similar. That said, the estimate for the probability of invasion of new sites was high, relative to other species examined here. Figure 17 shows logarithmic increase in occupancy between the earliest and latest time periods with the most rapid increases happening between the 1973-1975 and 1996-1998 time periods, followed by more modest gains towards present day. Overall, the changes marking a 78% increase in mean occupancy estimates between the first and last time period (95 % $CI\varphi_{1973\text{-}1975} = 0.19 \pm 0.00$–0.72; 95 % $CI\varphi_{2020\text{-}2022} = 0.85 \pm 0.68$–1.00).

While detectability changed through time there was no consistent pattern. The estimated detection probability across surveys and years was $p = 0.53$ (SE = 0.05). This indicates that, on average, fishing surveys detected Eastern gambusia when they were present at Yarra River sites a little more than 1/2 of the time, and thus missed detecting them about 1/2 of the time. This would suggest a moderately strong negative bias in naïve estimates Eastern gambusia occurrence in the Yarra River.

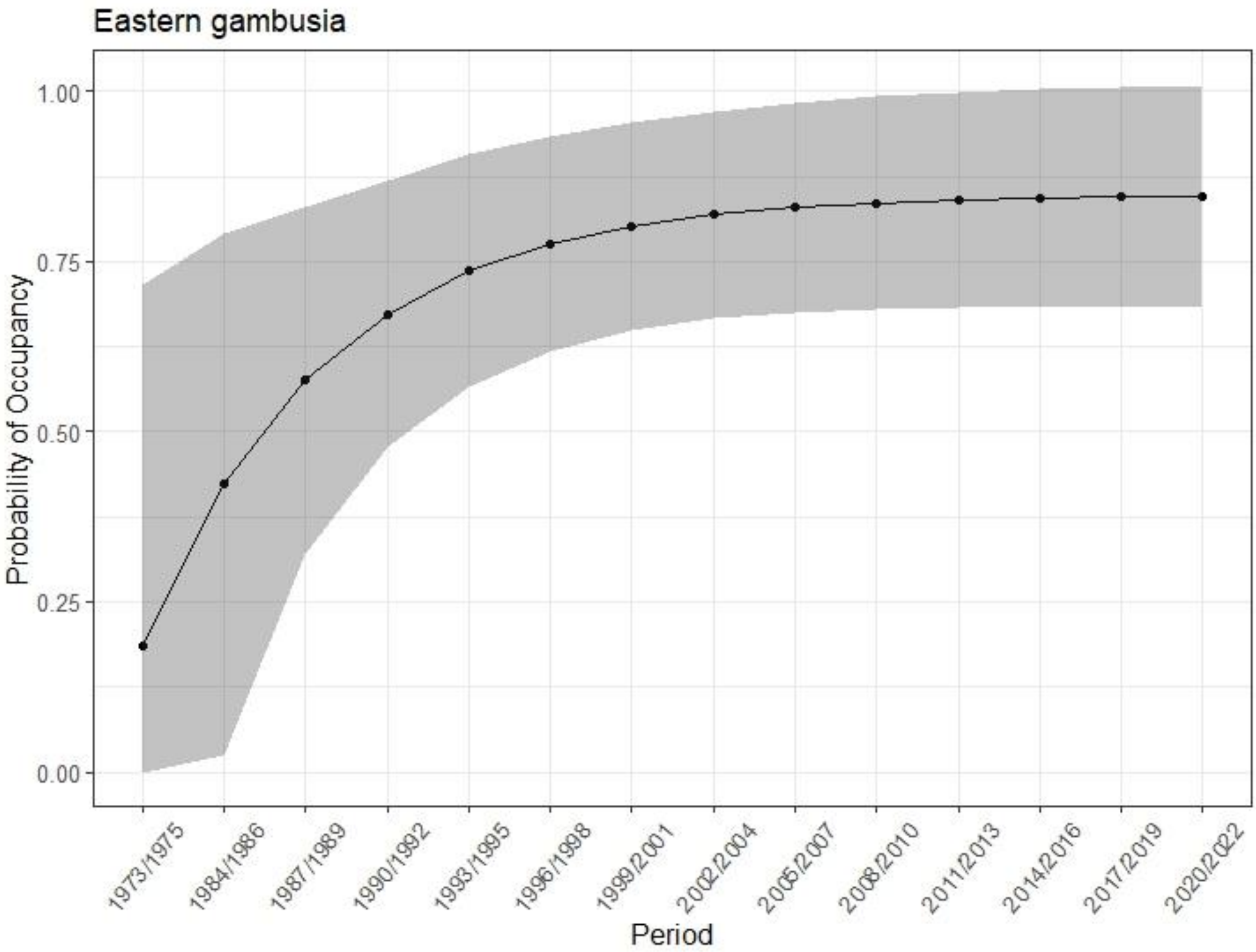
These results suggest that occupancy has increased through time, but that increase has plateaued towards present day as the species reached almost full occupancy. However, the increase was not significant due to wide confidence intervals around the earlier estimates. While detectability varied between time periods, it did not correlate with changes in occupancy suggesting that they are influenced by different factors.

Table 13 Occupancy models of Eastern gambusia in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	φ(YEAR),γ(.),ε(.),*p*(YEAR)	376.31	0.00	0.3482	1.0000	17	342.31
2	φ_{1-14}(.),γ(.),*p*(YEAR)	376.45	0.14	0.3247	0.9324	16	344.45
3	φ_{1-14}(.),ε(.),*p*(YEAR)	376.45	0.14	0.3247	0.9324	16	344.45

Figure 17 Model-averaged estimates of annual probability of site occupancy (φ_t) for Eastern gambusia.



Models 2 and 3 received reasonable support ($AIC_W = 0.3247$, Table 17), with 32 % weight of evidence each and being only 0.14 AIC units different from model 1. These models assumed occupancy (estimate = 0.79), invasion (estimate = 0.24), and extinction (estimate = 0.06) parameters were constant through time, while detectability varied between years. The invasion probability estimate was high relative to other species studied. These models support the hypothesis that occupancy has not changed through time. That said, this is contrary to expectations given the species was introduced to the Yarra River Basin in 1970s and would be expected to expand its range.

3.4.9 Oriental weatherloach

Oriental weatherloach were detected in 13 of the 18 sampled sites (72 %) between 1973 and 2022. The best-supported model describing Oriental weatherloach occupancy (Model 1, $AIC_W = 0.7014$, Table 18), with 70% weight of evidence, assumed fixed extinction, invasion, and detectability parameters between time periods, but variable occupancy between periods.

The results suggest that probability of occupancy has varied through time, while probability of extinction (estimate = 0.04) and invasion (estimate = 0.16) have remained similar. The estimate for the probability of invasion of new sites was moderately high, relative to other species examined here.

Separate to this analysis, the fish distribution database showed the first observation of Oriental weatherloach in the Yarra River Basin was made in 1983. With that in mind, Figure 18 shows a statistically significant logarithmic increase in occupancy between the earliest and latest time periods that appears to be tapering off. Overall, the changes mark a 91% increase in mean occupancy estimates between the first and last time period (95 % $CI\varphi_{1973\text{-}1975} = 0.06 \pm 0.00–0.34$; 95 % $CI\varphi_{2020\text{-}2022} = 0.76 \pm 0.56–0.95$).

The estimated detection probability across surveys and years was $p = 0.54$ (SE = 0.04). This indicates that, on average, fishing surveys detected Oriental weatherloach when they were present at Yarra River sites a little more than 1/2 of the time, and thus missed detecting them about 1/2 of the time. This would suggest moderately strong negative bias in naïve estimates Oriental weatherloach occurrence in the Yarra River.

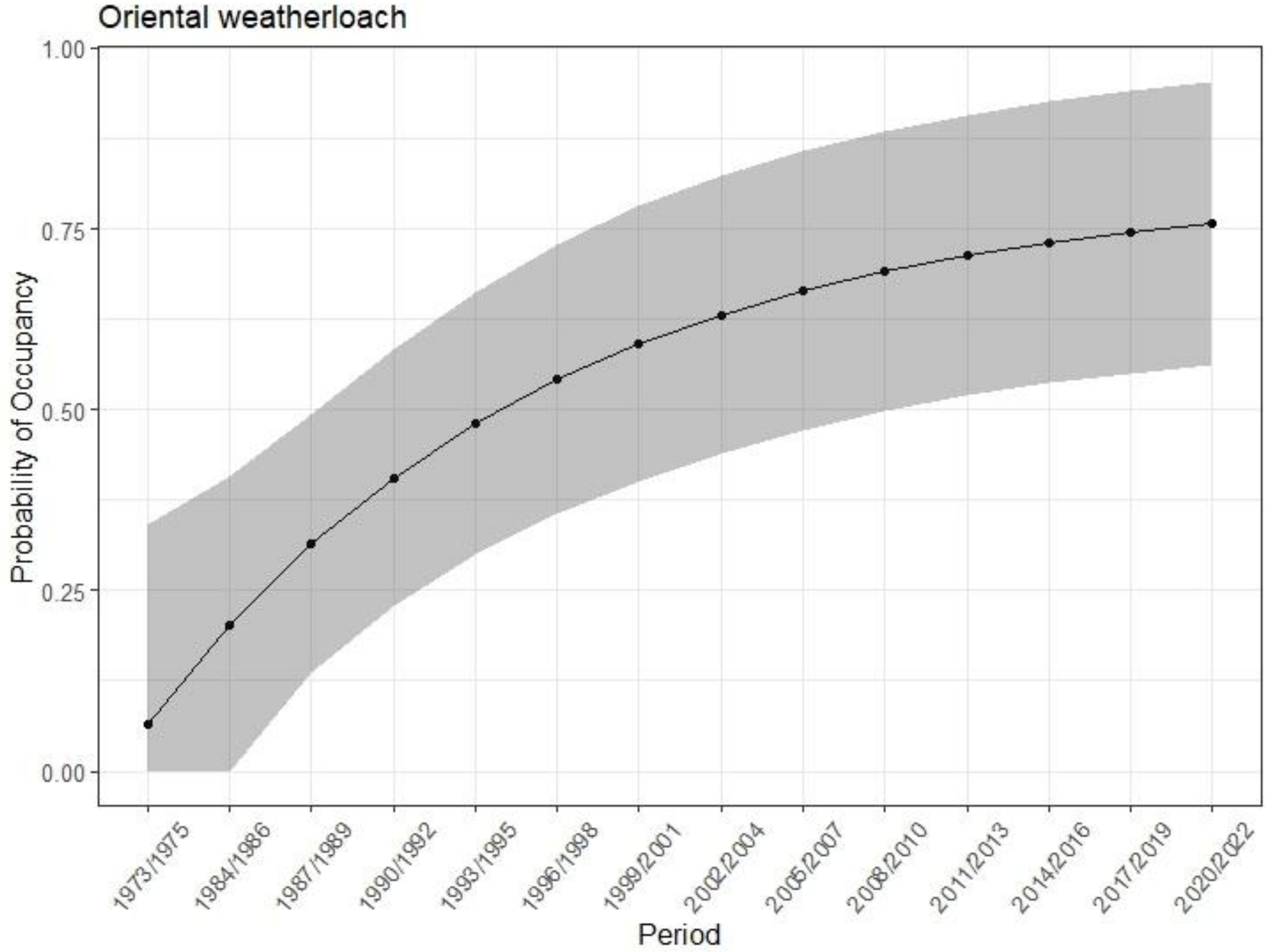
These results strongly suggest that occupancy has increased through time and that increases is beginning to plateau as the species reaches full occupancy of the river basin.

Table 14 Occupancy models of Oriental weatherloach in the Yarra River Basin, 1973–2022. Occupancy (φ) was modelled over time with estimates of river recolonization (γ) and extinction (ε), which were either constant (.), varied among YEARs. Probability of detection (*p*) was also constant (.) or varied by YEAR. The different model parameterizations follow Mackenzie et al. (2003). Only the highest scoring models that made up 0.99 AIC weight are presented.

ML = Model likelihood, based on AIC weights; #para = Number of parameters in the model; 2LL = -2 log likelihood of the model.

Model#	Model parameterization	AIC score	ΔAIC	AIC weight	ML[a]	#para[b]	2LL[c]
1	φ(YEAR),γ(.),ε(.),*p*(.)	332.51	0.00	0.7014	1.0000	4	324.51
2	φ(YEAR),γ(.),ε(.),*p*(YEAR)	335.04	2.53	0.1980	0.2822	17	301.04
3	$\varphi_{1\text{-}14}$(.),γ(.),*p*(YEAR)	338.19	5.68	0.0410	0.0584	16	306.19
4	$\varphi_{1\text{-}14}$(.),ε(.),*p*(YEAR)	338.19	5.68	0.0410	0.0584	16	306.19
5	$\varphi_{1\text{-}14}$(.),ε(.),*p*(.)	341.52	9.01	0.0078	0.0111	3	335.52
6	$\varphi_{1\text{-}14}$(.),γ(.),*p*(.)	341.52	9.01	0.0078	0.0111	3	335.52

Figure 18 Model-averaged estimates of annual probability of site occupancy (φ_t) for Oriental weatherloach.



3.5 Length-weight data

We investigated the availability of length and weight data available within the dataset for each of the priority species listed in the MERI to assess the ability to develop length-weight population health metrics that would allow an assessment of change through time. We present tables indicating the availability of data for each of the priority species over the last decade in the appendices (Tables 15 to 25) and discuss the general findings below for the sake of brevity. It's important to note that priority estuarine species such as Estuary perch, Black bream, Pale mangrove goby, and Glass goby have not been detected in the MWMR in the last decade. While these species may migrate in and out of the lower reaches of river systems, they are predominantly found in estuarine habitat where there is a general lack of sampling effort, so this result is not unexpected. It follows that length-weight analysis is not possible for those species.

In almost all cases there was insufficient data recorded to develop such metrics with any consistency through time. In most cases, length and weight were not entered in the database. Otherwise, they were entered as a range (i.e. minimum to maximum) of lengths and weights, or as length only. In several cases, too few records were available for a given year to conduct any meaningful analyses. As such, we did not pursue any such analysis here.

It is likely that there is considerably more length and weight data recorded with surveys of the MWMR, but they have not been uploaded into the VBA database due to formatting peculiarities. We elaborate on this in the Discussion section.

4. Discussion

The aim of this report was to investigate the long-term fish distribution dataset provided by Melbourne Water and, where possible, to use it to answer the three key questions: (1) How has species richness and nativeness changed over time at the sub-catchment and catchment scale?; (2) How has populations health changing over time for priority species highlighted in the MERI framework?; and (3) Is there any evidence of range expansions or contractions over time using all data? Based on the data available, these questions could be answered with varying effectiveness. In short, detailed information on individual fish such as length, weight and catch per unit effort was lacking, effectively ruling out any meaningful analysis of changes in population health. However, the dataset is rich in species presence records that that have considerable spatial and temporal breadth and can be used to explore trends in, and drivers of, species occupancy through time. Here we used multi-season species occupancy models to do this. Below we discuss our findings of our investigations and analysis and propose further avenues of research using the dataset.

4.1 Dataset summary

The dataset contained records from as early as 1900s, although records were sparse and inconsistent between years until the roughly1980s, depending on the river basin. As such the data gathered since 1980 has the greatest utility in quantitative occupancy analysis. Our summary of the dataset herein focusses on its utility for this that type of analysis.

An important point of note, gathered from the dataset, is that electrofishers began being used in 1964 and were the dominant survey type from the 1970s onwards. Electrofishing marked a significant leap in sampling effectiveness and standardisation in freshwater environments, with the method being effective at catching all species under most conditions. This provides some confidence that sampling method would have a limited influence on any observed changes in site occupancy.

At the river basin level, the Yarra stood out as having the longest and most consistent annual survey record in the MWMR, and the largest mean number of sites sampled in each year (44). Furthermore, the basin consists of 18 sub-catchments which have been sampled with some degree of consistency since the mid-1970s leading us to conclude that the dataset would be suitable for an analysis of changes in occupancy through time. As such we made it the focus of our occupancy modelling.

The next best sampled basin was Western Port, which had a near continuous annual survey record since 1970 and the second largest mean number of sampled sites per year (26). The basin is the largest within the MWMR and consists of seven 'primary catchments' that flow into Western Port which have varying degrees of connectivity between each other. The consistency and number of sites sampled across each of these primary catchments is thought to be sufficient for occupancy modelling. In this way, each primary catchment would constitute a site and the results would indicate how occupancy has changed at the primary catchment level across the broad Western Port basin. This would present a coarser spatial scale of analysis compared to the sub-catchment analysis of the Yarra basin but would provide important insight into changes in species distribution non-the less.

The Werribee, Maribyrnong, and Dandenong catchments received far more sparse sampling than the Yarra and Western Port at all spatial scales. The data would be sufficient for assessing changes in occupancy at the basin level, but any analysis done at primary or sub-catchment level within those basins would incorporate a large amount of missing data which may lead to potentially misleading results and certainly very wide confidence intervals.

These catchments may be divided into spatial units other than primary or sub-catchments such as lower, mid, and upper catchment zones to help overcome the limited spatial distribution of the data, whilst maximising its informativeness. In this way an individual river basin level analysis, like that conducted for the Yarra River Basin, would likely not be possible due to the small number of sites. However, this data would be extremely helpful in in a broader analysis of changes in occupancy across the whole MWMR.

By using courser spatial units, changes in species occupancy would have to be quite dramatic to be detected. For instance, if a river basin was split into lower, mid, and upper catchment sites, a species would have to disappear from 1/3 of the entire basin before that decline was detected. However, with migratory species, particularly the rare and threatened Australian grayling, appearance and disappearance from basins would be expected depending on the prevailing environmental conditions (particularly flows) and maintenance of fish passage. We anticipate that a MWMR wide analysis would be quite effective in detecting population-level changes in such migratory species.

We note that our analysis does not consider the spatial distribution of sites within river basins or sub-catchments, but this at the scale of the proposed analysis, this is not considered greatly important. Also, we considered looking at a narrower time period (i.e. the last decade) if it meant that more records were available across the river basins, but that wasn't the case. Some of the most data poor years occurred in the last 10 years.

4.2 How has species richness and nativeness changed over time at the sub-catchment and catchment scale?

The database indicated that exotic species have had a long history in the MWMR, with a number of European sports fishes being present from at least the early 1900s. Several other invasions followed, with most of the current exotic fauna being in place by 1983. As regular fish monitoring started around the 1980s in most MWMR rivers basins, it is difficult to identify changes in river basin nativeness before and after the introductions, based on species richness alone. Overall, nativeness scores were the lowest on average in the Dandenong Creek basin and highest in the Western Port basin.

While aquatic habitat in all river basins within the MWMR have been degraded by drainage diversions, piping, channel incision, concrete-lining, levees, weirs, vegetation clearing and changes in hydrology related to urban and industrial development, Dandenong Creek is disproportionally more impacted, at least in part because it is a smaller catchment that lies predominantly within heavily developed areas (Koehn 1986). On the other hand, the Western Port basin lies outside the main urban areas of Melbourne and the main land use is agriculture, although this is changing. Its waterways are, on the whole, less modified and in better condition than those in Melbourne's urbans centres (Melbourne Water 2018). Furthermore, the Dandenong Creek and Western Port basins drain into separate ports, and there would be limited opportunity for obligate freshwater species to disperse from Port Phillip catchments to Western Port catchments around the Mornington Peninsula or over the drainage divide. It is likely that this disconnect between Western Port and the more heavily developed Port Phillip region has helped to maintain a predominantly native fish fauna.

A nativeness metric based on species abundance would be a more sensitive and informative approach to assessing change in fish communities over time. Sufficient abundance data may exist in the dataset, over the last two decades in particular, but further interrogation of the dataset is required. It was considered beyond the scope of this project to investigate it further here. A nativeness metric based on biomass would also be helpful but is not possible given the lack of weight data in the dataset.

4.3 How has population health changing over time for priority species highlighted in the MERI framework?

Commonly applied metrics for monitoring fish population health include analysis of the length and weight of fish, which provides an indication of their condition, and analysis of species abundance which indicates the trajectory of the population (Pope et al. 2010). Abundance estimates must be standardised by the amount of fishing effort (e.g. the amount of electrofishing time applied at a site) so that they are comparable between years.

We found that length and weight data for individual fish were rare in the dataset so changes in population health through time could not be assessed using length-weight metrics. This information is rarely uploaded into the VBA as it must be entered in a particular, time-consuming format, that suits the VBA system. Each individual fish record much be entered as an individual line with a corresponding length and weight (e.g. following the Jacobs (2022) Healthy Waterways Strategy fish monitoring program). However, this is more time consuming than entering a single line for a species detection at a site that simply provides an abundance and size and weight range for all individuals caught, which many opt to do. As there are no strict guidelines for data entry, it is also common for entries to only include occurrence and not abundance, length, or weight data. It is likely that there is considerably more length and weight data recorded with surveys of the MWMR. For instance, ARI submit fish catch data to the VBA on a regular basis, but due to formatting incompatibility with the VBA database, this data has not been uploaded (pers comms. Mike Nicol, ARI).

While species abundance data was regularly recorded in the database, fishing effort was not so catch-per-unit-effort could not be calculated and changes in relative abundance between years could not be assessed.

Maintaining a database of abundance (relative to fishing effort), length, and weight data would greatly aid in the assessment of fish community health through time and would thus help inform effective management of those communities. Our analysis of the current long-term database has highlighted that the issues with the upload of data have led to an unfortunate missed opportunity. If this issue is to be overcome, the following steps are needed:

- Better communication between DELWP (VBA) and fish ecologists regarding data entry guidelines and ideally the development of a simpler method for uploading such data.
- Fishing effort should be accommodated in the database.
- Key agencies in charge of freshwater ecosystem management (e.g. MW and ARI) should advocate for the correct uploading of abundance, fishing effort, length, and weight data and insist that consultants follow such procedures moving forward.

4.4 Is there any evidence of range expansions or contractions over time using all data?

Our analysis of changes in species occupancy in the Yarra River Basin between 1973 and 2022 indicated three main trends in species range size: (1) the range size of native obligate freshwater is in equilibrium or is contracting; (2) the range size of native migratory species is in equilibrium or is expanding; and (3) the range size of more recently introduced exotic species has been expanding but has reached or is close to reaching equilibrium. Most of the observed trends were not statistically significant, owing in part to large confidence intervals around occupancy estimates, but in these cases the trends are consistent over almost five decades of data collection, and meet a-priori expectations, suggesting that they are reflective of actual changes in species occupancy. More data would be required to reduce the observed error and produce more precise estimates, but such data is unlikely to appear. The dataset, as it is, is likely to only be suitable for detecting statistically significant changes when those changes are dramatic. However, expansions and declines in species occupancy are more likely to occur gradually and detecting gradual changes is still important in species management. We discuss the results with this in mind, commenting on observed trends, regardless of their statistical significance, but bearing in mind the uncertainty around the results.

4.4.1 Native obligate freshwater species

Of the native obligate freshwater species, Southern pygmy perch exhibited the steepest decline in occupancy relative to their historic distribution, although the decline was not statistically significant. The decline in occupancy is in part calculated from the number and frequency of detections across the river basin, but the species does appear to have disappeared from Darebin and Merri Creek where it hasn't been observed since 1987 and 1988 respectively. This would contribute significantly to this result. Estimated occupancy in the Yarra basin is now below 50%.

River blackfish also exhibited a possible, non-significant decline in occupancy. Along with various changes in the frequency of detection across the catchment, the species has not been detected in Plenty River or Merri Creek since 2007 and 2009 respectively, which would help drive this pattern. The results suggest that the rate of River blackfish occupancy in the Yarra River Basin has also fallen below 50%.

While no change was detected in Ornate galaxias occupancy, the species has likely experience significant range contractions prior the study period. The species is part of southeast Australia's Mountain galaxias complex, which is known to be highly susceptible to trout predation. Most species within the complex can not co-exist with trout and only exist in small remnant populations in trout-free refuges (Raadik 2014). Given that Brown trout were introduced into the MWMR in the early 1900s, it seems likely that the declines in range of Ornate galaxias have already occurred and the remaining populations persist in refuges where they are protected from direct trout predation. They have thus found equilibrium for now. The results suggest that the rate of occupancy for Ornate galaxias in the Yarra River Basin is a little over 50%.

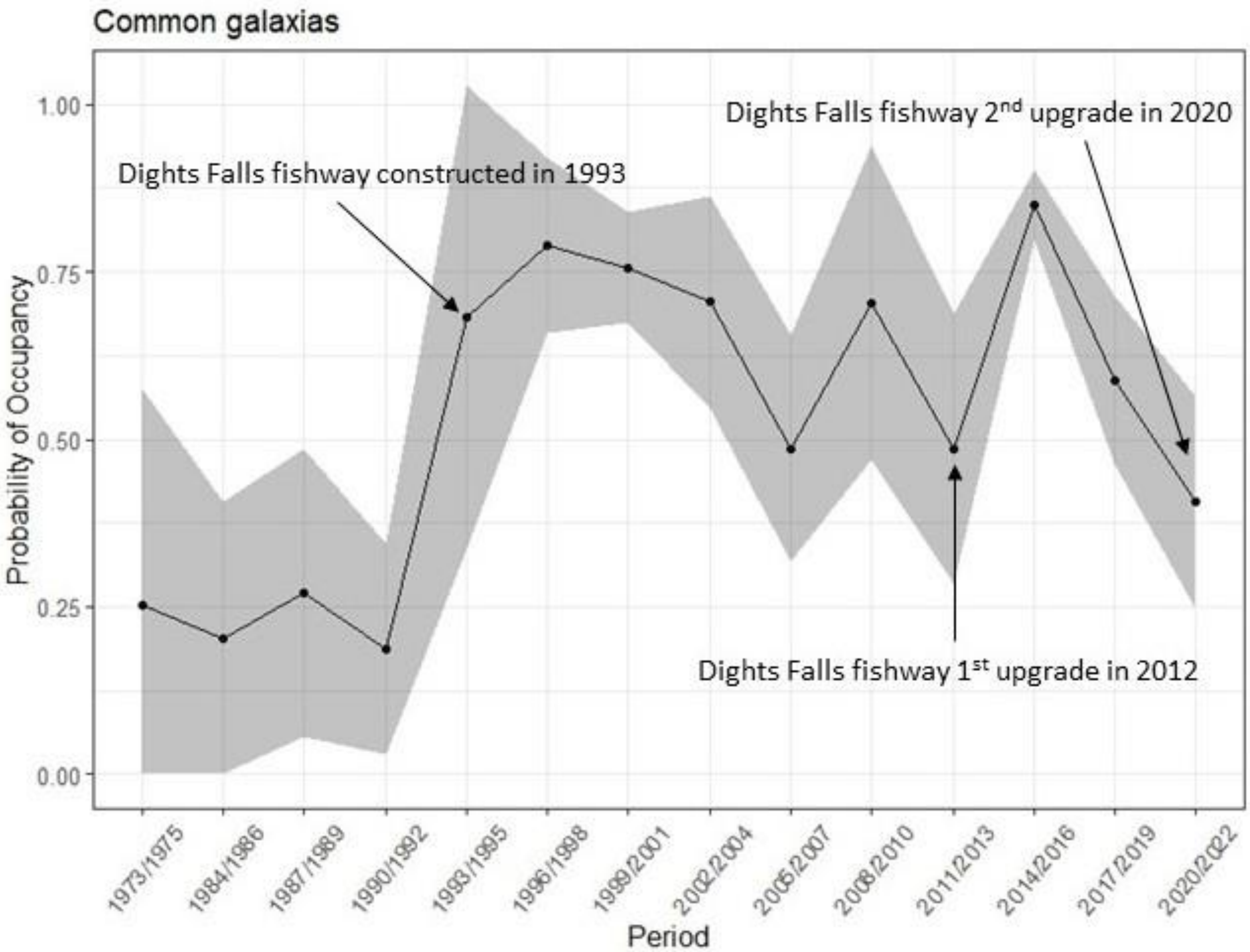
4.4.2 Native migratory freshwater species

Common galaxias were the only species where occupancy estimates didn't follow a set trend through time. Rather, there were two statistically significant increases between consecutive time periods, and one significant decrease. While we did not attempt to incorporate environmental variables into our analysis to help explain these patterns, such a dramatic increase in occupancy across the basin would suggest something had changed in a short period of time that made a large amount of habitat available to the species that previously was not.

Common galaxias are an amphidromous species where adults migrate downstream to lays eggs high up on riverbanks, amongst vegetation, near the coast. The eggs hatch when flood waters inundate the eggs and the larvae are swept out to the marine environment where they are carried by the prevailing currents as they feed and grow. They then migrate to the closest river and travel upstream, spreading widely throughout catchments (Augspurger et al. 2017). This migratory lifestyle leaves them susceptible to barriers to migration that can prevent upstream movement, but it also means they can be quick to recolonise newly available habitat.

With this in mind, the Dights Falls weir on the main channel of the Yarra River, in the lower part of the basin, was historically a barrier to upstream fish movement under most conditions. In 1993 a rock fishway was installed to better facilitate upstream movement of fish. The fishway was in operation for many years, but monitoring efforts suggested that it had become largely ineffective, and it was upgraded to a vertical slot fishway in 2012. Continued monitoring indicated that the new fishway was only operating adequately for most species under base flow conditions and not during floods. In response, the fishway was again upgraded in 2020 (pers coms. Matthew Jones, ARI). The two significant increases in Common galaxias occupancy coincided with the installation and initial upgrade of the fishway and these seem like a likely explanation (Figure 19). The reason for the recent decline in occupancy is less clear, but it may be that high flows have coincided with upstream migration of Common galaxias and they have not been able to overcome the barrier under those conditions. Furthermore, it may be that the most recent upgrade of the fishway, that would is expected to overcome this issue, has not yet been reflected in the dataset. It's important to note that focussed monitoring of the fishway has indicated that it is effective and being used by Common galaxias under most flow conditions (pers coms. Matthew Jones, ARI).

Figure 19 Model-averaged estimates of annual probability of site occupancy (φ_t) for Common galaxias with Dights Falls fishway construction and upgrades indicated.



Short-finned eel are a catadromous species were adults migrate far out to sea for spawning. The adults then die but the larvae return to their parents home streams, during which time they develop into juveniles and migrate upstream throughout the catchment over winter and early spring (Allen et al. 2002). In this sense they would also be subject to barriers to migration like Common galaxias. However, they are have considerable capacity to leave the stream environment and navigate across land to circumvent barriers. As such, it is not expected that the Dights Falls weir would have the same impact on their distribution.

Short-finned eel exhibited a large, but non-significant increase in occupancy between the 1973-1975 period and the 1984-1986 period, after which they consistently occupied the entire Yarra River Basin. As there was only a single time-period where Short-finned eel did not occupy all surveyed Yarra sub-catchments, we can only speculate as to the cause of this pattern. Short-finned eel recruitment is strongly affected by seasonal changes in temperature and rainfall, and it may be that conditions had been poor in the river basin and their abundance and distribution was impacted. As only a fraction of the Yarra River Basin sub-catchments was sampled in that period alone (10 of 18 sub-catchments), and the results are so stark compared to subsequent years, the confidence intervals are particularly wide around the estimates for this period. Regardless of the agent of change, the species has continued to do well since that time.

Tupong are also a catadromous species whose lifecycle broadly reflects that of the Short-finned eel. However, it typically occupies estuarine environments or the lower sections of streams. They potentially experienced a small, non-significant increase in occupancy over the study period. As the species typically occupies the lower parts of catchments, large increases in occupancy would be unlikely as upstream habitat is not suitable. There were no new observations of the species above Dights Falls since the construction of the fishway, suggesting that it was not a factor driving the result. Rather, the results reflect an increase in the frequency of observations in the lower Yarra River basin. Improvements have been made in environmental flows, water quality and habitat in the Yarra River basin since the initial 1973-1975 time period, which could explain the improvement, but further analysis is needed to clearly distinguish any such driver.

The Pouched lamprey is a catadromous species that also typically occupies the lower parts of catchments. No significant change in occupancy was observed with this species, which would suggest that it is in equilibrium in the Yarra River basin.

4.4.3 Exotic species

We analysed changes in occupancy in the two exotic species, Eastern gambusia and Oriental weatherloach that were introduced in 1973 and 1983 respectfully. Each of these species are habitat and dietary generalists, are highly tolerant of poor water quality, and are highly fecund, producing thousands of young throughout the year. Such ecological characteristics have allowed them to rapidly invade a great diversity of habitats across Australia and more broadly around the globe (Moore et al. 2008; Lintermans et al. 2014). As would be expected, each species exhibited evidence of a large increase in occupancy through time in the Yarra River Basin that is beginning to taper off as the number of unoccupied catchments, which they can potentially invade, dwindles.

Eastern gambusia are an aggressive species that compete for habitat with several small-bodied native fishes and damage them by nipping their fins (Moore et al. 2008). They have a significant negative impact on the occurrence, abundance, and condition of Southern pygmy perch where they occur in sympatry (Macdonald et al. 2012). It seems likely that the dramatic increase in occupancy of the species has at least in part responsible for the observed decline in occupancy of Southern pygmy perch in the river basin.

The results also provide insight into the speed at which such invasions take place and emphasises the importance of early intervention (e.g. eradication efforts) when new exotic species are detected.

4.4.4 Patterns in migratory versus non-migratory species

The observed difference in the general population trajectory of migratory and non-migratory species is reflective of national trends in the conservation status of freshwater fishes (Le Feuvre et al. 2016; Lintermans et al. 2020). Generally speaking, non-migratory species have narrower habitat and dietary niches, are less vagile, and are inherently more limited in their ability to disperse within and between river networks than migratory species (Stevens et al 2013; Le Feuvre et al. 2021).

Being entirely restricted to freshwater environments non-migratory species have limited capacity to avoid deteriorating environmental conditions. If the changing environmental conditions exceed the ecological tolerance of a species, the impacted population may go extinct and recolonisation may be slow or not possible. This often leads to the development of fragmented populations across catchments, with the isolated populations having a heightened risk of extinction due to their small size and often low genetic diversity (e.g. Brauer et al. 2016). Such patterns are observable in River blackfish, Southern pygmy perch, and Ornate galaxias, among others.

Migratory species have broad distributions across landscapes (i.e. they occupy multiple catchments), thus the risk of decline from changing environmental conditions is spread across a greater area. Furthermore, if conditions improve in a given area, they can be quick to recolonise as they are often rheotactic, having an innate sense to swim upstream, against the flow, as returning juveniles (Lotze et al. 2011). That said, this response is not as strong in lowland species that stay close to the coast.

The positive increases in occupancy observed in some of the migratory species analysed here likely reflects improvements in river management that have been made over the last five or so decades such as improved fish passage, environmental flow regimes, and habitat restoration. It's an important and heartening result.

The potential declines observed in non-migratory species are unfortunately not unexpected and they reflect the trajectory of these species elsewhere in Victoria, and elsewhere. It's important to note that they were not rapid or particularly large declines, rather reflecting a steady decline. It follows that targeted monitoring of key populations should be conducted to assess finer measures of population health, so that management interventions can swift in order to save ones that heading towards extinction. This is already recognised and is happening through programs such as the HWS. The pressures on these species are fairly well described and include predation and competition with exotic species, altered flow regimes and drought conditions, and habitat alteration. While some work is being done to address these issues, it appears that greater effort is needed.

4.5 Conclusions and recommendations

The collation of fish survey results into a central database such as the one dealt with here provides an important resource for managers who need to retrospectively assess the effectiveness of their management efforts in order to improve them moving forward. Even when such a database is imperfect or incomplete, occupancy modelling offers an effective method for utilizing historic presence and absence data for species for which little other data exist. The insight the analysis provides into large-scale and long-term population changes can provide a strong indication of weather river restoration or conservation efforts are working, and the positive trajectory observed among several of the migratory species is reflective of positive change. It also helps to identify species that are not responding well to management efforts, so additional efforts can be made to understand and arrest such declines.

The analysis stresses the importance of maintaining regular annual monitoring across the MWMR. Years with missing data impact significantly on the sensitivity of the analysis and confidence in the model estimates, reducing the utility of the dataset as a whole. We strongly recommend that a base level of monitoring be maintained by MW to improve the veracity estimates in the future. The current HWS fish monitoring program provides an excellent example a robust monitoring program for the region. In particular, it includes sites across each of the primary management basins, many of the sub-catchments, and upstream and downstream sites in most cases catchments that allows for the detection of changes in fish passage.

The analysis presented here should be considered preliminary as it only aims to describe trends in occupancy, rather than attempting to determine biotic (e.g. exotic species) and abiotic (e.g. stream flow metrics) drivers of those trends. We recommend that the analysis be extended so that it incorporates co-variates that might explain the observed pattens. The amount of environmental flow releases in a given time period for instance, may correlate with an increase in migratory species such as Australian grayling, or the length of low flow conditions may correlate with a decline in wetland specialists such as the Southern pygmy perch. Such information would provide greater insight into the effectiveness or ineffectiveness of management efforts.

Furthermore, this analysis uses the Yarra River Basin as a test case as it was the most data rich river basin within the MWMR. We recommend that the analysis be broadened across the whole MWMR so that large-scale trends can be assessed. This would be particularly important for assessing the lowland migratory species, such as Australian grayling, for which river basin scale occupancy trends are more difficult to detect than regional scale trends. Where data was scares at finer spatial scales, river basins could be divided up into broader spatial units such as lowland, mid, and upland regions to aggregate the data.

To summarise are recommendations are:

- MW should aim to maintain a base level of monitoring to improve the veracity occupancy estimates in the future.
- The occupancy models should be extended to incorporates co-variates (e.g. environmental flow releases and exotic species presence/richness) that might explain the observed pattens.
- The analysis should be broadened across the whole MWMR so that large-scale trends can be assessed. This would be particularly important for assessing the migratory species, such as the threatened Australian grayling.
- Steps should be taken to better facilitate the upload of catch-per-unit-effort, length, and weight data into the MW and VBA databases.

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Appendix A. Additional information

Table 15 Availability of River Blackfish length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014					
2013					

Table 16 Availability of Ornate galaxias length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015			Length only		
2014					
2013					

Table 17 Availability of Common galaxias length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014			Length only		
2013			Length only		

Table 18 Availability of Yarra pygmy perch length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available, Grey = no records from catchment.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014					
2013					

Table 19 Availability of Southern pygmy perch length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015			1 record		
2014					
2013					

Table 20 Availability of Dwarf galaxias length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available, Grey = no records from catchment.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014					
2013					

Table 21 Availability of Australian grayling length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019			1 record		
2018					
2017					
2016					
2015					
2014			Just length		
2013			Just length		

Table 22 Availability of Macquarie perch length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available, Grey = no records from catchment.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014					
2013					

Table 23 Availability of Tupong length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019			1 record		
2018					
2017	1 record				
2016		Length only			
2015					
2014			Length only		
2013			Length only		

Table 24 Availability of Short-finned eel length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					
2020					
2019					
2018					
2017					
2016					
2015					
2014			Length only		
2013			Length only		

Table 25 Availability of Short-headed lamprey length-weight data between 2013 and 2022 in each of river basin within the Melbourne Water management region. Green = length-weight data available for individual fish, Blue = range of lengths available (shortest-longest), Orange = no length-weight data available.

Year	Werribee	Maribynong	Yarra	Dandenong	Western Port
2022					
2021					1 record, length only
2020					
2019					
2018					
2017					
2016					1 record, length only
2015					
2014			Length only		
2013					