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RESEARCH ARTICLE

The impact of a decade of urbanisation on a semi-aquatic mammal in a subtropical freshwater ecosystem

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Abstract
Context Urbanisation replaces vegetation with impervious cover, impeding water filtration and increasing runoff contamination. These changes contribute to the 'urban stream syndrome'—a suite of negative impacts on freshwater ecosystems and associated species.
Objectives Assessed how urbanisation, measured using landscape metrics related to variations in impervious cover ('imperviousness') impacted platypus (*Ornithorhynchus anatinus*) occurrence over 11 years in southeast Queensland, Australia.
Methods Leveraging citizen science data (477 annual platypus observations, 67 sites across five catchments, 2013–2023) and satellite imagery, urbanisation was quantified using three remotely-sensed metrics: Normalised Difference Vegetation Index (NDVI), Normalised Difference Built-up Index (NDBI), and urban land cover. Five scenarios based on temporal variation in platypus occurrence and imperviousness were modelled for each metric.
Results All metrics showed imperviousness negatively impacted platypus occurrence, with the strongest effect observed for increased NDBI. Occurrence declined in highly urban, sparsely vegetated areas over the study period, indicating habitat selection preference for healthy waterways and a tolerance threshold to the accumulating effects of urban stream syndrome through time.
Conclusions These findings occurred despite minimal variation to the catchment landscape over the 11-year study, indicating platypus response was due to prolonged urban exposure rather than land-use change. This study supports concerns that platypus declines are being driven by urbanisation. It also presents a widely applicable approach for catchment managers to dynamically assess urban impacts in freshwater ecosystems using remote-sensing metrics and long-term distribution data collected by citizen scientists. To mitigate freshwater degradation and localised extinction risk of platypus, policy recommendations include riparian buffer protection (> 30 m) and water-sensitive urban design.

Keywords Freshwater ecosystem · Imperviousness · Land use · Remote sensing · Conservation · Citizen science

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Introduction

Freshwater ecosystems constitute only 0.01% of global water and cover 0.8% of the Earth's surface (Gleick 1998). However, relative to this limited distribution, their importance for biodiversity is disproportionate: freshwater provides habitat for up to 10% of all recorded species (Dudgeon et al. 2007; Balian et al. 2008; Strayer and Dudgeon 2010; Lynch et al. 2016; Reid et al. 2019). Despite their importance, freshwater ecosystems are among the most threatened habitats in the world (Harrison et al. 2018; Ahmed et al. 2022). Species that rely on freshwater are at high risk of extinction due to high rates of endemism and the natural fragmentation and physical isolation of freshwater ecosystems (Dudgeon et al. 2007). These vulnerabilities are exacerbated by the importance of freshwater as a human resource. Global industrial, agricultural and domestic demand for water is projected to increase by 20-30% within the next three decades—from 4,600 km^3 per year in 2016 to up to 5,500 to 6,000 km^3 per year by 2050 (Boretti and Rosa 2019; He et al. 2021), and maintenance of natural flow regimes is rarely considered as a primary objective in water resource management (Harrison et al. 2016; Vörösmarty et al. 2018). Of the 33,243 freshwater species assessed for the International Union for Conservation of Nature Red List, 10,571 (31.8%) are listed as threatened. A further 6,402 species (19.3%) are listed as Data Deficient, preventing their conservation status from being determined (IUCN 2024). He et al. (2019) recorded an 88% decline in freshwater megafauna between 1970 and 2012. Understanding the causes underlying these declines is crucial for effectively managing risks to freshwater ecosystems and biodiversity.

Over 80% of threatened freshwater species are affected by habitat degradation caused by agriculture, forestry, urbanisation, and infrastructure development (Collen et al. 2014). The degradation of freshwater ecosystems caused by the extraction, diversion, containment, contamination, and/or flow disruption of water in urban landscapes has been described as the 'urban stream syndrome' (Fig. 1) (Paul and Meyer 2001; Walsh et al. 2005). Broadly, the urban stream syndrome can be defined as anthropogenic impacts in urban landscapes which cause freshwater ecological degradation – including an increase in the surface area of infrastructure (hereafter 'imperviousness') proximate to freshwater ecosystems (Paul and Meyer 2001; Walsh et al. 2005). The scope of this study focused on imperviousness, which has a complex suite of impacts including increased urban runoff (Fig. 1a), reduced riparian vegetation (Fig. 1b) and increased impediments to natural hydrological connection and flow (Fig. 1c). These effects are associated with cumulative stressors including reductions in freshwater vegetation quality, habitat loss, degraded water quality, disrupted flow regimes, and declines in freshwater biodiversity (Walsh et al. 2005; Walsh et al. 2016; Craig et al. 2017; Ahmed et al. 2022). The urban stream syndrome is an important conceptual framework for freshwater science as it allows the impact of imperviousness on freshwater species to be assessed in a generalisable way across studies.

Imperviousness (a proxy for urbanisation) has been recognised as a primary driver of freshwater ecological degradation for the last five decades (Slonecker et al. 2001). There have been several reviews of the effects of the urban stream syndrome on freshwater quality, with all finding negative impacts (e.g., Walsh et al. 2005; Dudgeon et al. 2007; Ramírez et al. 2012). However, attempts to untangle the various mechanisms associated with this syndrome have been difficult. This is firstly due to limitations in data availability and methodology. Data covering relevant threats across a wide spatiotemporal scale are needed to adequately assess the impacts of urbanisation (Filbee-Dexter et al. 2017). Second, freshwater ecosystems respond to multiple stressors, which are unique to each system and have additive, synergistic and interactive effects on biodiversity that cannot be examined independently of one another (Folt et al. 1999; Ormerod et al. 2010; Piggott et al. 2015). Fortunately, these stressors do not need to be considered in isolation to inform catchment management and mitigate further declines in freshwater biodiversity (Selkoe et al. 2015). Advances in remote sensing technologies have resulted in open-access databases (e.g., Open Data Cube, Landsat, and Sentinel), which include freely-available, high-resolution satellite imagery. These databases have improved the ease and accessibility of quantifying imperviousness (Zhang et al. 2022; Kuiper et al. 2023), meaning that multiple methodologies have emerged. For example, vegetation indices such as Normalised Difference Vegetation Index (NDVI) and Normalised Difference Built-up Index (NDBI) are spectral imaging estimates of landscape features (net primary productivity and urban density, respectively) derived from satellite data. Reductions in catchment-level NDVI scores have revealed that riparian

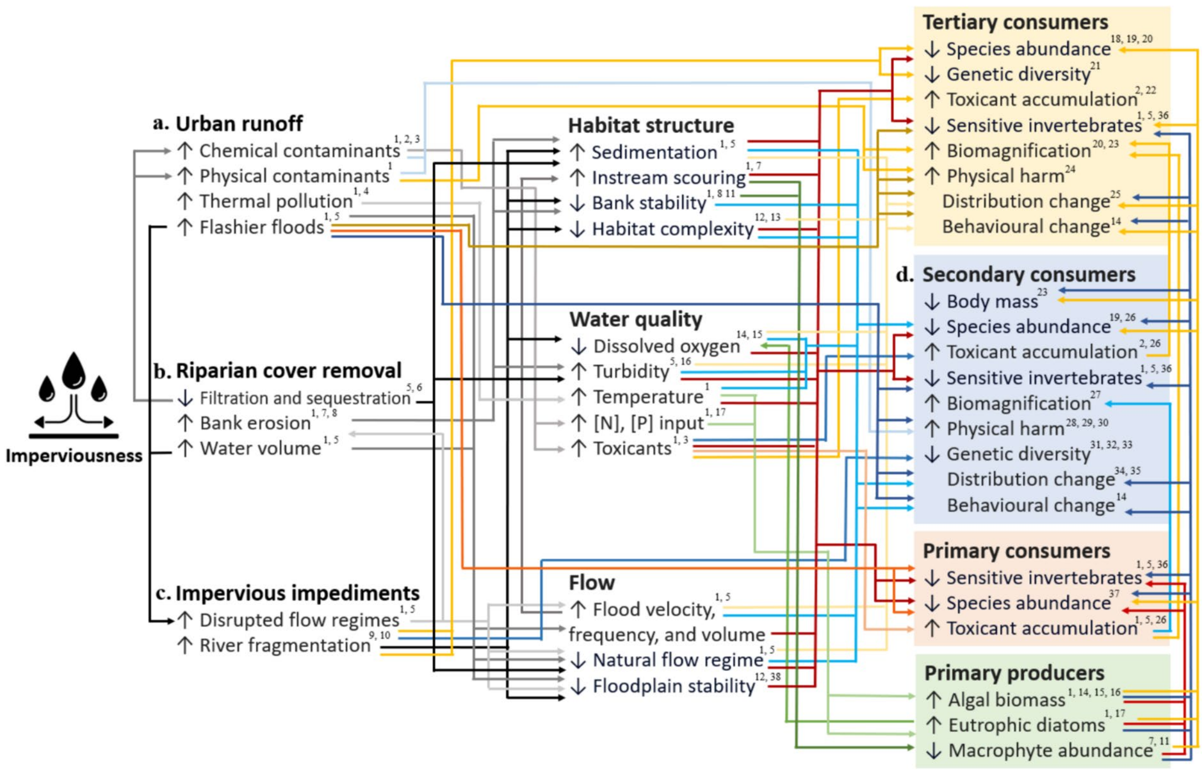

Fig. 1 Summarised effects of the urban stream syndrome—reflected by an increase in imperviousness—on freshwater fluvial ecosystems, derived from the scientific literature. Imperviousness is primarily associated with **a** increased urban runoff, **b** riparian cover removal and **c** impervious impediments. These impacts result in the degradation of freshwater ecosystem habitat features including habitat structure, water quality, and flow, which impact species in the freshwater food web. Species characterised by slow life history strategies, high body mass, complex habitat requirements, and specialised feeding strategies (e.g., platypus as tertiary consumers [d]), are highly susceptible (Brönmark et al. 1992); He et al. 2019; Su et al. 2021). Numbers refer to literature that supports these pathways: 1) Walsh et al. 2005; 2) Zhou et al. 2019; 3) Selaković et al. 2022; 4) Klamt et al. 2011; 5) Paul and Meyer 2001; 6) Li et al. 2020; 7) Lake 2000; 8) Grant and Temple-Smith 2003; 9) Meyer et al. 2005; 10) He et al. 2021; 11) Gell et al. 2009; 12) Kingsford 2000; 13) Serena et al. 2014; 14) Ferrão-Filho and Kozlowsky-Suzuki 2011; 15) Sitati et al. 2021; 16) Wiederkehr et al. 2020; 17) Herrero et al. 2018; 18) He et al. 2018; 19) Zhang et al. 2019; 20) Mor et al. 2022; 21) Holostenco et al. 2021; 22) Bonnineau et al. 2021; 23) He et al. 2019; 24) Iriarte and Marmontel 2013; 25) Rai et al. 2023; 26) Sullivân et al. 2021; 27) Hawke 2020; 28) Connolly et al. 1998; 29) Serena and Williams 2010; 30) Serena and Williams 2022; 31) Coleman et al. 2018; 32) Escoda, Fernández-González and Castresana 2019; 33) Mijangos et al. 2022; 34) Serena and Pettigrove 2005; 35) Martin et al. 2014; 36) White and Walsh 2020; 37) Collen et al. 2014; 38) Gregory et al. 1991

vegetation loss negatively affects river species richness, genetic connectivity and distribution patterns (e.g., Mcfarland et al. 2012; Chen et al. 2021; Benedetti et al. 2023; Brunt and Smith 2025). High NDBI values, indicative of increased urbanisation, have also been shown to negatively affect freshwater quality, riparian density, and species diversity (Andem et al. 2022). Despite the emergence of various approaches, the optimal approach for measuring catchment imperviousness has not yet been determined (Orr et al. 2020), suggesting a need for research examining different methods to develop a more unified framework for freshwater biodiversity conservation.

Freshwater species at high trophic levels, especially tertiary consumers, are disproportionately susceptible to urbanisation and associated ecosystem dysfunction caused by imperviousness (Fig. 1). Imperviousness reduces water quality, microhabitat heterogeneity (such as stable pool and riffle zones) and basal resources (e.g., allochthonous organic matter input) on which consumer species rely (Paul and

Meyer 2001; Walsh et al. 2005). These species often have slow life-history strategies, increasing their overall exposure to cumulative urban stressors and extinction risk (Cooke et al. 2019). Tertiary consumers also face increased biomagnification risk as contaminants in runoff move up through the food web (Kidd et al. 1995; Woodward and Hildrew 2002).

The platypus (*Ornithorhynchus anatinus*), a semi-aquatic monotreme endemic to eastern Australia, fits this high-risk profile (Fig. 1d). Despite rarely moving over land, platypuses are highly mobile – maintaining a typical home range between 0.5–15 km structured by the natural fragmentation and linearity of waterways (Serena and Williams 2013; Bino et al. 2015; Bino et al. 2018). Low flows due to altered hydrological regimes can limit the availability of water they depend on for movement and reproduction (Walsh et al. 2016; Brunt and Smith 2025). At the other extreme, flashier flows scour in-stream channels and erode riparian zones where platypus exclusively forage, rest, and nest (Walsh et al. 2005). Characterised by a specialised benthic foraging strategy, urban runoff introduced into these systems can also increase levels of sedimentation and contaminants, decreasing the diversity and richness of their pollution-sensitive macroinvertebrate prey (Paul and Meyer 2001; Collen et al. 2014). Platypus movement can be further restricted by the implementation of new barriers (e.g., dams and weirs), which reduces habitat quality (Serena et al. 2014; Bino et al. 2019). These structures increase predation risk, energy expenditure and the risk of heat stress as platypus dispersal is forced over land (Robinson 1954; Grant and Dawson 1978; Fish et al. 2001), ultimately limiting gene flow (Kolomyjec 2010; Furlan et al. 2013; Alberti 2015; Brunt and Smith 2025). Studies of platypus populations indicate urbanisation and river regulation has led to genetic differentiation (Furlan et al. 2013; Mijangos et al. 2022; Brunt and Smith 2025), population declines, and local extinctions (Grant 1998; Serena and Pettigrove 2005; Serena et al. 2014; Martin et al. 2014; Coleman et al. 2021). In southeast Queensland, water quality and connectivity has been found to be essential for platypus persistence in urban landscapes (Brunt and Smith 2025; Brunt et al. 2025). Around the city of Melbourne, imperviousness and urban stormwater runoff negatively impacted platypus through habitat degradation (Serena and Pettigrove 2005; Martin et al. 2014). These land use changes, combined with the species' long lifespan, low fecundity, and slow reproductive strategy, limit their ability to recover from population declines caused by cumulative urban stressors (Bino et al. 2019; Coleman et al. 2021; Brunt et al. 2025). However, the extent of these declines and whether these patterns are broadly generalisable across their range is unknown.

Another major limitation in our knowledge about platypus populations is how distributional patterns change over time. The platypus' elusive habit makes declines difficult to detect (Bino et al. 2019). Until recently, compared to their southern counterparts, studies of Queensland platypus have been sparse and irregular (e.g., Stone 1983; Grimley 1995; Brunt et al. 2021; Brunt 2023; Brunt and Smith 2025), particularly in regards to population estimates and species distribution patterns. While localised declines are widely reported (Grant 1998; Serena and Pettigrove 2005; Serena et al. 2014; Martin et al. 2014; Coleman et al. 2021; Brunt et al. 2025), the last attempt to list the species' conservation status as nationally threatened (Hawke, Bino and Kingsford 2020) was rejected due to understudied regions lacking evidence of baseline platypus population dynamics and the resultant reliance on assumption-based models about the magnitude of threats and their impact on platypus declines (Australian Government 2025). This is especially concerning given urbanisation is increasingly impacting southeast Queensland, with new infrastructure required to keep up with predicted population growth estimates from 3.1 million to up to 4.9 million by 2041 (Queensland Government 2019, 2021; ABS 2021). Thus, further assessment of the status of platypus in this region is necessary to provide population evidence that can contribute to conservation action.

The objective of this study was to evaluate the effect of urbanisation on platypus occurrence over an eleven-year study period using three different remotely-sensed metrics imperviousness. Specifically, we asked: has platypus distribution changed over time and can this be explained by changes in urbanisation? We combined the largest long-term observational dataset of platypus ever collected in Queensland (2013–2023) with satellite-derived vegetation indices (NDVI and NDBI) and urban land cover data to assess the impact of imperviousness on freshwater ecosystems. We predicted that imperviousness would negatively affect platypus occurrence for all three metrics of imperviousness. A strength of our study is that we examined changes in imperviousness dynamically: measures were derived for each year of the study separately, rather than static measures which are often used in other studies because of data limitations. Thus, each year of platypus observations had a corresponding level of imperviousness. Understanding the impact of imperviousness on platypus occurrence will help manage the threats posed by the urban stream syndrome, ultimately improving catchment management for conservation of freshwater biodiversity.

Methods

Study region

The study was conducted in southeast Queensland, a region with a subtropical climate influenced by warm northern tropical systems and fluctuations in the southern high-pressure ridge (Queensland Government 2023a). Seasonal average temperatures are 14 °C in winter, 20 °C in spring/autumn, and 24 °C in summer, with the frequency of hotter days predicted to increase in the future (Queensland Government 2020). Mean annual rainfall is 1030 mm, though there is a high variability in average precipitation. Rainfall variation is driven by local factors such as topography and vegetation, and broader-scale weather patterns such as the El Niño–Southern Oscillation, with precipitation predicted to decrease in the future (Queensland Government 2020). The study region encompassed five catchments: Albert River, Coomera River, Nerang River, Mudgeeraba Creek, and Currumbin Creek (Fig. 2). The region is characterised by a range of broad vegetation groups, including wet sclerophyll forests, dry open woodlands, subtropical, complex evergreen notophyll vine forests, palustrine wetlands, and freshwater swamps on coastal floodplains (Neldner et al. 2023). Waterways flowing through

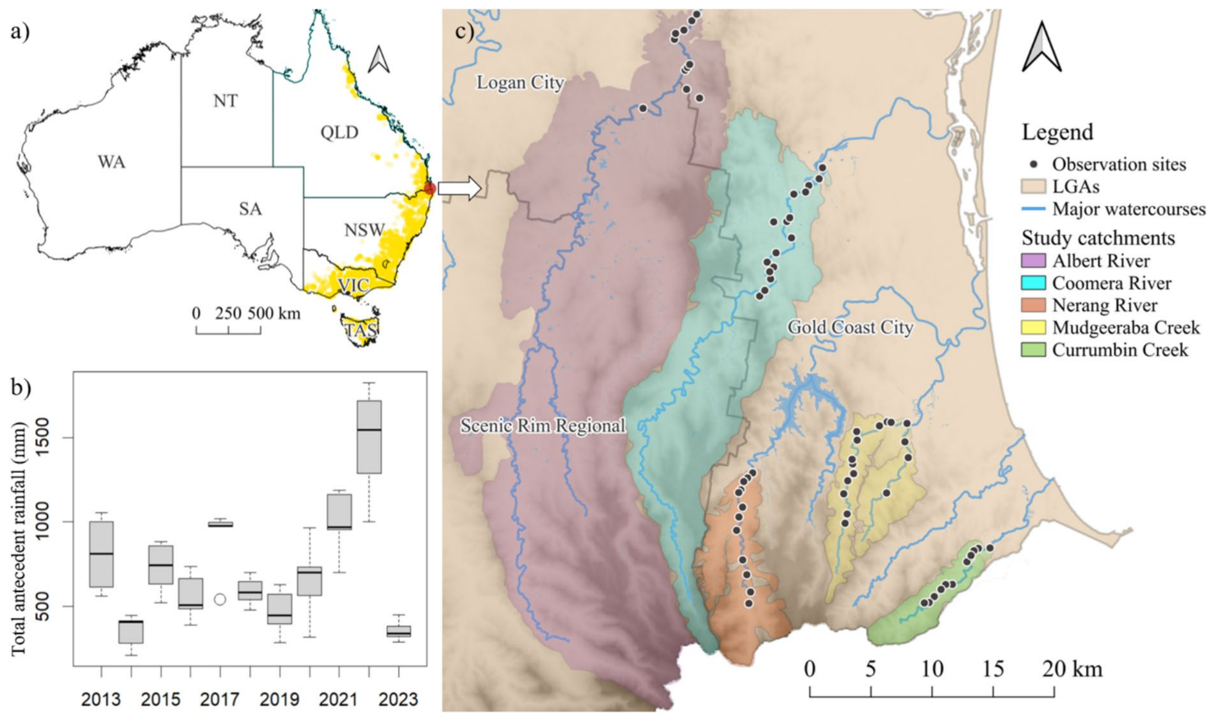

Fig. 2 This study examined platypus (*Ornithorhynchus anatinus*) distribution across southeast Queensland, Australia. **a** The study region is shown in red over the Australian distribution of platypus in yellow (ALA 2024); **b** Variation in antecedent rainfall (mm) (total rainfall for Feb-Jun, the five months prior to platypus breeding season) over the study period (2013–2023) (BOM 2024); **c** Map of the study area across five catchments and location of sixty-seven platypus observational sites (black circles) across three local government areas (LGAs). Background shading on the map represents elevation, with darker shading indicating higher elevation

these catchments have perennial flows, though some smaller tributaries may have seasonal or ephemeral flow. The historical removal of approximately 75% of native vegetation in southeast Queensland for agriculture and infrastructure (industrial, commercial, and residential) has impacted catchment hydrology and sedimentation, causing declines in water quality and losses in aquatic biodiversity (Bunn et al. 2007).

Platypus observational dataset

Platypus occurrence data were obtained from an observational survey database: Watergum's PlatypusWatch program. Watergum is a not-for-profit organisation involved in environmental citizen science, invasive species control and restoration, revegetation, and research projects in southeast Queensland (Watergum Community Inc. 2025). As a quality control and to foster long-term community engagement, volunteers involved in PlatypusWatch were required to attend a two-hour induction prior to conducting an observational survey to learn protocols for safety and platypus identification. The data produced by PlatypusWatch was identified as having potential to advance scientific knowledge and guide waterway management due to the standardised collection effort, sufficient timescale and geographic extent, and research indicating that engaged, trained volunteers can produce data that is of equivalent or higher quality than hired field assistants, particularly when the task procedure is straightforward (presence-absence determinations) and no equipment is requirement (Oberhauser and Prysby 2008; Kosmala et al. 2016). Up to five catchments were surveyed twice per year between 2013 and 2023, except in 2017 when only one survey was conducted due to severe tropical cyclone Debbie (Table S1). Surveys were timed to coincide with known periods of high platypus activity in this region: the mating season (July–October) and puggle emergence from burrows (March–April). Each survey event began fifteen minutes before first light. Observers were paired to reduce imperfect detection (Kosmala et al. 2016) and remained at their allocated site for 2.5 h. If platypus were present, observers immediately messaged the time of sighting to a survey coordinator to mitigate duplicate positive sightings. At the end of each survey event, all observers reconvened to tally sightings and report any further information to the coordinator.

A total of sixty-seven sites across five catchments was used in the current study (Fig. 2) but the sum of sites across all catchments included in the data set were not uniform across years, ranging from 31 in 2015 to 123 in 2021 (overall mean number of sites per catchment = 15 (Table S1). Coomera River, Mudgeeraba Creek, and Currumbin Creek were surveyed throughout the study period, while Nerang River and Albert River were added to the program in 2016 and 2018, respectively (Table S1). These differences are a result of the PlatypusWatch growing in size and funding since inception, allowing more observation sites and catchments to be included in recent years. As a council-funded citizen science project, sites were predominantly on public land and publicly accessible, however, fifteen sites on private land were included in collaboration with private landholders. We assumed, based on platypus movement and home range data (Serena and Williams 2013; Bino et al. 2015; Bino et al. 2018), that sites located within 200 m from another site were not independent. Considering this, and to mitigate the risk of spatial auto-correlation, twenty-five sites located within 200 m from another site were merged to form eleven independent sites. Seventeen sites that were only observed for a single year within the study period (2013–2023) were excluded. This resulted in a total of 824 biannual observations across the sixty-seven sites and eleven-year study period (Table S1).

Definition of catchment-scale study areas

We considered the catchment to be a more appropriate scale for spatial analysis than the local site-level (e.g., < 1 km around the observation site). This was because platypus are genetically structured by regional topography and river systems, forming discrete population units in catchments (Akiyama 1998; Kolomyjec et al. 2009; Gongora et al. 2012; Brunt and Smith 2025). Platypus are also sensitive to catchment-scale modification (Serena and Pettigrove 2005; Magierowski et al. 2012; Bino et al. 2019). For example, the nearest large stream and catchment area were found to be useful indicators of platypus presence in high-order streams (Lunn 2015). Catchment-scale urbanisation has also been found to reduce the abundance of sensitive macroinvertebrate taxa (the primary food source for platypus) due to habitat quality degradation (Walsh and Kunapo 2009; White and

Walsh 2020). Thus, to define catchment-scale site boundaries, a digital elevation model from QSpatial (Queensland Spatial Catalogue 2013) was processed with *r.watershed,* with the minimum size of drainage basins set to 10km^2 (GRASS Development Team 2024) in QGIS 3.38.2. Catchment boundaries then were defined from this model using *r.fill.dir* to generate depressionless elevation and flow direction layers. Mean catchment size was 86.8km^2 (Albert River = 248.2km^2; Coomera River = 85.8km^2; Nerang River = 28.9km^2; Mudgeeraba Creek = 24.4km^2; Currumbin Creek = 13.5km^2, Fig. 2c). Study area polygons were manually modified to include a 200 m buffer behind each study site, and to account for runoff and stream order, included all upstream rivers, tributaries, and drainage lines (Queensland Spatial Catalogue 2023) within catchment boundaries (drainage basins defined using the *r.watershed* tool), resulting in upstream site areas being larger than downstream sites. For example, the most upstream site in Albert River measured 248.2km^2, while the most downstream site was 216.5km^2. To account for this variation, mean imperviousness was calculated for each site based on the respective area. Hereafter, 'site' refers to the catchment-scale area related to each observation point.

Imperviousness metrics

Three different methods of measuring imperviousness were derived to assess changes in imperviousness over time; vegetation indices NDVI and NDBI, and urban land cover. These metrics were selected over others on merits of relevancy to spatiotemporal analysis in urban and freshwater systems, data accessibility (freely downloadable and open-source geospatial files), and widespread applicability and transferability to GIS programs likely to be utilised by catchment managers. Vegetation indices are calculated by transforming multiple spectral bands which have high sensitivity to leaf structure and morphology (Benedetti et al. 2023), allowing for contextual assessment of vegetation cover with imperviousness across a wide spatial scale (Slonecker et al. 2001). NDVI is one of the most widely used vegetation indices. It quantifies net primary productivity, with high NDVI (> 0.36) indicative of a higher vegetation cover and productivity, making it inversely related to imperviousness (Rouse et al. 1974). NDBI quantifies the extent of urbanisation, with greater NDBI values indicating greater imperviousness (Anderson 1976). NDVI and NDBI were derived from Landsat 8–9 Operational Land Imager (USGS 2024), which provide one image approximately every sixteen days at 30 m resolution. The earliest date with available imagery within the study area was April 27, 2013. Satellite imagery was downloaded from EarthExplorer on June 15, 2024. For each year of the study, we selected images taken between November-February of each year, a period which is typically greener after rain. For each study site, NDVI was calculated as:

$$NDVI = \frac{(NIR - Red)}{(NIR + Red)} \tag{1}$$

where NIR is the near-infrared portion of the electromagnetic spectrum (0.75–1.5 μm) and Red is the red portion of the electromagnetic spectrum (0.6–0.7 μm). NDBI was calculated as:

$$NDBI = \frac{(SWIR - NIR)}{(SWIR + NIR)} \tag{2}$$

where SWIR is the short-wave infrared portion of the electromagnetic spectrum (1.57–1.75 μm).

Water is characterised by NDBI values of 1 and NDVI values of -1 due to high infrared reflectance. This does not accurately reflect platypus habitat because water represents the highest quality habitat for that species. Thus, an additional polygon layer for watercourses was added and values were reclassified with the highest pervious value (Brunt and Smith 2025). Mean NDBI and NDVI values were then extracted for each site.

Land use databases are open-source mapping products, often provided by government authorities, which describe how parcels of land are classified within a region, allowing for the isolation of urban areas in mapping software (Anderson 1976). High urban land cover within catchments have been associated with sensitive macroinvertebrate loss (Snyder and Young 2020), reduced fish fitness (Uphoff et al. 2011), and declines in waterbird biodiversity (Xie et al. 2020; Sulliván et al. 2021). For this study, land use mapping series datasets were downloaded from Queensland Spatial Catalogue on June 23, 2024 (Queensland Government 2023b). Only three years of data were available within the study period (2013, 2019 and 2023) so land use analysis was restricted to that timeframe. Urban areas (classified as residential land-use, mining, and commercial services) were extracted in QGIS for each year, then urban land cover (ULC) was calculated as:

$$ULC = \left(\frac{\sum (UrbanLandCoverWithinStudyArea)}{\sum (StudyArea)} \right) \tag{3}$$

where the study area represents the catchment area defined above. Greater urban land cover values are related to increased imperviousness.

Statistical analysis

We were interested in long-term temporal changes in platypus occurrence, rather than seasonal variation. Thus, from the 824 biannual surveys (2013–2023) (Table S1), annual observations were consolidated, with presence values representing a platypus observation at any time during the two survey events for each year. Sites surveyed only once per year due to logistical constraints were also retained as an annual record. The resulting presence/absence (occurrence) comprised 477 annual observations. Data were analysed using binomial generalised additive mixed models (GAMMs), fitted by maximum likelihood using the *mgcv* and *gamm4* packages in R version 4.4.1 (R Core Team 2024). GAMMs allow non-linear relationships between predictor and response variables by fitting smooth functions to each predictor. This allows for variability of observations within groups specified with random effects. The tensor product spline function *t2* was used to focus on important patterns while ignoring unnecessary noise by separating each penalty into penalised and unpenalised components, allowing for basis functions and penalties for all pairwise combinations of components (Wood, Schiepl and Faraway 2013). Thin plate regression ($bs = tp$) (Wood 2003) and fused lasso splines ($bs = fs$) (Wood 2023) were included to construct flexible models which capture main effects and factor-level smooth deviations for each model term without overfitting data (Wood 2023). To mitigate temporal autocorrelation, for NDVI and NDBI (11 years of data available), year was modelled as a smoothed term (e.g., $k = 4$). For ULC (only 3 years of data available; 2013, 2019 and 2023), year was treated as linear ($k = 3$) to avoid overfitting. Each model included study site nested within catchment as a random effect to account for repeated sampling of sites across years and the spatially clustered nature of sites within catchments. Using the *ape* package (Paradis et al. 2024), Moran's *I* tests revealed no significant spatial autocorrelation in the residuals of the top-ranked models ($p > 0.05$).

Previous studies indicated that the amount of rainfall recorded five months prior to platypus breeding season (antecedent rainfall) positively influenced reproductive success and habitat quality (Serena et al. 2014; Brunt 2023). Thus, we conducted preliminary analysis to determine which of three temporal variables was the best predictor of temporal changes in platypus occurrence: year as a numeric variable (0–11), mean antecedent rainfall or total antecedent rainfall. We extracted mean and total antecedent rainfall for February and June of each year of the study from the closest weather station to each site (BOM 2024) (Fig. 2b). Each of the three imperviousness metrics *j* were then fitted separately in a full interaction model with each temporal variable *i* (temporal variable$_i$ × impervious metric$_j$). We ranked models within each *j* using the second-order Akaike's Information Criterion (AICc) in *AICcmodavg* (Mazerolle 2023) in R. This preliminary step indicated that year was the best temporal predictor for all three imperviousness metrics. Thus, we used year as the temporal predictor variable for the remainder of the analysis and did not consider rainfall further.

To determine how year and imperviousness affected platypus occurrence, a set of five GAMMs were fitted for each imperviousness metric separately. A global model, including an interaction between year and imperviousness, was fitted to assess whether the effect of imperviousness on platypus occurrence changed across years. An additive model was fitted to test the effects of year and imperviousness on platypus occurrence separately. Year-only and imperviousness-only models were then fitted to determine if there were independent effects of these variables. Finally, a null model with no variation was included to examine the relative strength of the parameterised models. Models were ranked by AICc. We considered the best model as the one with the lowest AICc values and highest model weight. Models with Δ AICc < |2| were assumed to be approximately equivalent (Hegyi and Garamszegi 2011). As a measure of model fit, we calculated marginal (variance explained by the fixed effects only) and conditional (variance explained by the full model, including random effects) *R*2 for each model using the *performance* package (Nakagawa and Schielzeth 2013; Lüdecke et al. 2025).

Results

Between 2013 and 2023, across 477 annual observations (derived from 824 pooled observations) (Table S1), 233 platypus observations (presences) were recorded (overall annual sighting probability = 47.8%, presence range across years = 35.0–61.5%). Presences were recorded in all five catchments, with fifty-five of the sixty-seven sites recording at least one positive sighting over the study's duration (35.5% overall presence). All three imperviousness variables were included in the top model as predictors for platypus occurrence (Tables 1, 2 and 3; Table S2). For all imperviousness metrics, all models including imperviousness variables improved the fit of the null model by Δ AICc > 2 and all first and second-ranked models were approximately equivalent Δ AICc < |2| (Tables 1, 2 and 3; Table S2). Each model indicated that platypus occurrence probability decreased with increasing imperviousness.

Table 1 Model selection results for platypus *(Ornithorhynchus anatinus)* occurrence as a function of Normalised Difference Vegetation Index (NDVI), ranked by smallest to largest AICc scores

Model	K	AICc	Δ AICc	AIC Weight	Culm Wt	Log Lik
NDVI only	5	585.12	0.00	0.57	0.57	−287.49
NDVI + year	7	586.69	1.57	0.26	0.84	−286.22
NULL	3	588.99	3.87	0.08	0.92	−291.47
Year only	5	589.47	4.35	0.07	0.98	−289.67
NDVI × year	11	592.33	7.22	0.02	1.00	−284.88

K, number of estimated parameters for the model; AICc, second-order Akaike's Information Criterion; Δ AICc, change in AICc; Culm Wt, cumulative AICc weights; Log Lik., log-likelihood

Table 2 Model selection results for platypus *(Ornithorhynchus anatinus)* occurrence as a function of Normalised Difference Built-up Index (NDBI), ranked by smallest to largest AICc scores

Model	K	AICc	Δ AICc	AIC Weight	Culm Wt	Log Lik
NDBI × year	11	583.52	0.00	0.45	0.45	−280.47
NDBI only	5	584.01	0.49	0.35	0.80	−286.94
NDBI + year	7	585.72	2.20	0.15	0.95	−285.74
NULL	3	588.99	5.47	0.03	0.98	−291.47
Year only	5	589.47	5.95	0.02	1.00	−289.67

K, number of estimated parameters for the model; AICc, second-order Akaike's Information Criterion; Δ AICc, change in AICc; Culm Wt, cumulative AICc weights; Log Lik., log-likelihood

Table 3 Model selection results for platypus *(Ornithorhynchus anatinus)* occurrence as a function of urban land cover (ULC), ranked by smallest to largest AICc

Model	K	AICc	Δ AICc	AIC Weight	Culm Wt	Log Lik
ULC × year	6	190.08	0.00	0.45	0.45	− 88.72
ULC only	4	191.35	1.27	0.24	0.68	− 91.53
NULL	3	192.18	2.09	0.16	0.84	− 93.00
ULC + year	5	193.26	3.18	0.09	0.91	− 91.40
Year only	4	193.90	3.81	0.07	1.00	− 92.80

K, number of estimated parameters for the model; AICc, second-order Akaike's Information Criterion; Δ AICc, change in AICc; Culm Wt, cumulative AICc weights; Log Lik., log-likelihood

For NDVI, the NDVI-only and additive models improved the fit of the null model by Δ AICc > 3 (Table 1). There were differences among catchments in NDVI but, across all sites, there was minimal change in urbanisation over the study period (range = 0.24–0.44) (Fig. S1a). The first-ranked model was the NDVI only model (Δ AICc relative to null model = 3.87, AICc weight = 57%, Table 1), indicating that platypus occurrence increased with mean NDVI (i.e. a positive effect of more vegetated, pervious surfaces on platypus) (Fig. 3a). The second-ranked, additive model was approximately equivalent to the NDVI-only model (Δ AICc < 2) and similarly showed a positive relationship between mean NDVI and platypus occurrence (Δ AICc relative to null model = 2.3, AICc weight = 26%, Table 1). Overlapping confidence intervals suggested that the interaction effect between year and NDVI was not strong (Fig. 3b).

For NDBI, all models improved the fit over the null model by Δ AICc > 5, except for the year only model (Table 2). There were catchment level differences in NDBI, but there was minimal overall change in urbanisation over the study period (range = -0.27 − -0.03) (Fig. S1b). The first-ranked model included an interaction between mean NDBI and year (Δ AICc relative to null model = 5.47, AICc weight = 45%, Table 2). This model indicated that, when NDBI

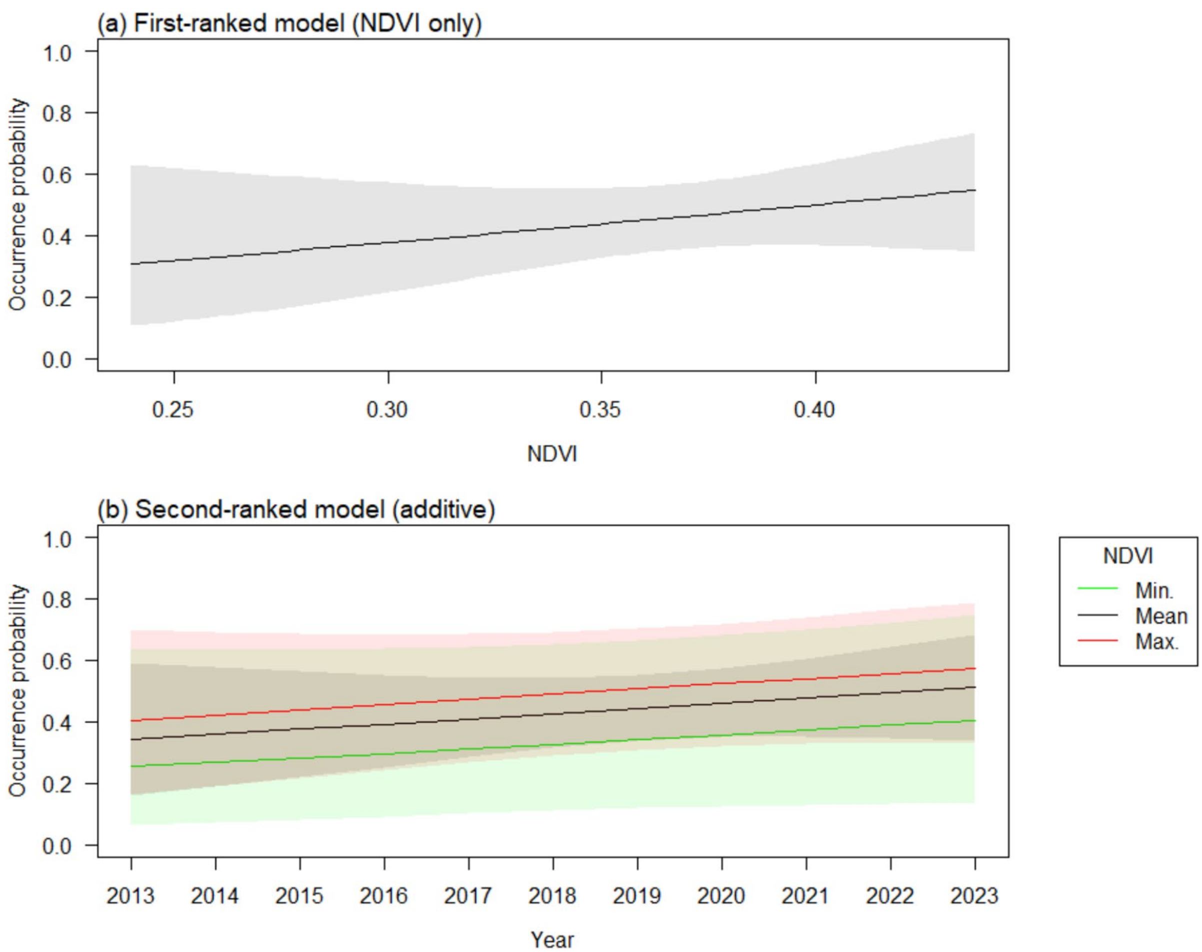

Fig. 3 Effect of Normalised Difference Vegetation Index (NDVI) on platypus *(Ornithorhynchus anatinus)* occurrence in southeast Queensland, Australia (model estimates and 95% confidence intervals). Positive NDVI values are correlated with higher vegetation and inversely related to high urbanisation. **a** The first-ranked model included NDVI only and **b** the second-ranked model included an additive relationship between NDVI and year (NDVI was modelled as a numeric predictor but presented here at minimum, mean, and maximum values for presentation)

values were high (i.e. high levels of imperviousness), platypus occurrence decreased over time. When NDBI values were low (i.e. low levels of imperviousness) platypus occurrence increased over time (Fig. 4a; Table S2). The probability of platypus occurrence was not strongly related to mean NDBI values. The second-ranked model included NDBI only (Δ AICc relative to null model = 4.98, AICc weight = 35%, Table 2), and was approximately equal to the first-ranked model (Δ AICc < 2) (Table 2). This model showed platypus occurrence probability decreased with increasing mean NDBI (i.e. a negative effect of imperviousness on platypus) (Fig. 4). Model fit estimates suggested that the influence of imperviousness metrics was not strong (conditional $R^2 < 0.5$), though NDBI was marginally stronger than NDVI (Table S2).

For urban land cover, the interaction and ULC-only models improved the fit of the null model by Δ AICc > 2 (Table 3). There was substantial variation in site level ULC (0.03 − 0.71) and notable differences among catchments, but there was minimal change in ULC over the study period (Fig. S1c). The first-ranked model was the ULC and year interaction model (Δ AICc relative to null model = 2.09, AICc weight = 45%, Table 3). This model indicated that, when urban land cover was high, platypus occurrence probability increased towards the

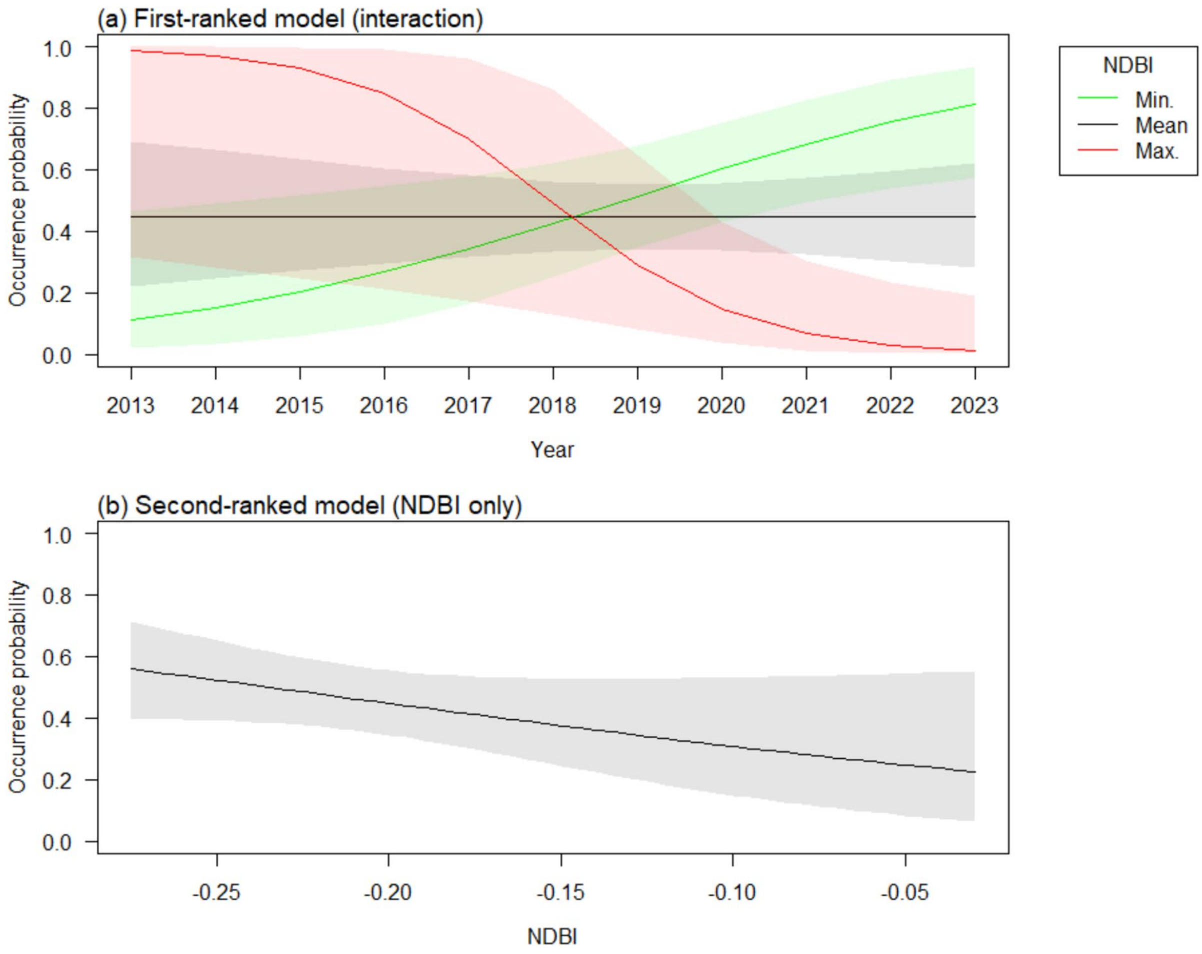

Fig. 4 Effect of Normalised Difference Built-up Index (NDBI) on platypus *(Ornithorhynchus anatinus)* occurrence in south-east Queensland, Australia (model estimates and 95% confidence intervals). High urbanisation is indicated by higher NDBI values. **a** The first-ranked model included an interaction between NDBI and year (NDBI treated as a numeric predictor in analysis but presented within minimum, mean, and maximum categories for the purposes of presentation) and **b** the second-ranked model included NDBI only

end of the study period. When urban land cover was low, platypus occurrence declined over time. However, platypus occurrence was approximately equal for all urban land cover values towards the end of the study period (Fig. 5a). The second-ranked ULC-only model was roughly equivalent to the interaction model (Δ AICc < 2) and to the null model (Δ AICc relative to null model = 0.82, AICc weight = 24%, Table 3) and showed platypus occurrence probability was negatively related to increasing urban land cover (Fig. 5b).

Discussion

Freshwater ecosystems are threatened by environmental consequences associated with the urban stream syndrome. By analysing three different remotely-sensed metrics of imperviousness and the largest long-term observational dataset of platypus in Queensland, Australia, this study showed that high levels of urbanisation were negatively associated with platypus distribution over the eleven-year period between 2013 and 2023 (Tables 1, 2 and 3; Figs. 3, 4 and 5). Platypus distribution was positively associated with sites characterised by high riparian

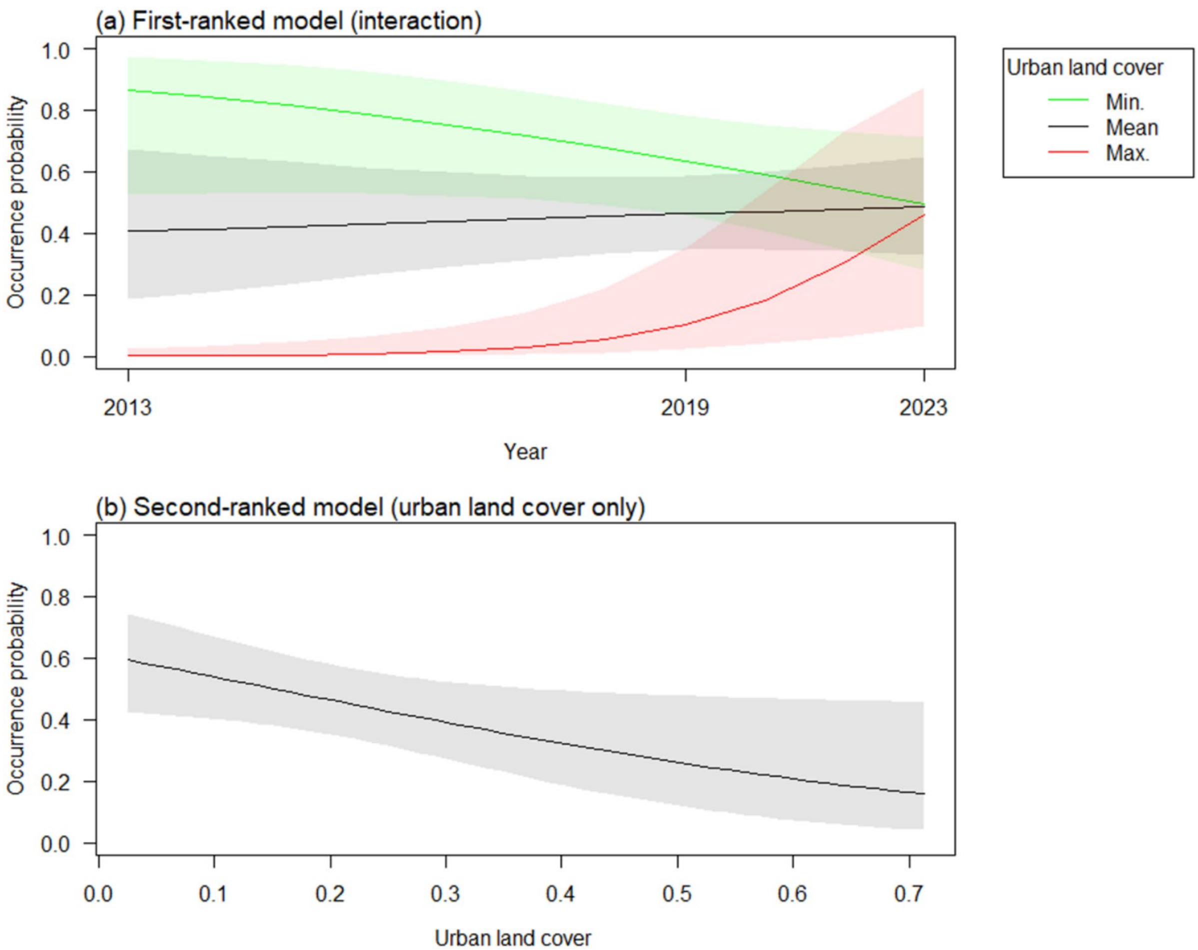

Fig. 5 Effect of urban land cover (ULC) on platypus *(Ornithorhynchus anatinus)* occurrence in southeast Queensland, Australia (model estimates and 95% confidence intervals). High urbanisation is indicated by higher urban land cover values. **a** The first-ranked model included an interaction between ULC and year (ULC treated as a numeric predictor in analysis but presented within minimum, mean, and maximum categories for the purposes of presentation) and **b** the second-ranked model included ULC-only

vegetation cover (increased NDVI) and negatively correlated to more urban sites (increased NDBI and ULC), reflecting the species' dependence on healthy, vegetated freshwater ecosystems. Despite overall catchment urbanisation changing minimally over this period (Fig. S1), high levels of imperviousness had measurable negative effects on platypus presence. Thus, our results indicate that how platypus respond to urbanisation changes over this eleven-year period, even if urbanisation itself does not change substantially. These findings suggest that prolonged urban exposure (i.e., the consequences of the urban stream syndrome through time) could act as an environmental filter capable of driving freshwater upper trophic consumers from urban to non-urban regions. This is likely due to the cumulative degradation of critical habitat features (e.g., riparian complexity, prey availability, hydrological integrity and connectivity) (Fig. 1). These impacts may be compounded by slow life-history strategies, pollution biomagnification, and dispersal barriers which limit recruitment and recovery from sustained stressors (Kidd et al. 1995; Woodward and Hildrew 2002; Olden et al. 2011; Cooke et al. 2019). This study presents a widely applicable approach for catchment managers to dynamically assess the impact of urbanisation on freshwater ecosystems using long-term distribution data of higher trophic consumers and simple remote-sensing metrics. As urbanisation continues to threaten freshwater ecosystems, effective waterway management must prioritise hydrological integrity and connectivity (e.g., water extraction regulations, barrier removal, stormwater treatment), riparian vegetation preservation and restoration (> 30 m riparian buffers, stock exclusion, habitat enhancement strategies), and water-sensitive urban design (e.g., green–blue infrastructure and harsher limits on developments near major rivers). Such management would need to be applied synergistically with private, public and government entities across entire the catchment, considering upstream and downstream linkages and addressing land-use impacts to mitigate cumulative stressors that drive population declines (Briggs 1995; Hughes 2018; Wu et al. 2020; Griffiths, Licul and Imprey 2022).

High imperviousness was demonstrated to negatively impact platypus distribution (Tables 1, 2 and 3; Figs. 3a, 4b, 5b), a species characterised as a carnivorous/insectivorous secondary consumer (Marchant and Grant 2015; Hawke et al. 2022). What could be the mechanisms driving these observations? Following an example pathway of urban stream syndrome (summarised in Fig. 1), the reduced stormwater filtration correlated with riparian cover removal reduces the provision of benthic organic matter for macroinvertebrate prey, which are also impacted by the negative effects these urban stressors have on water quality (Lunney et al. 2004; Walsh et al. 2005) (Fig. 3a). The abundance and diversity of macroinvertebrate prey are further impacted by in-stream scouring and increased contaminants carried by stormwater runoff (Paul and Meyer 2001; Walsh et al. 2005; Collen et al. 2014). The increased turbidity from bank erosion and runoff also impairs platypus foraging efficiency (Gust and Handasyde 1995; Serena and Pettigrove 2005) and the resultant sediment settles and accumulates in pools. These impacts affect platypus habitat complexity, with the species favouring healthy riparian zones characterised by overhanging vegetation, coarse benthic substrate, large woody debris, and undercut banks (Serena et al. 2001; Milione and Harding 2009; Coleman et al. 2021). As top predators, platypus presence is likely representative of far broader consequences within the freshwater food web than demonstrated here. Other apex predators indicate freshwater quality due to their functional role in the trophic system and interactions between urbanisation and their distribution, reproduction, behaviour, and life history traits (Fig. 1). For example, dippers *(Cinclus spp.)*, a passerine endothermic predator with a similar feeding niche to platypus (insectivores), are useful bioindicators of freshwater health (Maznikova et al. 2024). This is because urbanisation typically has a strong effect on their lower trophic level prey abundance and water quality (bottom-up reduction in energy flow) (Maznikova et al. 2024). Similarly, the extirpation of *Barbus meridionalis*—an endangered freshwater apex predator—triggered a trophic cascade resulting in major changes to the ecosystem's overall structure and function (top-down reduction in energy flow) (Rodríguez-Lozano et al. 2015). The negative consequences of platypus' reliance on quality freshwater habitat features and basal resources impacted by the accrued effects of urban stream syndrome is evident in their reduced presence in response to high levels of urbanisation (Tables 1, 2 and 3; Figs. 3, 4 and 5). However, as platypus can be found in degraded habitats where water quality and riparian vegetation are low, they do not reveal the high-impact, short-term shifts in habitat quality that are broadcast by the disappearance of more sensitive freshwater bioindicators such as those mentioned above. Instead, their distribution acts as a valuable general index for the long-term degradation of freshwater ecosystems and that a critical threshold has been crossed, highlighting their importance as a priority species for monitoring and research in urban rivers. Catchment managers can assess urban impact in freshwater ecosystems through the analysis of distribution patterns of similar freshwater predators capable of describing ecosystem response to urbanisation due to their susceptibility to urban stressors (He et al. 2019; Su et al. 2021).

Catchment managers require tools to understand the impacts of urbanisation on freshwater ecosystems, but the suite of available measures of urbanisation have made this task difficult. By comparing three different remotely-sensed measures of imperviousness, this study goes some way towards addressing this problem. As predicted, we found that of three different imperviousness metrics, NDBI showed the greatest increase over the null model in describing platypus distributions (Table 2). This is likely because NDBI was specifically designed to rapidly and accurately map urban areas using spectral bands that characterise imperviousness (Zha et al. 2003). Comparatively, while NDVI effectively assesses long-term changes in the density of riparian canopies, the index is influenced by temporal variations in vegetation greenness and regions characterised by reduced vegetation density such as cleared agricultural pastures (Pace et al. 2022). This made NDVI a slightly weaker representative of imperviousness in our study. The variability in site imperviousness across years for the vegetation indices (Fig. S1a-b) can be explained by the high temporal dynamics of river corridors and spatial heterogeneity which can be difficult to distinguish with spatial imagery at 30 m resolution (Knehtl et al. 2018). NDBI for example has been found to overestimate imperviousness due to its inability to differentiate between urban, sand, and barren landscapes with similar spectral properties (Agarwal and Verma 2022). While not ideal, these indices do have broad spatiotemporal applicability given how these scores are derived (spectral bands from freely available open-source global satellite imagery). Land cover data sets on the other hand are limited by spatial and/or temporal availability, depending on the region (Smith et al. 2010; Wickham et al. 2013). For this study, there were only three years' worth of publicly available spatial products that fit the study's region and duration, thus the derived models could not be reasonably compared to vegetation indices. Regardless, the metric coefficients similarly showed weak overall influence on platypus occurrence, with random effects more greatly contributing to change over time (Table S2; Fig. 2c). The three government-designated urban land cover classifications (mining, residential and commercial land use) limited the capacity to accurately represent fine-scale heterogeneity (areas with < 10% imperviousness) and change in these catchments (Smith et al. 2010). These factors likely contributed to the urban land cover models barely improving the fit over the null model (Table 3). Therefore, for the purposes of this study, given the limited availability of data for land cover, the vegetation indices more effectively represented changes in platypus distribution. Despite these constraints, this study indicated negative impact of catchment urbanisation on platypus occurrence for all imperviousness metrics (Tables 1, 2 and 3; Figs. 3, 4 and 5). Catchment managers should consider available resources (e.g., data sets, funding, time), relevant climate and environmental variables, and the variation in the number and severity of urban stressors when selecting a spatial approach to assess impacts on freshwater ecosystems.

The persistence of platypus at moderately low densities of vegetation (Fig. 3b) and high levels of imperviousness (Fig. 4a) at the beginning of the study period has been found in other studies. Platypus have been known to tolerate disturbed habitat conditions (Grant and Temple-Smith 2003), with populations occurring in waterways bordering cleared agricultural (Lunney et al. 1998) and urban landscapes (Serena et al. 1998; Serena and Pettigrove 2005; Brunt et al. 2018). Platypus are also rarely sighted in sparsely vegetated, heavily urbanised landscapes, such as within fifteen kilometres of Melbourne's inner-city suburbs in Victoria (Serena and Pettigrove 2005; Klamt 2017). Studies from Victoria have shown reaches capable of supporting platypus populations were characterised by a threshold of 11% catchment imperviousness associated with drainage connection (Serena and Pettigrove 2005; Martin et al. 2014), indicating platypus are sensitive to urbanisation and stormwater runoff. The decline in platypus occurrence at higher imperviousness over the study period

(Fig. 4a) correlates with other reports of declines and localised extinctions with prolonged exposure to urbanisation (Grant 1998; Serena et al. 2014; Brunt et al. 2025; Brunt and Smith 2025). This trend is further evidenced in recent southeast Queensland platypus environmental DNA studies (Wildlife Preservation Society of Queensland 2024; Brunt et al. 2021; Brunt 2023), which have revealed 24% of repeatedly sampled waterways in the Greater Brisbane region no longer show platypus presence; declines that have occurred within the span of two decades (1990–2016). This was similarly evident in our results (Figs. 3a, 4b), with platypus distribution positively associated with regions characterised by healthy riparian vegetation (Fig. 3a) and low urbanisation (Fig. 4b) despite there being little change in urbanisation over the 11-year study (Fig. S1). These findings (Tables 1, 2 and 3; Figs. 3b, 4a, 5a) led us to extrapolate that it was platypus response to urbanisation that changed with time, rather than urbanisation itself.

The suggestion that prolonged urban exposure is driving platypus population declines across an urbanisation gradient is consistent with source-sink metapopulation dynamics (Driscoll 2007). Local extinctions can be caused by temporary or permanent changes in habitat quality (in this study, vegetation removal, bank destabilisation, increased anthropogenic disturbance, flashier flows and waterway pollution) which can prevent habitat patches from being immediately recolonised (Hanski 1999). Global avian studies, for example, have provided evidence of this non-random filtering of regional species pools in urban areas (Santini et al. 2019; MacGregor-Fors et al. 2022). However, research into the long-term effects of urbanisation in river systems is relatively less developed. Importantly, platypus occupancy was negatively impacted by high levels of imperviousness over time (Tables 1, 2 and 3; Figs. 3, 4 and 5), despite little overall change in catchment imperviousness throughout the study period (Fig. S1). It is also possible that the trend of high platypus occurrence at high NDBI values at the beginning of the study period to low occurrence at low NDBI levels towards the end of the study (Fig. 4a) correlated with the peak in total antecedent rainfall in 2022 (Fig. 2b). The potential explanation for this phenomenon being that platypus were unable to take advantage of high rainfall conditions in highly urban areas due to the link between imperviousness and flashier flows and/or stormwater runoff (Fig. 1), resulting in platypus occurrence declines in these areas. Given that it is platypus response to urbanisation rather than the change in urbanisation itself, we suggest that prolonged urban exposure could be an environmental filter driving this shift in regional patch occupancy. This concept is corroborated by prior studies that have attributed differences in platypus distribution to stormwater runoff (Martin et al. 2014) and stream flow (Coleman et al. 2021). Further research is required to investigate this pattern of significant rainfall periods followed by decreased platypus presence in urban regions. The accrued impacts of urban stream syndrome through time may have serious implications for a species such as the platypus, which is sensitive to urbanisation and selective in its choice of habitat, potentially resulting in stochastic extinction if densely vegetated refugia is no longer available in catchments. Further investigation (e.g., combining occupancy modelling approaches with detailed demographic research) would be required to interrogate whether prolonged exposure to urbanisation imposes a non-random environmental gradient on freshwater species from urban to vegetated regions.

To reduce the cumulative stress driven by long-term urban impacts to freshwater ecosystems and prevent extinction risks, proactive, adaptive and multi-disciplinary conservation is critical. Such efforts must pull together scientists, government entities and stakeholders at the catchment-scale to be effective (Hughes 2018). Isolated, non-integrative interventions that do not directly address the causes of the urban stream syndrome are unlikely to be effective due to the numerous and interlinked impacts on freshwater systems (Hughes 2018). Catchment managers should prioritise preserving and/or restoring vegetation by actions such as widescale revegetation and the implementation of riparian buffers (> 30 m on both sides of the waterway) to enhance habitat quality by accommodating platypus nursing burrows, increasing bank stability, providing woody substrate and coarse organic matter, regulating water temperature due to increased shade, increasing habitat connectivity with the expansion of green corridors, and improving water quality through the filtering of urban runoff (Bino et al. 2019; Wu et al. 2020; Griffiths et al. 2022). Overall, improving riparian vegetation may increase platypus movement due to increased concealment as they swim, forage, and disperse through waterways, especially in urban environments where threats intensify (Bino et al. 2019). To limit impervious encroachment and stormwater impacts, zoning laws should be enforced to restrict major development near waterways and green–blue infrastructure should be promoted (e.g., permeable pavements, bioswales and raingardens) (Perini and Sabbion 2017). Innovative water-sensitive urban designs should also be considered such as the network of "smart" rainwater tanks to be deployed in an urban catchment east of Melbourne, Australia, designed to regulate flows and provide water during periods of drought (Fletcher et al. 2021; Melbourne Water 2025). Incentives for weed management, revegetation, and stock exclusion via fencing by private landholders have also proven to be a highly successful and sustainable strategy (Manci 1989; Griffiths, Licul and Imprey 2022; Malcher et al. 2023). Further actions designed to improve water quality and hydrological integrity such as upgrading stormwater treatment through biofilters or sediment traps to reduce contaminant loads and removing or modifying barriers like dams and weirs to restore natural flow and allow platypus dispersal will also mitigate threats to freshwater ecosystem health and platypus persistence.

The greatest limitation of the imperviousness metrics used in this study is that they preclude inference as to which stressors associated with urban stream syndrome are causing declines in platypus occurrence. Disentangling multiple stressors from imperviousness (Fig. 1) is difficult due to the various ways they can interact with one another (e.g., additively, synergistically, antagonistically) (Folt et al. 1999; Ormerod et al. 2010; Piggott et al. 2015), as well as how often these stressors are masked by background climate variables (Lin et al. 2020). Additionally, the imperviousness metrics applied in this study do not fully encompass the full suite of anthropogenic impacts described by the urban stream syndrome and are, at best, proxies for a much wider array of processes impacting these catchments. The severity of stressors, as well as the influence of high-impact, single-point stressors are also not considered. For example, because freshwater ecosystem function is highly reliant on river connection and regular flow regimes (Meyer et al. 2007), dams (spatially constituting a very small area of imperviousness) are highly detrimental – disrupting freshwater species dispersal pathways, fragmenting populations, and reducing genetic diversity and adaptive potential in semi-aquatic mammals (Coleman et al. 2018; Escoda, Fernández-González and Castresana 2019; Mijangos et al. 2022). Being a primarily fluvial species, dams have proven to act as barriers to platypus movement and gene flow (Kolomyjec et al. 2009; Gongora et al. 2012; Furlan et al. 2013; Hawke, Bino and Kingsford 2020; Mijangos et al. 2022). Because no dams impacted the study catchments during the 11-year period (Seqwater 2025), conclusions capable of being drawn from this study have limited applicability to the management of systems with high-impact, single stressors like dams. Given these limitations, modelling the effect of habitat variables (e.g., river discharge rates, river connectivity, attenuated runoff) on freshwater biodiversity may more effectively identify stressors contributing the greatest impact. We also emphasise the value of corroborative methods such as combining environmental DNA and observational surveys for independent quality control in future studies (Brunt and Smith 2025). Such techniques and knowledge will be important to inform targeted catchment management actions to conserve freshwater biodiversity. The importance of using high-resolution, region-wide mapping for assessing urban impact over time, identifying priority regions for intervention, and taking appropriate efforts to reduce the consequences of urbanisation should also not be overlooked. The identification of urban impact is a first step which will hopefully galvanise efforts towards a multi-disciplinary framework to investigate the underlying threatening processes driving these declines so targeted management actions can be developed. The continued monitoring and development of longitudinal datasets will assist in identifying further contractions in platypus distribution and/or provide baseline estimates of population sizes which can be used as evidence to improve the species' conservation status in future State and Federal assessments, ensuring the persistence of this iconic animal.

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Author Contributions CJR, TB and ALS conceived and designed the study. CJR performed material preparation, data curation, formal analysis, investigation, methodology, figure creation and project administration under guidance from ALS. The manuscript draft was written by CJR. All authors contributed to subsequent versions of the manuscript, read, and approved the final manuscript, and agree to be accountable for all aspects of the study in ensuring that questions related to the accuracy or integrity of any part of the work are appropriately investigated and resolved.

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Data Availability All data and R code generated and/or analysed during this study have been archived on the Zenodo Public Repository: https://doi.org/10.5281/zenodo.14279089.

Declarations

Conflict of Interest The authors declare no competing interests.

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