Connectivity, Fire, and Land Use: Understanding Platypus (*Ornithorhynchus anatinus*) Persistence in Fragmented Watersheds

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Key Words:

Connectivity, distribution, environmental DNA, fire, freshwater, habitat fragmentation, land use, macroinvertebrates, multiple stressors, platypus

Abstract

Aim Effective biodiversity conservation requires improved understanding of species distributions, and of the influence of threatening processes on those distributions. This is particularly important for freshwater species, which are difficult to survey even as they are exposed to disproportionately high levels of threat. Here we address this issue for the platypus (*Ornithorhynchus anatinus*), an evolutionarily-distinct, ecologically-important species whose threat status is jurisdiction dependent and subject to conjecture. We aim to improve our understanding of platypus distribution across a large, little-studied region, and to quantify the association between occurrence and environmental metrics, including those thought to influence population viability.

Location Central-eastern Australia, including south-eastern Queensland and northern New South Wales.

Methods We used environmental DNA to survey for platypus at 174 sites distributed across 43 waterways and seven river basins in the central part of the species' range. We related platypus occurrence to local food availability (deduced from our multispecies eDNA samples), and to remotely-sensed data on water availability, river fragmentation, recent fire severity, land use, vegetation cover, and topographic metrics.

Results Platypuses were not detected at 89 sites (51%), of which 56 had nearby historic sightings (within 25km), suggestive of a potentially significant range decline. We also detected platypus in seven locations with no known historical sightings, likely reflecting previously undocumented populations in areas with low historical survey effort. Platypus had lower occurrence in south-eastern Queensland (in the north). Occurrence was positively related to unobstructed river length and had scale-dependent associations with fire severity.

Main conclusions Our findings highlight the urgent need to improve our understanding of the conservation status of this iconic, cryptic species. Improving our knowledge of the

viability of populations in heavily fragmented watersheds and in response to extreme events such as droughts and fire is crucial for developing conservation strategies to halt further declines.

Introduction

Building a complete understanding of the determinants of species occurrence, and particularly the role of threatening processes, is critical for effective environmental management and species conservation (Boulangeat et al., 2012; Olden et al., 2008; Pullin, 2002). A priority in this context should be to reinforce and extend what is known about the drivers of species distributions to relatively understudied geographic areas and across larger spatial and temporal scales. It is also important to quantitatively refine the roles of putative threatening processes. Achieving these objectives has often been difficult, and even more so for species that are difficult to detect and/or occupy habitats that are intrinsically difficult to survey (Ducros et al., 2023; Ficetola et al., 2008).

Such challenges are particularly acute of many species relying on freshwater streams and rivers (Nogueira et al., 2021; Tisseuil et al., 2013). Not only are these ecosystems and the species they support likely to be exposed to unusually high levels of threat from anthropogenic drivers (Brauer & Beheregaray, 2020; WWF, 2024), but species in these habitats are typically difficult to detect and/or survey (Ficetola et al., 2008). As such, improving our understanding of the drivers of, and threats to, the distribution of freshwater species across large geographic areas is a clear priority for management and conservation (King et al., 2019).

Improving our understanding of species distributions and their drivers is being rapidly facilitated by cost-effective technological advances that allow for biodiversity sampling at scale (Shrestha & Lapeyre, 2018; Wearn & Glover-Kapfer, 2017). A particularly prominent example is the rapid adoption of environmental DNA (or eDNA) for biodiversity surveys (Altermatt et al., 2020; Darling, 2019; Lugg et al., 2018). eDNA is DNA that is extracted from environmental samples such as soil, water, or air (Deiner et al., 2017), and that can then be analysed for the presence of particular taxa based on the presence of taxonomically-unique gene regions (Kress et al., 2015). eDNA has been proposed to be a uniquely powerful tool for improving our understanding of the biodiversity of riverine systems in particular (Altermatt et al., 2020; Dejean et al., 2011). By facilitating the relatively simple detection of otherwise difficult-to-detect species in freshwater systems, high-quality survey data can be rapidly collected at multiple sites and across large geographic areas. Moreover, when paired with information on threatening processes, large-scale surveys can offer general insights into the drivers of species' distributions, complementing the conclusions of more labour-intensive studies that can only be done at a few locations. Such large scale surveys are likely to be particularly valuable for species living in rivers and riverine systems because the long linear and dendritic structure of these systems mediate exposure to threatening processes over very large spatial scales (Bino et al., 2015; Campbell Grant et al., 2007).

It is now well understood that riverine ecosystems are exposed to high levels of simultaneously acting threatening processes, making freshwater systems one of the most imperilled of all ecosystems (Maasri et al., 2022; WWF, 2024). High among these threats are river regulation through damming and water extraction, land-use change, and threats directly and indirectly associated with climate change (Dudgeon et al., 2006; Sanders et al., 2024). River regulation due to the construction of dams now affects 63 percent of flowing freshwater systems worldwide (Grill et al., 2019). Where they occur, dams have large effects on water availability, flow variability, and water temperatures (Bice et al., 2017; Lu et al., 2018; Lugg & Copeland, 2014; Parisi et al., 2020), and their physical structure can strongly impede the ability of freshwater species to move and migrate (Barnett & Adams, 2021).

Meanwhile, while occupying a small area in absolute terms, rivers and streams are exposed to the effects of changing land-use patterns over much larger areas as they trace their way through landscapes (Ding et al., 2019; Pervez & Henebry, 2015). For example, rivers and streams are exposed to the effects of different kinds of runoff as they flow through and drain natural, agricultural, and urban areas (Allen et al., 2021; Dudgeon et al., 2006). Exacerbating these effects on freshwater systems are direct effects of climate change – rising temperatures and extreme events (heatwaves and storms) – and climate-associated effects such as increases in fires (Bixby et al., 2015; Gomez Isaza et al., 2022). Given the range and intensity of threats to which riverine systems are exposed, and the relative paucity of data on the distribution and ecology of many freshwater species (Maasri et al., 2022), it is important to be able to identify threats of most importance, particularly for keystone species and/or species of otherwise high conservation value.

The diversity of rivers and streams tends to be dominated by invertebrates, vertebrate fish and amphibians. By contrast, only a relatively small and strongly paraphyletic group of mammals have become adapted to living in rivers and streams. Despite their relatively low local and global diversity, mammals living in freshwater systems are invariably taxonomically and phenotypically unique (Hood & Brierley, 2020). Moreover, because they are often relatively large and/or typically occupy higher trophic levels, freshwater mammals can have outsized influence on ecosystem structure and function (He et al., 2024). Because of their habitat requirements, and their unique taxonomic and functional status, freshwater mammals are particularly vulnerable to population decline, and a large proportion of freshwater mammal species are of conservation concern (He et al., 2021; Torres-Romero et al., 2024), even as their distribution and ecology is little understood (Sanders et al., 2024). Unique among this unique group of species is the platypus, *Ornithorhynchus anatinus*, one of only five species of egg-laying mammals, and one of only two species of aquatic/semi-aquatic mammal occurring in Australia. Despite its iconic status, and Near Threatened designation by the IUCN (Woinarski & Burbidge, 2016), there remains considerable uncertainty about its distribution and abundance (Bino et al., 2019; Hawke et al., 2019; Hawke et al., 2020; Kolomyjec, 2010). Indeed, most of the research on the distribution and ecology of platypus has been done in the cool temperate areas of southern Australia (Coleman et al., 2022; Connolly et al., 2016; Grant et al., 1999), and often at just a few well-studied sites (e.g. Bino et al., 2015). Even at these sites, and despite confident assertions in the literature, conclusions about the effects of at least some threatening processes (e.g. dams, urbanization) remain equivocal (e.g. Ahrens et al., 2025; Hawke et al., 2021; Mijangos et al., 2022). This leaves large parts of the range of the platypus relatively understudied, and the drivers of and threats to the distribution of the species at least partly unknown.

To address these issues, here we use eDNA to survey for platypus across a large geographic area encompassing 174 sites distributed across 43 rivers and streams and seven river basins in the central part of the known range of this species, including in regions dominated by agriculture, as well as natural and urban areas. We then relate the presence-absence of platypus across this region to a wide range of putative drivers of, and threats to, platypus occurrence, including river fragmentation due to dams, land-use, and fires. Our study represents an important case study for the use of eDNA combined with data on threatening processes across large scales to expand our understanding of the biology of a unique and important component of the global freshwater fauna.

Methods

eDNA sampling and DNA extraction

Multispecies eDNA surveys were conducted at 174 sites across 43 waterways in south-east QLD and northern NSW between September 2022 to June 2023 (Figure 1, see Appendix S1 in the Supporting Information). Approximately two thirds (105/174) of sites were in south-east QLD, largely flowing to the east while the rest of sites (39.6%, 69/174) were within the northern Murray Darling basin, which consists of waterways that flow west and south-west. At each site, three replicate samples were collected using Wilderlab sampling kits (https://www.wilderlab.co.nz/). Each water sample was collected with a syringe, used to draw up 50mL of water from the edge of the waterway, which was then filtered through a 1.2µm filter. This was repeated until the filter was clogged, or 1L of water had been filtered. Excess water was removed and preservative added. Samples were extracted and sequenced for DNA by Wilderlab (https://wilderlab.co/). To detect platypus within eDNA samples, we used a platypus-specific primer that targeted a 57 base pair region of the mitochondrial cytochrome B gene. In addition, we used a cytochrome oxidase subunit 1 assay to detect a broad range of freshwater macroinvertebrates, including those commonly consumed by platypus. A QuantStudio 1 qPCR instrument (Applied Biosystems) containing 10µL SensiFast SYBR Lo-ROX mix (Bioline), 1µL of each primer, 1µL Bovine Serum Albumen (10 mg ml−1, Sigma Aldrich) and 7µL DNA template solution, was used for quality check qPCR reactions. An Internal Positive Control, and positive and negative controls, were included for all assays.

Data was aggregated based on taxonomic ID and the DNA reads from the three replicate samples at each site and converted to presence-absence. Platypus, and macroinvertebrate phyla (Annelida, Arthropoda, Nematoda, Cnidaria, Mollusca and Nemertea), as taxa representing platypus prey (Faragher et al., 1979; Hawke et al., 2022; Klamt et al., 2016; McLachlan-Troup et al., 2010), were selected. Macroinvertebrate family and order richness was determined for all sites. A revised SIGNAL score was calculated based on presence-absence was calculated at the family level based on macroinvertebrate sensitivities to environmental condition (Chessman, 2003; Lush et al., 2025).

Remotely sensed data

We considered variables that may be important for platypus habitat. These included elevation, runoff, Normalized Difference Vegetation Index (NDVI), fire severity from the 2019/20 bushfires, land use and total unobstructed river length (see Appendix S2 in the Supporting Information). Elevation data was collated from the Australian Government branch, Geoscience Australia (Gallant et al., 2009). Zonal statistics in QGIS (2024) was used to calculate minimum elevation in a 100m buffer zone around each eDNA sampling point.

To calculate NDVI, near infrared and red raw data bands were downloaded from Sentinel 2 (European Space Agency, 2022) at high resolution (10m), from November 2021 to February 2022. These data bands were used to calculate NDVI using the raster calculator in QGIS (2024), as (NIR – R)/(NIR+R). The average value of NDVI was calculated for 1km and 2km buffer zones around each eDNA sampling point.

To delineate watersheds for each survey point, we first pre-processed a digital elevation model (Gallant et al., 2009) using WhiteboxTools. Hillshade was generated for visual validation, and the DEM was conditioned for hydrological analysis by breaching depressions and filling sinks. Flow direction and accumulation layers were derived using the D8 pointer and flow accumulation algorithms. Survey points were snapped to the nearest stream channel, and corresponding watersheds (i.e. total upstream contributing area) were delineated using the wbt_watershed function. We followed this workflow using a series of operations in R, incorporating the packages 'tidyverse', 'raster', 'sf', 'whitebox', 'tmap', and 'terra' (Hijmans et al., 2025; Hijmans et al., 2024; Pebesma et al., 2025; Tennekes et al., 2025; Wickham, 2023; Wu & Brown, 2024). The resulting watershed polygons were used to extract relevant environmental variables for each survey site.

Runoff data was sourced from the Australian Water Outlook (Australian Bureau of Meterology, 2024) and used to calculate the total annual runoff for each river region for the five years preceding to sample collection. Data was analysed within the R environment (R Development Core Team, 2024) using the 'ncdf4' package (Pierce, 2024) and the 'raster' package (Hijmans et al., 2024), and then clipped to watershed areas (Gallant et al., 2009). Runoff from the year prior to sampling was included as a variable, as was an average of the five years prior to sampling.

Fire severity data for the 2019/20 bushfires was extracted from the Queensland Spatial Catalogue (2020), and from the NSW Department of Climate Change, Energy, the Environment and Water (2020). These data sets were combined. Total area burnt at extreme, high, moderate and low severities was calculated at the watershed scale, and for a 2km buffer around each eDNA sampling site. Extreme, high and moderate severities were summed at each scale to give a measure of burnt area.

Land use data was sourced from the "Catchment Scale Land Use of Australia – Update December 2023" dataset (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023) and extracted at the watershed scale, and for a 1km buffer area around each eDNA sampling site. CL19 land use categories were combined into six broader categories of Agriculture (Dryland cropping, Dryland horticulture, Grazing modified pastures, Grazing native vegetation, Intensive horticulture and animal production, Irrigated cropping, Irrigated horticulture, Irrigated pastures, Land in transition, Rural residential and farm infrastructure), Natural (Managed resource protection, Nature conservation, Other minimal use), Forestry (Plantation forests, Production native forests), Mining (Mining and waste, Other intensive uses), Urban (Urban residential) and Water (Water).

Habitat fragmentation was quantified by calculating the total length of uninterrupted waterway available at each site. Dams were treated as complete barriers to movement, and river networks were segmented accordingly. For each site, the total length of all connected upstream and downstream waterway branches, excluding segments interrupted by dams, was summed to produce a measure of unobstructed river length. This variable was log transformed to account for skew introduced by a small number of sites with extremely long connected waterways. Dam wall locations were sourced from the National Dam Walls dataset (Geoscience Australia, 2016) and river networks were measured using the HydroRivers dataset (Lehner, 2013) in QGIS (2024).

Data analysis

To investigate the environmental and spatial predictors influencing platypus presence-absence (our response variable), we used generalized linear modelling with binomially distributed errors. We employed a multi-step statistical approach that included multicollinearity assessment, model selection, and model averaging. This methodology enabled us to address collinearity issues, systematically evaluate candidate models, and derive robust parameter estimates while accounting for model uncertainty. Before fitting predictive models, we assessed multicollinearity among predictor variables to ensure model interpretability and reliable coefficient estimation (Dormann et al., 2013). To do this, we calculated Variance Inflation Factors (VIF) for all predictors in the dataset. Variables with VIF values exceeding 5 (Dormann et al., 2013; Menard, 2002) were iteratively removed, starting with the one exhibiting the highest VIF, until all remaining predictors had VIF values below the threshold (see Appendix S3 in Supporting Information).

To account for spatial non-independence, we included terms for spatial autocorrelation between sites. Spatial autocorrelation was calculated using the ‘spdep’ package (Bivand et al., 2024), which generates a spatial weights matrix from the geographical coordinates of each site in order to compute a spatial lag vector of similarity between response variable values at sites in close proximity to each other. We calculated weights based on two separate distances between sites, 10km and 45km, aligning with upper estimates of platypus home range size and dispersal distance, respectively (Bino et al., 2018; Serena et al., 1998; Serena & Williams, 2012). The spatial lag vector was used as an auto-covariate in models.

Following the construction of the global model, we performed model selection using the dredge function in the R package 'MuMIn' (Barton, 2024). This function generates all possible candidate models by systematically exploring combinations of predictor variables from the global model. To avoid overfitting, we set a limit of four variables per model. To account for model selection uncertainty and derive robust parameter estimates, we implemented model averaging across the set of candidate models with ΔAICc < 2 (Akaike, 1974; Burnham & Anderson, 2002). Averaged coefficients were weighted by AICc, providing effect size estimates conditional on each predictor's inclusion in the selected models (Grueber et al., 2011). This conditional model averaging approach avoided biases associated with unconditional shrinkage across all models (Burnham & Anderson, 2002; Dormann et al., 2018). All analyses were performed in R version 4.3.2 (R Development Core Team, 2024).

Comparison to sightings data

To evaluate the reliability and ecological significance of eDNA results, we compared them with historical and recent platypus sightings data obtained from the Atlas of Living Australia and other publicly reported sources. Records were classified by recency of observation into four categories: most recent (2018 to 2024), semi recent (2011 to 2017), older (2004 to 2010), and old (pre 2003). Sightings were summarised within 25 by 25 km grid cells across the study area to account for spatial uncertainty in observation records. We conducted two complementary comparisons. First, to assess the potential for false negatives in eDNA detections, we examined whether individual sampling sites that yielded negative eDNA results were located in grid cells containing recent sightings. Second, to identify areas where eDNA non detections may reflect local extinctions, we evaluated whether sites with negative eDNA results were located in cells with only older or historic sightings. All spatial analyses were conducted in QGIS (2024).

Identification of obstructed waterways

We examined the entire mainland platypus distribution to identify obstructed river regions, to highlight areas at risk of local extinctions. As with the measurement of unobstructed river length, we took dam wall locations from the National dam walls dataset (Geoscience Australia, 2016) and measured waterways from the HydroRivers dataset (Lehner, 2013) in QGIS (2024). We did not explore dam surrounds in Tasmania, as platypuses are known to travel over land far more frequently in the cooler climate (Bino et al., 2019; Otley et al., 2000), so dams there may not act as complete barrier to movement. We visually examined the river networks, and for river regions isolated by dams, we summed the total length of all connected upstream and downstream waterway branches, in the same way as for the measurement of unobstructed river length. In doing so, we identified regions around dams with less than the maximum recorded juvenile dispersal distance (45km, (Serena & Williams, 2012)) accessible to platypus. We removed areas if they did not have previous recorded platypus sightings.

Results

Platypus DNA was detected at 85 of the 174 surveyed sites (49%, Figure 1). There were lower rates of detection in QLD than in NSW, with detection rates per catchment ranging from 19.6% - 66.7% in QLD, and 52.6% - 88.8% in NSW. The Border Rivers catchment had sites in both states, with an overall detection rate of 75%, but an 11.1% detection rate for sites within QLD, and 96% detection rate for sites in NSW.

A total of 5036 models were examined, systematically exploring all combinations of variables included in the global model (R^2=0.453). The top-ranked model (AICc=168.13, weight=0.207, R^2=0.405) included Fire at the 2km scale, Water land use at the watershed scale, log unobstructed river length and the 45km spatial autocovariate. Models within ΔAICc < 2 contained various combinations of Fire at the 2km scale, and Fire at the watershed scale, Water land use at the watershed scale, log unobstructed river length, and the 45km spatial autocovariate (See Appendix S4 in the Supporting Information).

The model averaging results identified three predictors significantly associated with the presence of platypus (Table 1, Figure 2). Unobstructed river length and water were positively associated with platypus presence and showed a significant effect ($p=0.004$ and $p=0.025$ respectively), as did the 45km spatial autocovariate ($p<0.001$). These results suggest that both water connectivity and landscape structure are key factors influencing the occurrence of platypus. Fire history at both the local and watershed scale was marginally significant ($p=0.054$ and $p=0.048$ respectively). The other variables (SIGNAL score; macroinvertebrate order richness; macroinvertebrate NMDS; elevation; NDVI; 5-year average runoff and natural, forest, urban and mining land use) did not appear in the top candidate models, and as such were considered to have negligible effects on platypus occurrence.

Of the 35 grid cells containing eDNA sampling sites, only three had recent sightings (2018–2024) but no eDNA detections. These cells included 6, 3, and 1 sites respectively. At the site level, 56 eDNA-negative sites were located in grid cells with recent sightings, suggesting some potential for false negatives. However, in fragmented river systems, where false negatives could directly affect conclusions about habitat connectivity, such cases were rare (See Appendix S5 in the Supporting Information), indicating that overall confidence in the eDNA signal is high. Of the grid cells with negative eDNA results, all either had recent (2018 – present) records, or no historical records, suggesting that sites with older records but lacking current detection were uncommon. In contrast, eDNA identified platypus presence in seven grid cells with no historical sightings on record. These detections occurred along the western edge of the species' known distribution, including parts of the Condamine-Culgoa, Brisbane, Mary, Border Rivers, and Namoi catchments. At the site level, 15 eDNA-positive sites were located in cells with no sightings history.

Throughout the mainland distribution, we identified 50 regions with less than 45km of connected waterway available to platypus (Figure 3, see Appendix S6 in the Supporting Information). Of these, 13 were in QLD, 24 in NSW, 11 in Victoria, and 2 in SA. Five of these regions were in the study area, and fed directly into the model variable of unobstructed river length. The heights of the dams in question ranged from 5 to 166m. This differed state by state (NSW: 14-91m, QLD: 5-63m, VIC: 13-166m).

Discussion

Leveraging the relative ease of environmental DNA for conducting broad scale surveys of cryptic species, we provide new insights into platypus occurrence across a large, under surveyed region of the species' range. By relating detection patterns to variation in land use, fire history, and river connectivity, we have identified a dominant role for river fragmentation and to a lesser extent, fire, in shaping the occurrence of platypus across seven river basins. While consistent with broader concerns about the vulnerability of freshwater biodiversity to multiple stressors (Bixby et al., 2015; Gomez Isaza et al., 2022; Grill et al., 2019; Sanders et al., 2024), our findings add to those of earlier studies that reported equivocal effects of these drivers on platypus specifically (Bino et al., 2021; Bino et al., 2020; Griffiths et al., 2020; Hawke et al., 2021; McColl-Gausden et al., 2023; Serena et al., 2022). Notably, we observed marked spatial variation in detection, with consistently lower occurrence rates in the state of Queensland compared with New South Wales. This suggests potential regional differences in threat intensity, habitat condition, and/or conservation management. Given the widespread fragmentation of rivers, increasing frequency and severity of fires, and the compounding influence of drought leading to reduced water availability, our results have important implications for understanding and managing the persistence of this evolutionarily-distinct and ecologically-important species.

Broad patterns

While broadly consistent with patterns of occurrence inferred from *ad hoc* historical observations (Hawke et al., 2020), our more systematic survey approach fills a critical gap in our understanding of platypus distribution across the lesser-studied central portion of the species' range. Previous research has focused primarily on southern populations (Grant et al., 1999; McColl-Gausden et al., 2024), with long-term data available only from single rivers such as the Shoalhaven (Bino et al., 2015). In contrast, platypus occurrence was markedly lower in the north of our survey area in south-east Queensland, than in the more southerly area in northern NSW. This result raises questions regarding cross-jurisdictional conservation priorities and the consequences of differing state-level protection statuses. These findings also reinforce concerns about a persistent geographic imbalance in research effort, particularly in northern areas where platypuses are genetically distinct (Kolomyjec, 2010). This distinctiveness has clear conservation significance, and our findings of lower occurrence in the north emphasize the need for targeted surveys and protection in these regions to safeguard unique evolutionary lineages.

We detected no platypus DNA at 56 sites that fell within 14 grid cells (25x25km) in which recent sightings of platypus had been reported. Given that our survey relied on single time point sampling, we cannot rule out the possibility of false negatives, but our broader analysis suggests these are relatively rare (Table 2). Moreover, previous research suggests our sampling protocol based on three samples per site allows for platypus detection probabilities exceeding > 0.95 (Lugg et al., 2018), providing additional confidence in our results. Therefore, our findings might plausibly be interpreted as signals of contraction or, at best, low-density persistence. Importantly, low density and/or declining populations are more likely to be vulnerable to threatening processes and to a reduced likelihood of successful breeding and long-term persistence. Thus, these results provide additional important information for spatial prioritization of future survey effort and conservation action. While temporally-replicated eDNA surveys would allow more robust inference about temporal trends and so should be encouraged (Ruppert et al., 2019), eDNA surveys such as ours may still serve as an early warning tool in areas with previous records but repeated absence in new surveys.

In contrast to non-detections within the previously known range of the species (above) we also detected platypus DNA in seven grid cells with no known historical sightings. These cells were located near the western edge of the species' known distribution and likely reflect previously undocumented populations in areas with low historical survey effort. Our study, therefore, expands the known distribution of platypus in under-surveyed regions, demonstrating the power of eDNA to provide a contemporary snapshot of presence across broad spatial scales. When combined with historical data, these surveys contribute to a more complete picture of the species' status and help identify both areas of persistence and emerging local decline.

River Fragmentation

The effects of river fragmentation on freshwater biota are well established (Bice et al., 2017; Grill et al., 2019), and our study demonstrates that these impacts extend to platypus occurrence across large spatial scales. We found a strong positive relationship between unobstructed river length and the probability of detecting platypus (Figure 2). Importantly, at the lower end of this, our results show large and rapid increases in the probability of platypus occurrence with relatively small increases in unobstructed river length. For example, the probability of platypus presence increases from 20.2% at 20 km to 27.6% at

50km and 34.2% at 100 km of unobstructed river length. These results highlight the importance of river connectivity and habitat size as a key determinant of occupancy, likely reflecting the habitat requirements of individual platypus, resilience of populations, and broader metapopulation processes. For example, 45 km is the maximum recorded juvenile dispersal distance (Serena & Williams, 2012), a distance that may represent an important threshold for the maintenance of viable populations. More generally, our empirical results emphasize that at small scales even small increases in river connectivity likely have large consequences for the persistence of this species in waterways.

Our demonstration of the likely importance of river connectivity on occurrence is consistent with work showing the impact of dams – and large dams in particular – on platypus biology and population-genetic structure. While platypuses are capable of overland movement, their capacity to bypass barriers such as dams is limited by predation risk, energy expenditure, and the physical scale of obstruction (Bino et al., 2020; Furlan et al., 2013; Serena & Williams, 2010). Recent genetic evidence confirms that tall dams severely restrict gene flow. For example, Mijangos et al. (2022) found that FST values between platypus groups separated by major dams (>70 m) were up to 20 times higher than in comparable unregulated rivers, and genetic divergence increased proportionally with time since dam construction. In contrast, Ahrens et al. (2025) showed that smaller dams (30–40 m) did not necessarily impede gene flow, suggesting a height-dependent effect on connectivity. Together, these findings indicate that tall dams fragment platypus populations and contribute to long-term genetic isolation, particularly where there are no natural dispersal alternatives.

Importantly, fragmentation limits the capacity of individuals to reach refuge habitats during extreme climatic events. During droughts, platypuses may persist in deep pools, but impoundments above dams do not provide a suitable refuge. These reservoirs are typically deep, exposed, and ecologically distinct from flowing riverine habitats, with low prey abundance and poor burrowing substrate (Mijangos et al., 2022). These impacts are likely to be exacerbated in warmer, drier regions of mainland Australia, where high temperatures and low summer flows increase physiological stress and reduce overland dispersal opportunities (Bino et al., 2019). Indeed, our result showing the general importance of landscape-level water availability to platypus occurrence may emphasize this point. Isolated populations in such areas may be unable to escape drying reaches or recolonise formerly suitable habitat once conditions improve (Khurana et al., 2024). This compounds the demographic risks associated with isolation, including reduced abundance, increased local extinction probability, and limited adaptive potential.

While our study focused on occurrence, the effects of fragmentation likely extend to population density and long-term viability. Platypus densities are known to vary considerably, both temporally and spatially. For example, estimates for the Thredbo River in southern NSW have ranged from 2.5 to 10.8 individuals per kilometre over the past four decades (Goldney, 1995; Grant, 1992; Hawke et al., 2021), while densities at other southerly locations have ranged from 1.3 to 19.3 individuals per kilometre (Hawke et al., 2020). Although our single-visit eDNA survey did not permit density estimation, the strong relationship between connectivity and presence suggests that more fragmented regions likely support smaller, more vulnerable populations. We recommend future targeted population studies in the more than 50 mainland regions we identified as having less than 45 km of unobstructed river length (Figure 3, Appendix S6), particularly those isolated by tall dam walls. Such surveys would clarify the demographic and genetic consequences of fragmentation and inform prioritisation of conservation actions, including bypass infrastructure and assisted translocation where appropriate.

Fire

Our results show that the extent of the 2019–2020 bushfires influenced platypus occurrence, but in a scale-dependent manner. At the local scale, we observed a negative association between fire extent and platypus detection, consistent with expectations that fire degrades riparian and in-stream habitat (Gomez Isaza et al., 2022; Verkaik et al., 2014). In contrast, at the broader watershed (catchment) scale, fire extent was positively associated with occurrence, though with considerable variance. This counterintuitive pattern likely reflects a correlation between fire and conservation land use; the largest burned areas occurred in national parks and reserves, where platypus populations are more likely to persist due to reduced development pressure and higher habitat quality. Thus, the positive effect at larger scales may reflect the broader condition of protected areas rather than a beneficial effect of fire itself.

The negative association with fire at smaller scales aligns more directly with ecological expectations. Fire can reduce riparian vegetation, increase erosion and sedimentation, and elevate stream temperatures (Gomez Isaza et al., 2022; Verkaik et al., 2014) – all of which may impair key platypus habitat features such as burrow banks, refuge pools, and benthic foraging zones. Our findings reinforce earlier work from the southern part of the species' range documenting post-fire declines in the Manning River (Bino et al., 2021), south-eastern New South Wales, and north-eastern Victoria (Griffiths et al., 2020; McColl-Gausden et al., 2023). However, some studies suggest platypuses may be relatively resilient to fire in certain contexts (Serena et al., 2022), likely reflecting variation in fire intensity, habitat condition, and post-fire recovery dynamics.

Crucially, the 2019–2020 fires followed an extended period of drought, which likely reduced population resilience. Lack of water due to drought limits the availability of refuge pools and flow permanence, increasing susceptibility to subsequent disturbances (Bino et al., 2021). In this context, the compounding nature of drought and fire may pose a more substantial risk than either stressor alone. Access to deep, permanent water bodies is likely critical for persistence in increasingly fire-prone landscapes. Taken together, our results suggest that while platypuses may tolerate fire under certain conditions, their vulnerability is likely to increase in fragmented or drying catchments. Given climate projections for more frequent and intense fires, developing a mechanistic understanding of fire's ecological effects on platypus populations, including how these effects vary with hydrology, connectivity, fire severity, and protected-area status, should be a priority.

Macroinvertebrates

We did not detect an effect of macroinvertebrate diversity on platypus occurrence. While platypuses rely heavily on benthic macroinvertebrates for food, this result is consistent with growing evidence that they respond more to prey biomass and accessibility than to diversity *per se* (Lush et al., 2025). Platypuses are generalist feeders, consuming a broad range of invertebrate taxa including Coleoptera, Diptera, Ephemeroptera, Megaloptera, Odonata, Plecoptera, Trichoptera, Hemiptera, Decapoda, Ostracoda, and some annelids and nematomorphs (Faragher et al., 1979; Hawke et al., 2022; Klamt et al., 2016; McLachlan-Troup et al., 2010). They have also been shown to shift their diet following natural disturbances (Hawke et al., 2025) and can remain long-lived and healthy in captivity on diets that are simpler than in the wild (Krueger et al., 1992; Serena et al., 2024). Recent behavioural evidence further supports this interpretation. Lush et al. (2025) showed that platypuses concentrate their foraging activity in areas with high prey biomass, particularly in shallow edge habitats, regardless of overall invertebrate diversity. These findings suggest that diversity alone is not a reliable indicator of food quality or availability, especially at broad spatial scales.

Given platypuses' dietary flexibility and spatial decoupling from prey richness, macroinvertebrate diversity may be a poor proxy for food-related constraints. It is plausible that macroinvertebrates are sufficiently widespread and persistent that platypuses are rarely food limited (Lush et al., 2025; Serena et al., 2001), and that their occurrence is instead governed by other variables such as water permanence, refuge availability, or habitat complexity. Alternatively, food-related factors may still be important, but more directly linked to prey biomass or energetic productivity (Marchant & Grant, 2015), which are rarely measured in distributional studies. There remains a notable gap in empirical studies linking food availability to platypus distribution and condition. Future work should prioritise quantitative assessments of macroinvertebrate biomass, energy content, and spatial accessibility to better understand how food resources shape foraging success, condition, and population persistence.

Other potential drivers of platypus occurrence

Land use and urbanisation had negligible effects on platypus occurrence in our study. The only land use variable retained in the top models was the proportion of land at the watershed scale designated for water-related use, which showed a positive effect (Figure 2). This likely reflects the presence of managed water bodies such as storages or wetlands that may support consistent aquatic habitat, rather than an effect attributable directly to land use classification. We did not detect an effect of NDVI, but this result is best understood in the context of limited variation across our sampling sites. NDVI values ranged from 0.35 to 0.85, with only 10 of 174 sites falling below 0.4, the approximate threshold below which open or urbanised land dominates (Zaitunah et al., 2021). The absence of lower NDVI values indicates that the study did not capture a meaningful gradient of urban intensification. In contrast, previous studies that have sampled across both urban and non-urban areas have documented negative impacts of urbanisation on platypus occurrence, including reduced habitat quality and increased mortality from stormwater runoff and infrastructure (Brunt & Smith, 2025). Although our findings suggest that vegetation cover and land use were not key predictors within this relatively non-urban dataset, they do not imply that urbanisation is unimportant more broadly. Targeted studies along urban-to-rural gradients, particularly in rapidly expanding peri-urban regions, would help clarify how landscape change influences platypus persistence.

Limitations

While this study provides one of the most extensive assessments of platypus occurrence across eastern Australia, several limitations must be considered when interpreting our findings. First, although environmental DNA is a powerful tool for detecting aquatic species, its sensitivity can be influenced by numerous environmental factors. Detection probabilities may decline in high-flow systems due to dilution, or in heavily vegetated or turbid waters due to DNA degradation or adsorption to sediments (Beng & Corlett, 2020; Darling, 2019; Dejean et al., 2011). The risk of false negatives, particularly at low population densities or in highly disturbed environments, cannot be ruled out (Andres et al., 2023; Pinfield et al., 2019; Ruppert et al., 2019). Our single time point sampling design per site limited the ability to explicitly model detection probability, which should be a priority for future work using repeated or replicated eDNA sampling. Second, while our approach offers a pragmatic way to assess concordance between eDNA detections and sightings records, interpreting eDNA non-detections in areas with older sightings is inherently uncertain. Inferences about local extinctions or recolonisation should thus be made cautiously, particularly in the absence of repeated temporal sampling. Third, although spatial coverage was extensive, it was not evenly distributed across the study region. Urban and peri-urban areas were underrepresented. This limited the scope for robust inference about land use impacts, particularly urbanisation, and may have introduced regional biases in model outputs. Fourth, several covariates were derived from broad-scale datasets (e.g. NDVI, land use) that may not capture fine-scale habitat features most relevant to the platypus. For example, riparian condition, bank structure, sedimentation, and local hydrology, all known to influence platypus habitat quality, were not directly measured. Likewise, macroinvertebrate data were based on taxonomic richness rather than biomass or prey accessibility, which may explain the weak predictive power of this variable. Finally, our focus on presence-absence data limits inference about platypus abundance, reproductive condition, or population viability. While occupancy is a useful metric for broad-scale assessments, demographic measures are required to evaluate population viability, particularly in fragmented or fire-affected systems. Despite these limitations, this study advances understanding of likely drivers of platypus persistence across a broad geographic area. It reinforces the importance of river connectivity and post-fire recovery in shaping distribution, while identifying knowledge gaps that can inform more targeted future research.

Conservation implications

There is an urgent need for systematic surveys of platypus distribution across the subtropics and tropics. In Queensland, the last statewide assessment was conducted over two decades ago (Nattrass, 2002), and recent research has been largely restricted to the Greater Brisbane area (Brunt et al., 2021; Brunt et al., 2025; Brunt & Smith, 2025). Environmental DNA offers a powerful and cost-efficient tool for detecting presence across large spatial scales and should be used to address this major geographic knowledge gap. At the same time, more detailed population-level studies should continue be prioritized, and expanded across the species' range.

Our findings suggest that habitat fragmentation, especially from dams, is a strong predictor of platypus occurrence. Areas with less than ~45 km of unobstructed waterway –

particularly those isolated by tall dam walls – should be prioritised for further demographic investigation (see Figure 3 and Appendix S6). Key examples include: the Tumut Dam in New South Wales (86 m tall, with upstream and downstream unobstructed segments of ~10 km and ~16 km), the Bellfield Dam in Victoria (55 m tall, ~15 km of connected river upstream), and the Perseverance Dam in Queensland (53 m tall, ~19 km upstream). These systems exemplify the kinds of highly fragmented catchments where long-term persistence is uncertain. Focused population surveys in these areas would help determine whether platypus populations are declining in size, demographic and genetic structure, or spatial extent. In fragmented systems with short unobstructed reaches, the risks of local extinction are likely to be elevated, particularly where degraded in-stream conditions or past disturbances limit recolonisation potential. Populations confined above or below tall dams may also be more vulnerable to stochastic events, such as fires, prolonged dry periods, or sedimentation pulses, and their resilience to disturbance is likely to be lower than that of populations in more connected, intact systems.

In this context, there is a clear need to re-evaluate the conservation status of the platypus. A more systematic assessment of distributional decline, demographic isolation, and extinction risk across both northern and southern regions would provide the empirical basis for uplisting at state and national levels. Such a change in status would enable stronger legal protections, improved access to conservation funding, and a more coordinated framework for habitat protection and threat mitigation. This study identifies priority landscapes and ecological mechanisms that merit further investigation. By focusing future efforts on fragmented catchments, poorly surveyed regions, and systems with known structural barriers, we can better understand the limits of platypus persistence and design more effective conservation responses.

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Data Accessibility Statement

On publication, the data that support the findings of this study will be made openly available in Dryad at DOI: 10.5061/dryad.wpzgmsc1x.

Tables

Table 1: *Model averaging results using the conditional average method for variables influencing platypus occupancy. Estimates are on the scale of the logit link function. The study region is northern NSW and south-east QLD, Australia.*

	Estimate	Std. Error	P
Intercept	-4.34	0.968	<0.001
Fire – 2km scale	-0.833	0.430	0.054
Water land use – watershed scale	0.146	0.0645	0.025
Unobstructed river length	0.444	0.153	0.004
Fire – watershed scale	0.951	0.477	0.048
Spatial autocovariate - 45km	1.57	0.466	<0.001

Table 2: *Comparison of historical platypus sightings data (ALA) to eDNA detections, within the study region in northern NSW and south-east QLD, Australia.*

Last Recorded Sighting	eDNA grids absent	grids present	eDNA sites absent	sites present
Most recent (2018-2024)	3	18	56	65
Semi recent (2011-2017)	0	2	1	4
Older (2004-2010)	0	0	0	0
Old (pre-2003)	0	1	2	1
Never	4	7	30	15
Total	7	28	89	85

Figures

Figure 1: Study site locations across QLD and NSW. Platypus DNA was detected at 85 sites (green circles) and not detected at 89 sites (purple circles).

Figure 2: Model estimates and 95% confidence intervals from the model averaging results for variables influencing platypus occupancy across QLD and NSW, Australia. The variables included A) Unobstructed river length: the inset plot shows predictions on the natural scale at relatively low levels of unobstructed river length, B) Water: area designated as water for land use purposes within the watershed, C) Fire: area burnt at moderate, high and extreme severity within 2km of each site, and D) Fire within the watershed of each site. Unobstructed river length was significant (p=0.004), as was water (p=0.025). Fire at the 2km scale, and at the watershed scale, were marginally significant (0.054 and 0.048 respectively).

Figure 3: Fragmented areas within the mainland platypus distribution. Fragmented areas (red lines) are those around dams (black triangles) with less than 45km of waterway branches accessible to platypus, assuming dams are full barriers to platypus movement. The platypus distribution (grey shading) data is sourced from "A national assessment of the conservation status of the platypus" (Hawke et al., 2020).

Figure 1

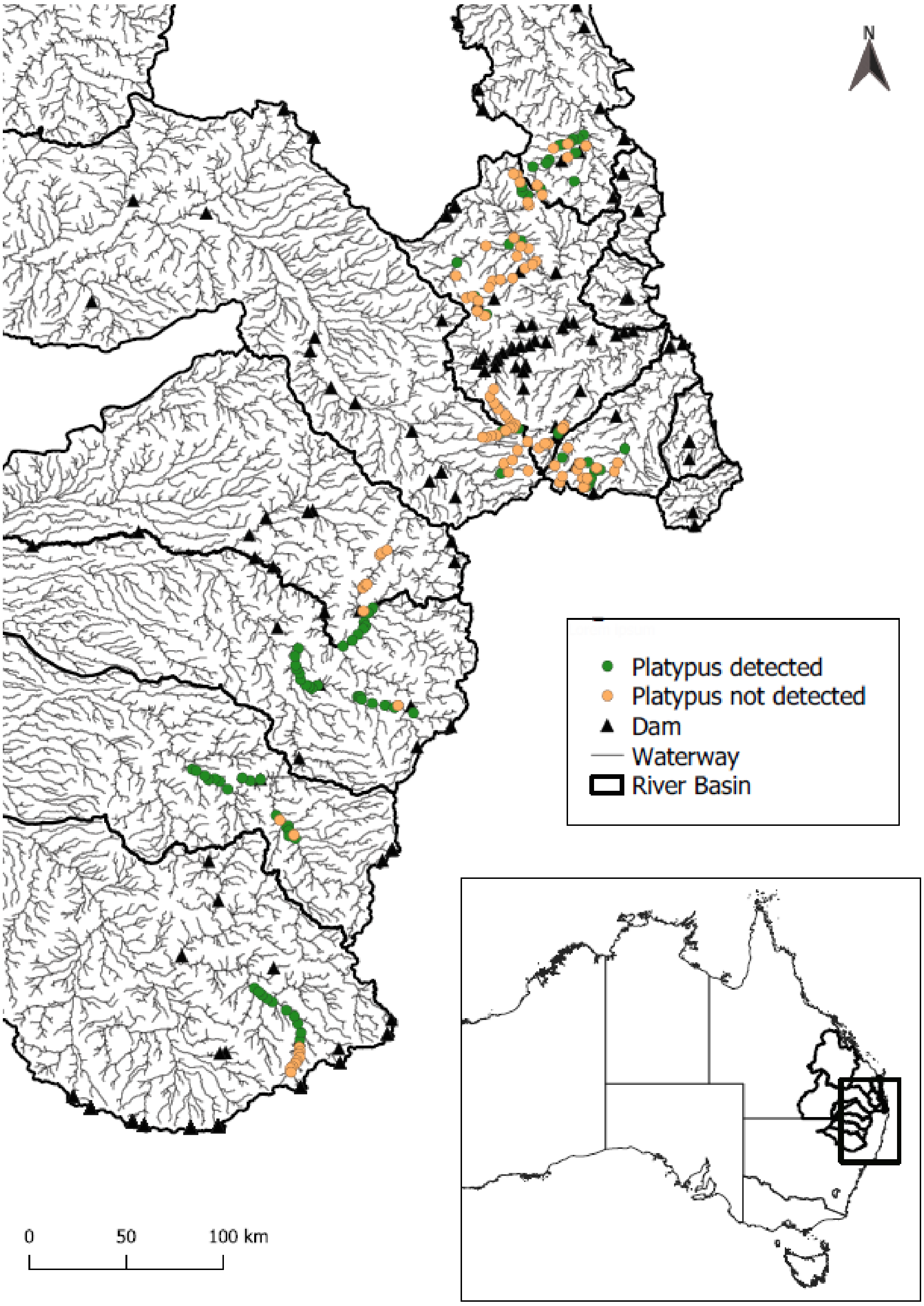

N
Platypus detected
Platypus not detected
Dam
Waterway
River Basin
0
50
100 km

Figure 2

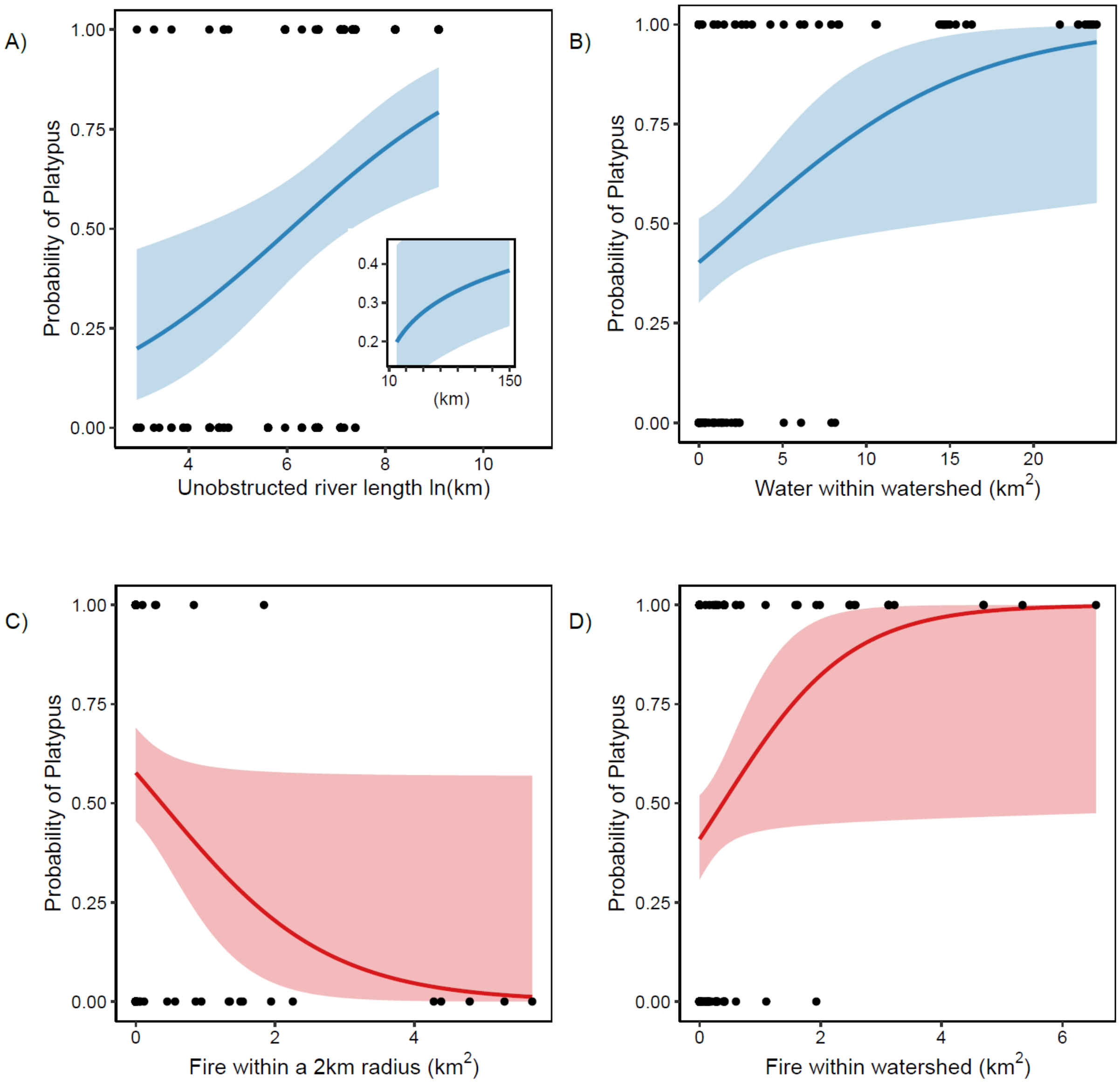
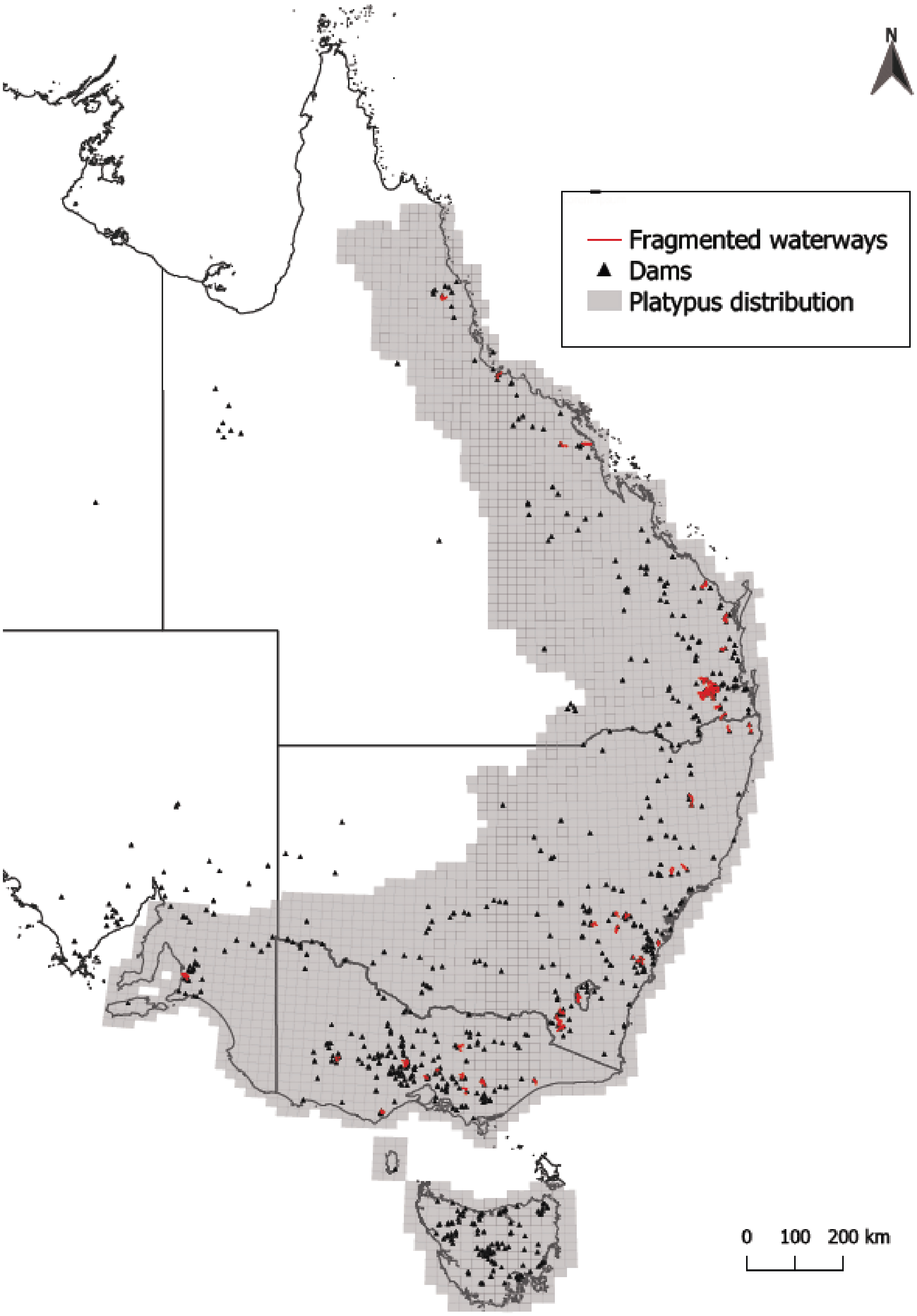

A)
1.00
0.75
0.50
0.25
0.00
Probability of Platypus
4
6
8
10
Unobstructed river length ln(km)
0.4
0.3
0.2
10
150
(km)
B)
1.00
0.75
0.50
0.25
0.00
Probability of Platypus
0
5
10
15
20
Water within watershed (km^2)
C)
1.00
0.75
0.50
0.25
0.00
Probability of Platypus
0
2
4
Fire within a 2km radius (km^2)
D)
1.00
0.75
0.50
0.25
0.00
Probability of Platypus
0
2
4
6
Fire within watershed (km^2)

Figure 3


N
Fragmented waterways
Dams
Platypus distribution
0 100 200 km

Appendices

Appendix S1:

***Table S1**: Waterways surveyed*

River Basin	Waterway	# sites sampled	# sites with eDNA detections	Site codes
Mary River	Kandanga Creek	8	6	KC0 – KC7
	Kingaham Creek	3	3	JM3, YC4, YC5
	Little Yabba Creek	1	1	Z2
	Six Mile Creek	1	0	Z1
	Yabba Creek	5	2	JM1, JM2, YC1 – YC3
Brisbane River (North section)	Anduramba Creek	1	0	CN21
	Bald Hills Creek	1	0	CN1
	Brisbane River	4	1	CN6, CN8, CN9, CN10
	Buckamara Creek	1	0	CN22
	Cressbrook Creek	3	0	CN5, CN7, CN15
	Crows Nest Creek	2	1	CN2, CN3

	Diaper Creek	2	0	DP1, DP2
	Emu Creek	4	2	CN11, CN12, CN16, CN17
	Ivory Creek	2	0	CN13, CN20
	Maronghi Creek	1	0	CN19
	Monsildale Creek	6	3	MD1 – MD4, MN1, MN2
	Oaky Creek	1	1	CN14
	Pierce Creek	1	0	CN18
	Perseverance Creek	2	0	CN4, CN23
Brisbane River (South section)	Coulson Creek	3	0	CC1, CC2, Q43
	Gap Creek	1	0	Q61
	Kulgin Creek	8	0	GR1 – GR8
	Reynolds Creek	3	1	RC1 – RC3
Condamine-Culgoa Rivers	Dalrymple Creek	9	2	MR1 – MR9
	Emu Creek	2	1	Q53, Q55
	Freestone Creek	1	0	Q48
	Rocky Creek	1	0	Q54

	Swan Creek	2	0	A4, Q51
Logan-Albert Rivers	Burnett Creek	4	1	A2, A3, Q13, Q15
	Cronan Creek	1	0	Q5
	Egan Creek	1	0	Q6
	Logan River	7	6	A1, A6, Q3, Q4, Q8, Q17, Q19
	Mount Barney Creek	1	0	Q10
	Rocky Creek	1	0	Q9
	Tamrookum Creek	1	0	Q18
	Teviot Brook	3	1	A5, Q62, Q63
	Waterfall Creek	2	0	Q14, Q59
	White Water Gully	1	0	Q12
Border Rivers (North section)	Pike Creek	9	1	Pike1 – Pike3, Pike1DS, Pike2DS, Pike1US, Pike5US – Pike7US
Border Rivers (South section)	Dumaresq River	8	8	Dumaresq3DS – Dumaresq10DS

	Severn River	19	18	Severn1 – Severn7, Severn1DS – Severn10DS, Severn8US, Severn9US
Gwydir River	Gwydir River	18	16	Gwydir1 – Gwydir8, Gwydir1DS – Gwydir10DS
Namoi River	Peel River	19	10	Peel1 – Peel9, Peel1DS – Peel10DS

Appendix S2:

Table S2: *Variables used to analyse platypus occupancy across south-east QLD and northern NSW.*

Variable	Variable description	Scale	Resolution	Source
Macroinvertebrate family richness		Site		
Macroinvertebrate order richness		Site		
SIGNAL score		Site		
Macroinvertebrate NMDS scores		Site		
Fire	Fire severity from the 2019/20 bushfires (Area burnt at moderate, high and extreme severities)	Watershed and 2km buffer	QLD: 10x10m NSW: 10x10m	QLD - Queensland Spatial Catalogue: "Fire Extent and Severity 2019-2020 – South-East Queensland" (Queensland Government, 2020) NSW - "Fire Extent and Severity Mapping" (NSW Department of Climate Change Energy the Environment and Water, 2020)

Agriculture land use	A combination of the following CLU19 categories: Dryland cropping, Dryland horticulture, Grazing modified pastures, Grazing native vegetation, Intensive horticulture and animal production, Irrigated cropping, Irrigated horticulture, Irrigated pastures, Land in transition, Rural residential and farm infrastructure	Watershed and 1km buffer	50x50m	"Catchment Scale Land Use of Australia – Update December 2023" (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)
Natural land use	A combination of the following CLU19 categories: Managed resource protection, Nature conservation, Other minimal use	Watershed and 1km buffer	50x50m	"Catchment Scale Land Use of Australia – Update December 2023" (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)

Forestry land use	A combination of the following CLU19 categories: Plantation forests, Production native forests	Watershed and 1km buffer	50x50m	“Catchment Scale Land Use of Australia – Update December 2023” (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)
Mining land use	A combination of the following CLU19 categories: Mining and waste, Other intensive uses	Watershed and 1km buffer	50x50m	“Catchment Scale Land Use of Australia – Update December 2023” (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)
Urban land use	CLU19 category: Urban residential	Watershed and 1km buffer	50x50m	“Catchment Scale Land Use of Australia – Update December 2023” (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)

Water land use	CLU19 category: Water	Watershed and 1km buffer	50x50m	"Catchment Scale Land Use of Australia – Update December 2023" (Australian Bureau of Agricultural and Resource Economics and Sciences, 2023)
Runoff	Runoff for the year prior to sampling	Watershed	0.05 degrees	The Australian Water Outlook (Australian Bureau of Meterology, 2024)
5-year average runoff	Average runoff for the five years prior to sampling	Watershed	0.05 degrees	The Australian Water Outlook (Australian Bureau of Meterology, 2024)
Normalised Difference Vegetation Index		1km buffer and 2km buffer	1 arcsecond (27m pixel size)	Sentinel 2 (European Space Agency, 2022)
Elevation		Minimum value within 100m	3 arcsecond (90m pixel size)	Geoscience Australia (Gallant et al., 2009)

Unobstructed river length	Total length of waterway available from the sampling point (all branches are summed and dams are considered as breakpoints).	Waterway	N/A	National dam wall dataset (Geoscience Australia, 2016), HydroRivers (Lehner, 2013)
Spatial Autocovariate	Spatial lag vector for sites within a set distance of each other, calculated using the spdep library in R.	Site		

Appendix S3:

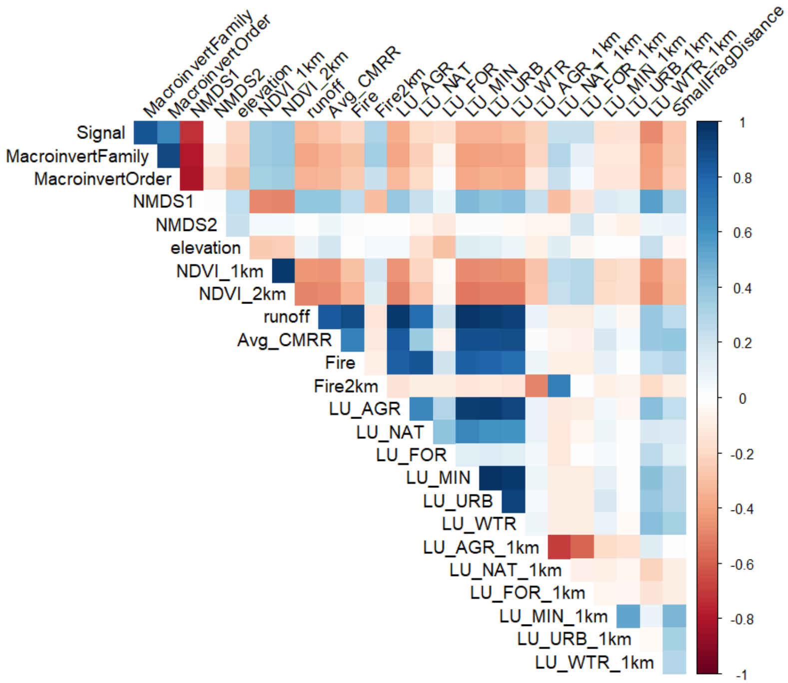


Figure S3.1: *Correlation matrix of all potential variables. From left to right, these were Signal score; macroinvertebrate family and order richness; macroinvertebrate NMDS scores; elevation; NDVI (1km and 2km scale); runoff for the year preceding sampling; average runoff of the five years preceding sampling; Fire (watershed and 2km scale); land use categories of agriculture, natural, forestry, mining, urban and water (watershed and 1km scale); and unobstructed river length.*

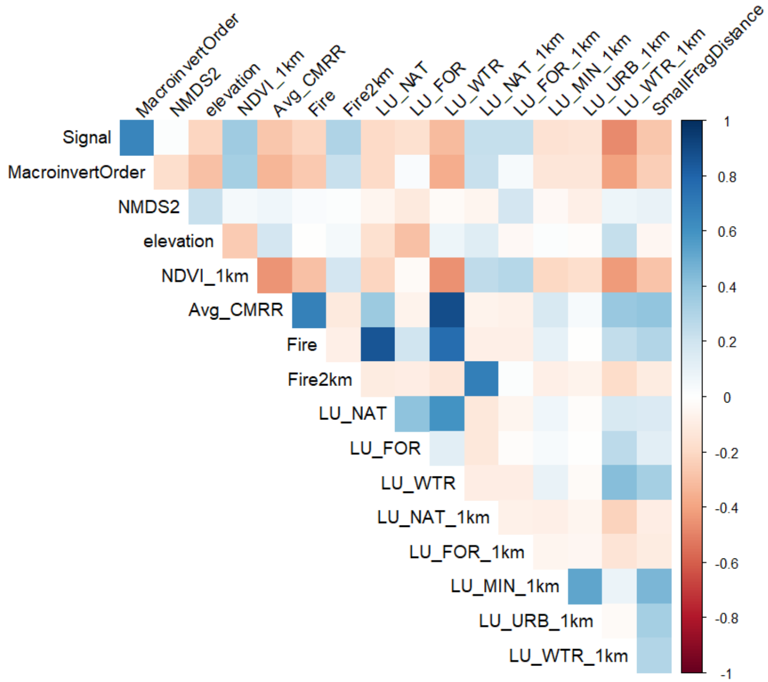


Figure S3.2: *Reduced correlation matrix, after variables with high VIF values were removed. Remaining variables were Signal score; macroinvertebrate order richness; macroinvertebrate NMDS2; elevation; NDVI (1km scale); average runoff of the five years preceding sampling; Fire (watershed and 2km scale); land use categories of natural, forestry, and water (watershed scale); land use categories of natural, forestry, mining, urban and water (1km scale), and unobstructed river length.*

Appendix S4:

Table S4: *Models for platypus occupancy.*

Model	Fire	Fire2km	Water (Land use)	Unobstructed River length	Spatial Autocovariate: 45km	df	AICc
1st		-0.797	0.146	0.440	1.50	5	168.13
2nd	0.951	-0.877		0.449	1.65	5	168.50
Null						1	243.15

Appendix S5

Table S5: *Comparison of sightings records and our eDNA results in fragmented areas*

Waterway/region	# sites	Number of eDNA detections	ATLAS records (up to May 2025)
Peel River - upstream of Chaffey dam	9	0	One 2024 sighting, but in a farm dam, not Peel River itself (public sighting submitted to cplatypus.org). 2008 record in the river (NSW BioNet Atlas)
Pike River - upstream of Glenlyon dam	7	0	One 1975 sighting in the dam itself (Wildnet).
Mary River basin, between Borumba dam and Imbil Weir Sites: YC2, YC3	2	1	2023 sighting, other recent sightings.
Brisbane River catchment - upstream of Cressbrook dam	7	2	One 2021 sighting (iNaturalist), Wildnet records 2002 and prior.

Sites: CN1, CN14, CN2, CN22, CN23, CN3, CN4			
Brisbane River catchment - upstream of Moogerah Sites: CC1, CC2, Q43, Q61	4	0	None
Logan/Albert River catchment (upstream of Wyaralong dam) Sites: A5, Q63, Q62	3	1	None

Appendix S6:

Table S6: *Fragmented areas throughout the mainland platypus distribution. Platypus sightings data is sourced from cplatypus.org (Classifications: Newest: 2019-2025; Recent: 2012-2018; Semi-recent: 2005-2011; Past: 2004 or before).*

Location	Length	State	Platypus Sighting	Dam Height (m)	Dam Name	Other Information
Woodford	0	NSW	Newest	18	Lake Woodford	
MossVale3	5.5	NSW	Past	14	Fitzroy Falls	
MossVale2	7.53	NSW	Newest	19	Wingecarribee	
MossVale4	8.78	NSW	Newest	35	Bundanoon	
Byron	9.14	NSW	Past	28	Rocky Creek	
Tumut	9.515	NSW	Newest	86	Tumut	also, Tumut2, height 46
Byron2	13.31	NSW	Past	42	Clarrie Hall	
Guthega2	14.63	NSW	Newest	34	Guthega	
Tumut2	15.99	NSW	Newest	86	Tumut	also, Happy Jacks, height 29
MossVale	16.38	NSW	Past	23	Medway	
Woronora	18.37	NSW	Semi-recent	66	Woronora	

Winburndale	19.22	NSW	Recent	25	Winburndale	
Byron3	20.22	NSW	Past	44	Toonumbar	
Corin2	21.15	NSW	Newest	76	Corin	also, Bendora, height 47
Guthega	27.11	NSW	Newest	49	Island Bend	also, Guthega, height 34
Blayney3	28.18	NSW	Past	20	NULL	
Tooma	28.65	NSW	Recent	67	Tooma	
Geehi	28.86	NSW	Newest	91	Geehi	
Corin	31.11	NSW	Past	76	Corin	
Oberon	35.44	NSW	Newest	34	Oberon	
Chichester3	41.02	NSW	Past	41	Chichester	
Chichester	42.52	NSW	Newest	67	Glennies cr	
Lithgow	43.28	NSW	Newest	46	Lilyvale	also Wallerawang, height 14; Lyell, height 46; Lithgow no 2, height 27
Armidale	44.27	NSW	Recent	31	Malpas	also, Gara
Townsville	14.87	QLD	Past	5	Alpins Weir	

Gatton/Laidley	15	QLD	Newest		Multiple weirs in this area	
Maryborough	18.63	QLD	Semi-recent	8	Mary barrage	also Tinana barrage and Teddington Wr
Brisbane	18.97533	QLD	Newest	53	Perseverance	CN14, CN22, CN23
Logan-Albert	20.44624	QLD	Past	52	Maroon	Q12, Q13
Mary	26.88825	QLD	Newest	53	Borumba	also, Imbil Weir, height 2; YC2, YC3
BrisbaneSouth	29.93134	QLD	None	40	Moogerah	Q61
Maryborough2	34.56	QLD	Past	7	Teddington Wr	also Tallegalla weir
Eungella	34.93	QLD	Newest	49	Eungella	
Brisbane2	38.37338	QLD	Newest	63	Cressbrook	Also, Perseverance, height 53; CN1-4
Kolan	40.32	QLD	Newest	15	Bucca weir	also, Kolan Barrage, height 5
Collins	41.33	QLD	Newest	14	Collins weir	
Eungella4	41.52	QLD	Newest	7	Marian weir	
Sturt2	8.85	SA	Recent	38	Sturt	
Sturt	34.26	SA	Recent	38	Sturt	

KinglakeNP	7.02	VIC	Past		Toorourrong	
Stawell	14.88	VIC	Newest	55	Bellfield	
Jacana	18.94	VIC	Newest	16	Jacana	
Nicholson	20.73	VIC	Recent	16	Nicholson	
Barwon	21.02	VIC	Newest	43	West Barwon	
Malmsbury	23.55	VIC	Newest	33	Lauriston	also, Malmsbury, height 24
Tarago	24.04	VIC	Recent	34	Tarago	
Strathbogie	24.42	VIC	Newest	13	Polly Mcquin weir	
Malmsbury2	25.86	VIC	Newest	33	Lauriston	also, Upper Coliban, height 28
Yarra	26.63	VIC	Newest	34	O'Shannassy	
Thomson2	40.49	VIC	Newest	166	Thomson	also Swingler, height 18