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RESEARCH ARTICLE

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Land use surrounding wetlands influences urban populations of a freshwater turtle

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Abstract

1. Urbanization is one of the most influential land use changes globally and continues to affect wetland ecosystems and their biota. Freshwater turtles, which rely on both terrestrial and aquatic habitats to complete their life cycles, are one of the most endangered vertebrate groups, with approximately 60% of species threatened. Although habitat alteration caused by urbanization is recognized as one of the main threats to freshwater turtles, there is a paucity of studies quantifying the effects of terrestrial habitat change on turtle populations.
2. The aim of this study was to determine how terrestrial land use change, associated with urbanization, influences the viability of freshwater turtle populations. Thirty-three wetlands were sampled for the southwestern snake-necked turtle (*Chelodina colliei* Gray, 1856) (Chelidae) between October 2016 and February 2017 within a region of continuing urban intensification. Land use and habitat types were classified at the aquatic–terrestrial interface and within a 300-m band around each wetland. Generalized linear mixed models were used to identify the land use variables that best explained the relative abundance of *C. colliei*.
3. Turtle abundance and population structure varied widely among wetlands. The percentage of residential land use, and the presence and accessibility of fringing native vegetation, was positively associated with the relative abundance of *C. colliei*. The association with residential land use may be an artefact of historical land use, whereas the association with native vegetation is probably because adjacent vegetation provides connectivity with suitable nesting sites, and thus facilitates increased recruitment.
4. This study shows how the modification of terrestrial habitat around wetlands may directly influence the population viability of freshwater turtles. Protection and restoration of native vegetation fringing urban wetlands is crucial to support the viability of remnant freshwater turtle populations.

KEYWORDS

Chelodina colliei, extinction debt, habitat fragmentation, population decline, southwestern snake-necked turtle, urbanization

1 | INTRODUCTION

Urbanization is a major cause of land conversion globally, resulting in the conversion of natural environments into environments dominated by buildings, roads, and other impervious surfaces (McDonnell & Pickett, 1993), and often degrading, fragmenting, isolating, or completely removing terrestrial and aquatic habitats (McKinney, 2002, 2008). Reductions in biodiversity are correlated with increasing levels of urbanization (DeStefano & DeGraaf, 2003; McKinney, 2002). As urbanization is permanent compared with other forms of habitat loss, it is one of the most influential factors in current and predicted species extinctions (McDonald, Kareiva, & Forman, 2008; McKinney, 2002).

Wetlands provide ecosystem functions and services for terrestrial and aquatic species, as well as for humans (Ramsar Convention Secretariat, 2013; Semlitsch & Bodie, 2003). Anthropogenic processes, including agriculture and urbanization, are major causes of wetland modification such as draining, infilling, conversion, and often the complete removal of the wetland (van Asselen, Verburg, Vermaat, & Janse, 2013). It is estimated that approximately 64–71% of natural wetlands around the world have been lost since the year 1900 (Davidson, 2014). Those that have not been removed are often modified or isolated within a terrestrial landscape dominated by urban development (Gibbs, 2000). A large proportion of the terrestrial environment surrounding urban wetlands, as well as their flood-adapted fringing vegetation, is often degraded or replaced by built environments, adversely affecting native species that require this habitat for breeding sites, shelter, provision of food, and migration (Semlitsch & Bodie, 2003).

Freshwater turtles have undergone substantial population declines in recent decades, with approximately 61% of all species under threat of extinction or already extinct; resulting in them being one of the most endangered vertebrate groups globally (Lovich, Ennen, Agha, & Gibbons, 2018). As freshwater turtles can occur in high densities and many species are omnivorous, they play an important role in aquatic food webs, including energy transfer (Aresco, 2009; Spencer, Thompson, & Hume, 1998). Whereas many wetland species exclusively inhabit the aquatic environments of wetlands, freshwater turtles nest in terrestrial environments so require both aquatic and terrestrial environments to complete their life cycle (Burke, Lovich, & Gibbons, 2000), and are vulnerable to a variety of human impacts in these environments (Marchand & Litvaitis, 2004; Midwood, Cairns, Stoot, Cooke, & Blouin-Demers, 2015).

Urbanization is a serious threat to the survival of freshwater turtles. Its effects include increased mortality, as a result of wildlife–vehicle collisions (Aresco, 2005; Steen et al., 2006; Steen & Gibbs, 2004); a reduction in nest and recruitment success, owing to increased predation rates (Congdon, Dunham, & Van Loben Sels, 1993; Marchand & Litvaitis, 2004; Thompson, 1983); reduced fitness through decreasing genetic connectivity between subpopulations, compromising metapopulations (Forman & Alexander, 1998; Reid, Mladenoff, & Peery, 2017); and population loss, owing to the complete removal of wetlands (Bodie, 2001). Although many of these negative effects are well understood, surprisingly little information exists on how the level of terrestrial habitat alteration surrounding wetlands influences the population viability and abundance of freshwater turtles. Understanding this relationship is crucial for wetland conservation and restoration plans to enable turtles to coexist with humans in urban environments.

The southwestern snake-necked turtle (*Chelodina colliei* Gray, 1856) (formerly *Chelodina oblonga* Gray, 1841) (Georges & Thomson, 2010) is endemic to south-west Western Australia, a globally recognized biodiversity hot spot (Myers, Mittermeier, Mittermeier, da Fonseca, & Kent, 2000). It is currently listed as 'Near Threatened' by the International Union for Conservation of Nature (IUCN) (Tortoise & Freshwater Turtle Specialist Group, 1996) although its status has not been assessed for more than 20 years. A recent study of *C. colliei* suggested that the majority of urban wetlands surveyed had very small populations, indicating these populations may be in decline (Bartholomaeus, 2016). *Chelodina colliei* populations may have declined in urban areas as a result of the continuing land use development surrounding wetlands, which is reducing the availability and access to terrestrial habitat (Giles, Kuchling, & Davis, 2008; Guyot & Kuchling, 1998). However, the latter studies were conducted over 10 years ago, at three or fewer wetlands, and thus how surrounding land use influences population size and structure has not been quantified.

Using *C. colliei* as a model species, the aim of this study was to assess the impact of habitat alteration surrounding urban wetlands on the abundance and population structure of freshwater turtles. Specific aims were to determine the abundance and population structure in a range of wetlands with differing degrees of land use modification, and to identify factors most associated with *C. colliei* abundance and population structure in urban wetlands. It was predicted that abundance of *C. colliei*, and presence of juvenile turtles would be positively associated with the amount of and degree of access to terrestrial buffers of remnant native vegetation around wetlands, as relatively undisturbed habitat is required by turtles for nesting (Bodie, 2001; Burke & Gibbons, 1995; Clay, 1981).

2 | MATERIALS AND METHODS

2.1 | Study region and wetland selection

The range of *C. colliei* includes wetlands on the Swan Coastal Plain (SCP), upon which lies the Perth metropolitan region, in southwestern Australia (Figure 1). Perth has undergone considerable urban sprawl since the 1950s, and is ranked 63rd in area but 277th in population size among cities globally (Demographia, 2018). In the 160 years since European settlement, more than 70% of SCP wetlands have been destroyed by urbanization and agriculture (Halse, 1989). The remaining wetlands are still under threat from continued urban development, including land clearing (Hiller, Melotte, & Hiller, 2013) and increased groundwater extraction (McFarlane et al., 2012). Therefore, the SCP represents an ideal region to quantify the impacts of habitat alteration on urban populations of freshwater turtles.

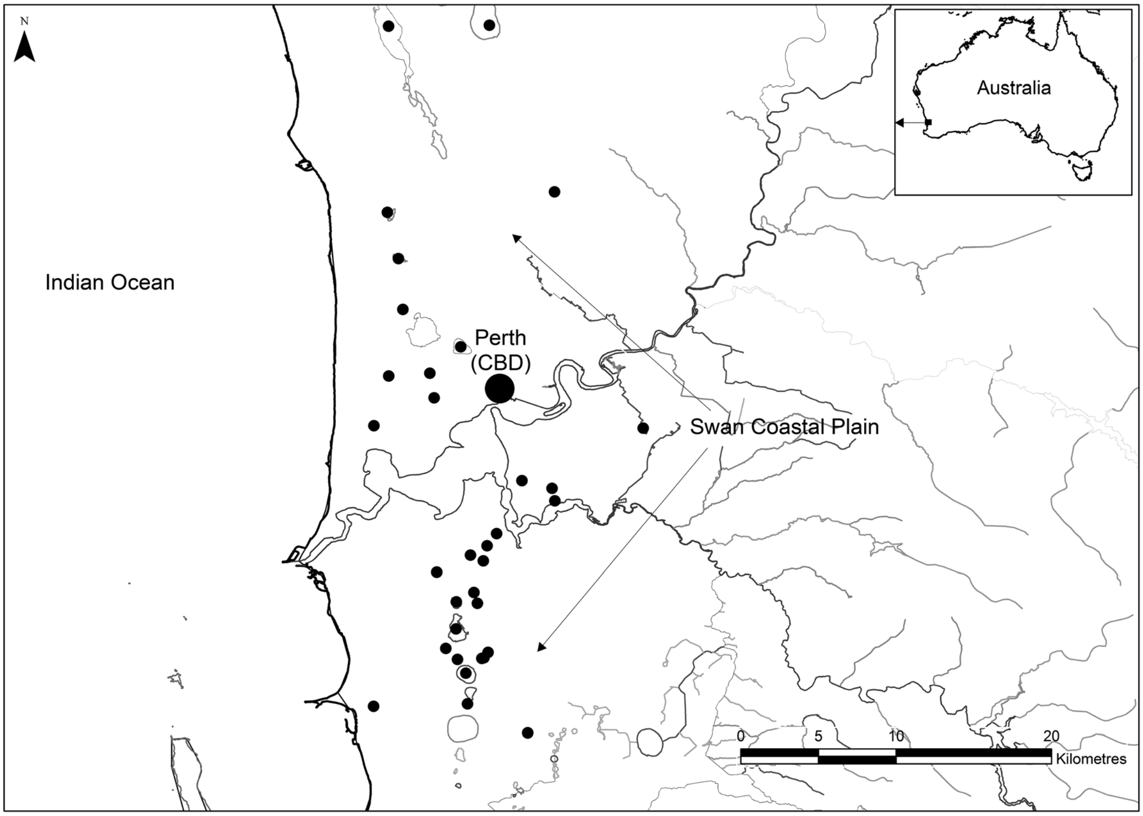

FIGURE 1 The locations of the 33 wetlands sampled for *Chelodina colliei* between October 2016 and February 2017 on the Swan Coastal Plain, south-west Australia

South-western Australia experiences a Mediterranean climate, typified by cool, wet winters and hot, dry summers (Sturman & Tapper, 1996). SCP wetlands are relatively shallow (<3 m deep), are located primarily in inter-dunal swales (Horwitz, Rogan, Halse, Davis, & Sommer, 2009), and have a range of water regimes (permanently, seasonally, or episodically flooded) (Hill, Semeniuk, Semeniuk, & Del Marco, 1996). Naturally occurring SCP wetlands are surface expressions of groundwater influenced by annual rainfall (Boulton et al., 2014), of which nearly two-thirds falls in winter (June–August) (Australian Bureau of Meteorology, 2017). Many wetlands were created as part of urban development. These include sites that were probably once natural wetlands that have been significantly modified to meet human needs, as well as newly created lakes (Chester, Robson, & Chambers, 2013). These modified wetlands are primarily managed for human purposes; many have artificially maintained water levels and contain artificial structures such as limestone, concrete, or timber walls, and plastic or concrete liners. On the SCP, both natural and modified wetlands have considerable but varying degrees of modification of the surrounding terrestrial environment, including the replacement of native vegetation with lawn or other exotic vegetation (Chester et al., 2013).

Thirty-three SCP wetlands (lakes, ponds, and swamps) within the Perth metropolitan region (≤30 km radius from the Central Business District (CBD)) were chosen for this study (Figure 1). Wetland choice was based on the known presence of *C. colliei* populations from previous studies (e.g. Bartholomaeus, 2016; Clay, 1981; Giles, 2001; Guyot & Kuchling, 1998; Hamada, 2011; Tysoe, 2005) and to represent a range of different sized wetlands across a range of urban land uses (see below).

2.2 | Environmental variables

A combination of aerial imagery and ground-truthing was used to quantify a range of environmental variables in and around all wetlands. Land use surrounding each wetland was assessed in a band of land 300-m wide extending from the water's edge into the terrestrial zone. The 300-m zone comprises the area where turtles are observed most frequently during nesting (Bartholomaeus, 2016), and encompasses the average distances of nesting during both spring (86.6 m) and summer (25.4 m) nesting seasons (Clay, 1981). Land use within the 300-m zone was classified into one of seven land use categories, the areas of which were quantified with the polygon function in GOOGLE EARTH PRO (Google, 2017; Table 1). Residential and industrial areas contain roads, so the road category was used only for a road that was not part of, or contained within, a residential or industrial area. Land use area was standardized for wetland size by converting it to a percentage of the total area within each 300-m zone.

The perimeter of each wetland (i.e. the water's edge) was also assessed for land use type as it represented the interface between

TABLE 1 Explanatory variables used in the generalized linear mixed model (GLMM) analyses of *Chelodina colliei* catch per unit effort (CPUE) and the surrounding land use of urban wetlands in Perth, Western Australia

Variable name	Variable type	Variable description
Native vegetation	Continuous (percentage)	Percentage of remnant bushland within a 300-m band around each wetland
Rural	Continuous (percentage)	Percentage of rural land use (e.g. grasslands or agricultural areas) within a 300-m band around each wetland
Open water[a]	Continuous (percentage)	Percentage of open water land use (e.g. neighbouring wetlands and other water bodies) within a 300-m band around each wetland
Lawn	Continuous (percentage)	Percentage of lawn land use (e.g. regularly maintained grass: mowed, fertilized, etc.) within a 300-m band around each wetland
Road	Continuous (percentage)	Percentage of road land use (e.g. roads not within or adjacent to residential or industrial areas) within a 300-m band around each wetland
Industrial	Continuous (percentage)	Percentage of industrial land use (e.g. factories, hospitals, shopping centres) within a 300-m band around each wetland
Residential[a]	Continuous (percentage)	Percentage of residential land use (e.g. residential housing and small parks within suburbia) within a 300-m band around each wetland
Vegetated perimeter[a]	Continuous (percentage)	The percentage of each wetland perimeter that allowed turtles direct access to native vegetation (i.e. suitable nesting habitat)
Wetland surface area	Continuous (scaled)	The surface area (m^2) of each wetland (scaled to values from 0 to 100)
Presence of a limestone wall[a]	Discrete (Yes or No)	Whether a limestone wall (or similar) was present for the entirety of a wetland (water's edge) perimeter
Presence of an island	Discrete (Yes or No)	Whether an island (natural or artificial) was present within the wetland
Wetland hydro-regime[a]	Discrete (Seasonal or Permanent)	Whether each wetland was seasonally or permanently inundated

[a]Factors that were included in the final GLMM.

the wetland and the surrounding terrestrial environment through which *C. colliei* must pass to nest (Clay, 1981). The perimeter around the wetlands was dominated by native vegetation and/or lawn (see Results), which were highly negatively correlated. Therefore, direct accessibility of native vegetation was selected as a factor and was quantified as the percentage of the total perimeter of each wetland that was native vegetation, which was measured with the path function in the RULER tool of GOOGLE EARTH PRO (Google 2017, imagery dated 15 November 2015).

In addition to surrounding land use and perimeter vegetation, several other potential explanatory variables were included that were expected to influence *C. colliei* abundance (Table 1). Wetlands were categorized as permanently (n = 17) or seasonally inundated (n = 16) (Table S1), based on the authors' personal observations of water-level fluctuation during the current and past sampling of the wetlands. Wetlands were placed into five size classes based upon their water surface area (m^2, determined using the methods described above for land use) (Table S1). The water surface area (m^2) values for each wetland were scaled to values of 0–100. This allowed standardization with the percentage values of the land use variables before analysis. Binary variables included the presence or absence of islands (that could potentially be used for nesting), and whether a wall bordered the wetland (that could prevent individuals leaving the wetland) (Table 1).

2.3 | Sampling regime

Trapping for *C. colliei* occurred twice in each wetland in spring–summer (from 27 October 2016 to 21 February 2017) when all wetlands were inundated, and in order to avoid low temperatures in winter and early spring when the capture rate of chelonians is reduced (Chessman, 1988; Rowe & Moll, 1991). Each trapping session consisted of one overnight period: modified funnel traps (Kuchling, 2003) were set and baited (tinned sardines in vegetable oil) in the afternoon, left overnight, and checked the following morning. Individual *C. colliei* were prevented from eating the bait by only partially opening the tins, ensuring that the bait remained active through to trap retrieval. Between three and 15 traps were placed in each wetland, depending upon the size of the wetland, with a minimum distance of 25 m between traps (Table S1). Wetlands were trapped in a predetermined order that provided the most efficient routine to allow all 33 wetlands to be trapped twice in the few months available, and between 1 and 2 months passed between sessions at each wetland. Between one and four wetlands were trapped, and a maximum of 35 traps were set per session.

Upon retrieval of each trap, each *C. colliei* was weighed to the nearest gram (using Kern HDB 5KN5 electronic scales; Kern & Sohn, Balingen, Germany). Carapace length, width, and depth, plastron length, and extended tail length (from the base of the plastron to the tip of the tail) were measured to the nearest 1 mm with vernier calipers. Carapace length was used to classify individuals as juvenile (male, <129 mm; female, <159 mm) or adult (male, >130 mm; female, >160 mm) (Kuchling, 1988, 1989). Individual *C. colliei* were marked for identification by filing small v-shaped notches on the marginal scute/s following the standard numbering system for the species (Burbidge, 1967). Tail length relative to carapace length was used to determine an individual's sex (Burbidge, 1967). Following data collection, individuals were released within 5 m of their capture location.

2.4 | Data analysis

Relative abundance was estimated with catch per unit effort (CPUE) for each trapping session at each wetland. As trap effort differed among wetlands, CPUE was calculated for each wetland with the formula CPUE = T/Tn/TH, where T is the total number of *C. colliei* captured, Tn is the total number of traps, and TH is the number of trap hours. This equation standardized turtle captures to relative abundance so that they could be compared among wetlands of different size.

A generalized linear mixed model (GLMM) was used to determine the environmental variables (Table 1) that best explained variation in the relative abundance of *C. colliei*. To avoid oversaturating the GLMM with factors, univariate models were first calculated for each predictor variable, with CPUE as the dependent variable. Variables with *P* values greater than 0.3 were removed from subsequent analyses. To avoid collinearity among environmental variables, bivariate correlations (Pearson's) were calculated between remaining land use and other spatial variables, which confirmed that all bivariate correlations were <0.7. Multi-collinearity among variables was then assessed by calculating variance inflation factors (VIFs), which determine the correlation of each predictor with all others (using the R package CAR; Fox, Weisberg, & Price, 2019). In order to account for temporal autocorrelation of CPUE between sampling events, 'sampling event' was included as a random variable in the GLMM (Harrison et al., 2018), which was fitted with the package LME4 (Bates, Maechler, & Walker, 2016) in R (studio version 0.99.893, 2016; R Development Core Team, 2013).

Chi-square tests were used to determine whether the presence or absence of juveniles was associated with the proportion (high, >50%; low, <50%) of remnant native vegetation around wetland perimeters. Chi-square tests of homogeneity were used to analyse whether sex ratios were biased in *C. colliei* populations. Sex ratios (male:female) at each wetland were compared against a 1:1 ratio. These analyses were performed in R (studio version 0.99.893, 2016; R Development Core Team, 2013).

3 | RESULTS

3.1 | Land use within the 300-m terrestrial zone around wetlands

Land use in the 300-m zone around the wetlands varied considerably from wetland to wetland. Residential, native vegetation, and lawn were the most frequently occurring and greatest proportion of all land uses within the 300-m zone around the wetlands (Figures 2, S1).

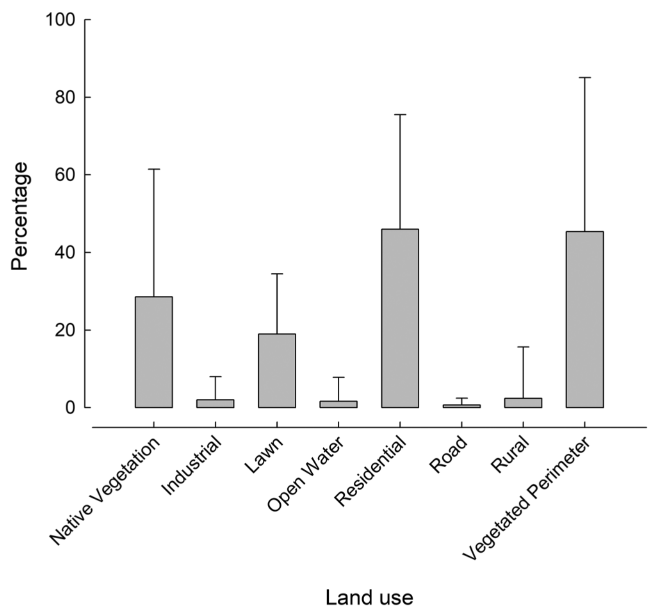

FIGURE 2 Mean (+SE) of vegetated perimeter and each land use within the 300-m zone of each wetland sampled for *Chelodina colliei*

Residential land use was present around 90.9% of wetlands and overall made up 46.0 ± 29.5% (SE) of land use within the 300-m zone of all wetlands, whereas native vegetation and lawn were present at 81.8 and 84.8% of wetlands, with means of 28.5 ± 32.9% (SE) and 19.0 ± 15.4% (SE), respectively (Figure 2).

3.2 | Land use around the perimeter of wetlands

The percentage of native vegetated perimeter around the wetlands ranged from 0 to 100% (Figure S2), with a mean of 45.3 ± 39.7% (SE) (Figure 2). Twenty-one per cent of wetlands had no direct access to remnant native vegetation, 58% had <50% of the perimeter with access to native vegetation, and a further 18% had direct access to native vegetation around the entire wetland perimeter (Figure S2).

3.3 | Relative abundance and population structure of *C. colliei*

In total, 1028 *C. colliei* (including eight recaptures) were captured during the study, 84 (8.2%) of which were juveniles. The species was captured in 32 (97.0%) wetlands, and juveniles were not captured in 13 (39.4%) wetlands (Table S2). At wetlands where *C. colliei* were captured, the CPUE (averaged across both sessions) ranged from 0.004 to 0.744 (turtles/trap/hour) (Table S2). The CPUE was <0.100 (turtles/trap/hour) at 21 (63.6%) wetlands, and was <0.200 at 32 (97.0%) wetlands, with an overall average of 0.098 (±0.022) (turtles/trap/hour). Carapace lengths ranged from 41 to 281 mm, with an average size of 181 mm (Figure 3). Only two individuals with a carapace length of <100 mm were captured (Figure 3). Sex ratios did not significantly differ from 1:1 at 25 wetlands, but were significantly different at seven wetlands, five of which showed male-biased populations (Table S3).

3.4 | Factors explaining the relative abundance of *C. colliei* and the presence of juveniles

The percentage of residential land use within the 300-m zone and the percentage of wetland perimeter bordered by native vegetation were the only significant predictors of *C. colliei* CPUE (t = 3.377, P = 0.001 and t = 2.916, P = 0.005, respectively) (Table 2). Both predictors were positively associated with CPUE. The GLMM explained 21% of the variation in CPUE of *C. colliei*. The non-significant variables in the GLMM included the continuous variable percentage of open water (in other water bodies) within the 300-m zone of each wetland, and the following categorical variables: wetland seasonal or permanent and limestone wall present or absent (Table 2). The VIFs of all variables included in the GLMM were <2.11.

The capture of juvenile *C. colliei* was associated with the wetland having >50% of its perimeter covered by remnant vegetation (χ^2_1 = 4.7239, P = 0.030). Juveniles were captured in 85.7% of wetlands that had >50% of their wetland perimeter surrounded by remnant vegetation *cf* 42.1% of wetlands with <50% remnant vegetation.

TABLE 2 Generalized linear mixed model (GLMM) results for the effects on *Chelodina colliei* catch per unit effort (CPUE) of land use surrounding urban wetlands in Perth, Western Australia

Effects	Estimate	Degrees of freedom	t	P
Vegetated perimeter	0.067	60	2.916	**0.005**
Hydro-regime (seasonal)	0.037	60	1.041	0.302
Limestone wall (yes)	−0.033	60	−0.678	0.500
Open water	−0.003	60	−0.144	0.886
Residential	0.071	60	3.377	**0.001**

Significant effects are in bold font (P < 0.05).

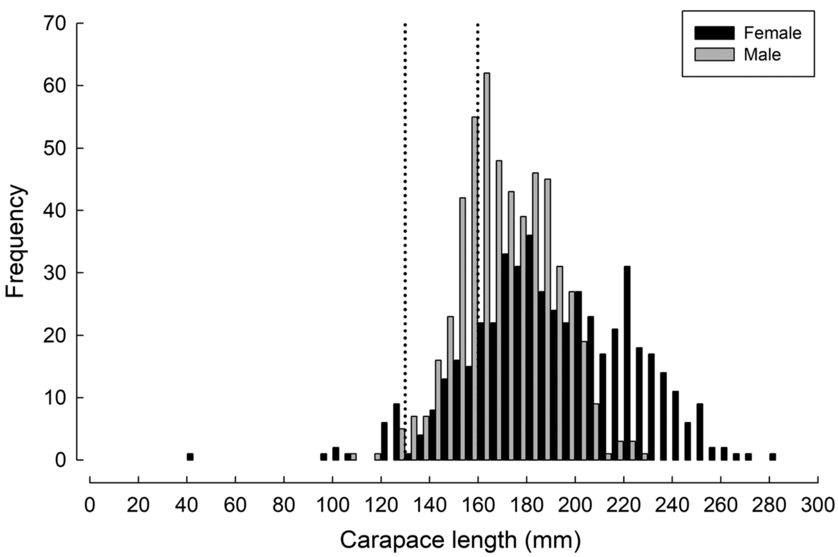

FIGURE 3 Carapace length distribution of *Chelodina colliei* pooled among the wetlands sampled. Dashed lines indicate size at sexual maturity for males (130 mm) and females (160 mm), respectively

4 | DISCUSSION

The percentage of residential land use within 300 m of the perimeter of wetlands and the percentage of the perimeter of wetlands having native vegetation were both significant predictors of the relative abundance of *C. colliei*. The relative abundance and population structure of *C. colliei* varied considerably among wetlands; however, mainly adult individuals were captured. These findings suggest that historical urbanization has affected the viability of the turtle populations, which has implications for the conservation of *C. colliei* and for other freshwater turtles occupying urban environments.

4.1 | Delayed impact of urbanization on freshwater turtles

The positive relationship between the percentage of residential land use around the wetlands and CPUE of *C. colliei* appears counter-intuitive, considering that the accessibility of native vegetation also had a significant positive effect on CPUE. This effect may be explained, however, by the fact that historical land use was not included in our model, which confounds the interpretation of the relationship. Owing to the longevity of freshwater turtles (Gibbons, 1987) and their limited capacity for rapid population increase (Beaudry, Demaynadier, & Hunter, 2008), the wetland history is likely to be highly influential. Many of the wetlands where relatively large abundances of turtles were captured during the current study are natural wetlands that have had much of their native vegetation removed within the last half-century: see Jackadder Lake (City of Stirling, 2016), Lake Gwelup (City of Stirling, 2015), Tomato Lake (City of Belmont, 2015), Neil McDougall Park (City of South Perth, 2015), and Lake Monger (City of Vincent, 2008). It is possible that the high abundances of turtles within these selected wetlands are associated with the historical surrounding land use (native vegetation) rather than with the present land use (residential), and this is reflected in the considerable level of unexplained variation in our abundance model. Despite the high abundances, the turtles captured in the aforementioned modified wetlands consisted almost entirely (99.5%) of adult turtles. This suggests that the urbanization of the terrestrial buffers surrounding these wetlands may be adversely affecting the recruitment rates of these turtle populations (Bodie, 2001; Semlitsch & Bodie, 2003), and that their abundances will inevitably decline in the future without management intervention. Unfortunately, *C. colliei* has been the focus of very few studies, and the methods and aims of the studies conducted have been diverse, preventing an accurate quantification of temporal declines in abundances. Nonetheless, the current study provides a robust baseline upon which future changes in abundances may be monitored.

Chelodina colliei has been known to use residential gardens as nesting sites (Bartholomaeus, 2010; Guyot & Kuchling, 1998). This behaviour has also been recorded previously for *Emydoidea blandingii* (in York County, Maine; Beaudry, Demaynadier, & Hunter, 2010). Residential gardens may provide 'islands' of nesting habitat within urban areas that are largely unsuitable, owing to the high percentages of impervious surfaces such as roads and sealed surfaces (Arnold & Gibbons, 1996); however, nesting females may need to traverse longer distances through urban environments in search of suitable nesting habitat (Baldwin, Marchand, & Litvaitis, 2004), thereby increasing the likelihood of crossing roads to gain access to these sites. This makes residential garden nest sites a high-risk option as roads have been linked to increased mortality in many freshwater turtle populations (Hamer, Harrison, & Stokeld, 2016; Santori, Spencer, Van Dyke, & Thompson, 2018). This mortality is generally biased towards nesting females (Steen et al., 2006; Steen & Gibbs, 2004). Population models for another Australian freshwater turtle, *Chelodina longicollis*, suggest that a yearly increase as small as 1% in adult mortality can drastically increase the risk of population extinction (Spencer, Van Dyke, & Thompson, 2017), so it is unlikely that the possibility of successful nesting in residential gardens would provide enough recruitment to sustain these populations.

It is possible that our sampling regime failed to account for juveniles and hatchlings because trapping methods similar to those used in the present study have been found to show bias against the capture of juveniles of painted turtles (*Chrysemys picta*) (Gamble, 2006; Ream & Ream, 1966). However, these methods have successfully caught juvenile *C. colliei* in previous studies of the species (Bartholomaeus, 2016; Giles et al., 2008), and juveniles were caught in some wetlands during the current study. Thus, we assume that if juveniles were present in the wetlands there would have been some representation in the traps here. The absence of juveniles in the captures from these wetlands strongly suggests a lack of recent recruitment, providing further support that the high abundances of *C. colliei* in these wetlands are likely to be the result of recruitment occurring during historical rather than current surrounding land use periods. This implies that there is a likelihood of an extinction debt for *C. colliei* populations in those wetlands.

4.2 | Role of fringing vegetation in enhancing turtle recruitment

The percentage of native vegetation on the perimeter of wetlands was positively associated with the relative abundance of *C. colliei* and correlated with the capture of juveniles. These findings suggest that past removal of native vegetation around wetlands is likely to be affecting *C. colliei* populations unfavourably. The results here broadly support the findings of a recent study on *C. longicollis* in eastern Australia, where the probability of survival in an urban landscape was positively correlated with the proportion of green open space (which included parkland) within 1 km of a wetland edge (Hamer, Harrison, & Stokeld, 2018). The current study suggests, however, that terrestrial land use change can also influence turtles at much smaller spatial scales: specifically, at the aquatic–terrestrial interface.

The terrestrial environment surrounding a wetland plays an essential role in the life cycle of freshwater turtles by providing nesting habitat (Burke & Gibbons, 1995). Although no association was found between the percentage of native vegetation within the 300-m zone and the relative abundance of *C. colliei*, this did not consider the accessibility of the native vegetation to turtles leaving the wetland (i.e. through direct connectivity). Importantly, juveniles were not captured in 58% of wetlands that had less than 50% of the perimeter connected to native vegetation. Lawn (principally parkland) was the alternative perimeter land use to native vegetation. Clay (1981) observed that *C. colliei* nested in relatively open sites, free of vegetation besides native grasses, suggesting that areas dominated by lawn may not provide suitable nesting sites. Our findings suggest that a lack of direct access to suitable nesting sites may be one of the mechanisms by which reductions in proximate native vegetation affect turtle populations.

Clay (1981) also observed that all available vegetation cover was used as protection during nesting movements. Parkland largely lacks understory species, reducing the availability of vegetative protection to adult *C. colliei* from predators such as the introduced European fox (*Vulpes vulpes*) and domestic dog (*Canis lupus familiaris*), and native birds, such as the raven (*Corvus coronoides*) (Dawson, Adams, Huston, & Fleming, 2014; Giles et al., 2008; Guyot & Kuchling, 1998; Thompson, 1983). The predation of nests and hatchlings by the introduced European fox have been shown to increase around wetlands lacking vegetative cover or a terrestrial buffer zone (Dawson et al., 2014; Giles et al., 2008). Therefore, it is likely that nesting success, and adult and hatchling survival, are impeded in wetlands with less direct access to native vegetation, thereby limiting recruitment and reducing the relative abundance of populations. These results suggest an explanation for the results of previous studies where it has been found that urban turtle populations suffer from reduced reproductive success (De Lathouder, Jones, & Balcombe, 2009).

4.3 | Implications for conservation and management

At present there is no specific conservation plan for *C. colliei* as it is not listed under threatened species legislation; however, when the results of the current study are compared with the limited number of previous studies on this species (Bartholomaeus, 2016; Clay, 1981; Giles et al., 2008; Guyot & Kuchling, 1998; Hamada, 2011; Tysoe, 2005), they suggest that declines in urban populations may have occurred and the conservation status of the species should be reviewed. It is likely that these populations will suffer continuing decline in many of these wetlands unless there is management intervention. Although mature individuals dominated the captures in most populations, the fact that *C. colliei* persists at present in even highly modified wetlands suggests that the recovery of populations may still be possible. Unfortunately, the proactive conservation of aquatic species in the early stages of decline is relatively uncommon and species are often on the brink of extinction before actions are taken (Fazey, Fischer, & Lindenmayer, 2005).

The correlations between proximate surrounding land use and turtle populations found in the current study have important conservation and management implications. The results suggest that nesting and recruitment by *C. colliei* (and possibly other freshwater turtles) may be enhanced by retaining or restoring a proportion (roughly 50% of the perimeter) of the native vegetation surrounding wetlands. This notion is broadly supported by Steen et al. (2012), who suggested that freshwater turtle population persistence is likely to be increased by managing and protecting nesting sites close to wetlands, as this will help prevent excess mortality in female turtles travelling further to nest. Restoration of fringing vegetation can also reduce erosion, improve water quality, reduce water temperatures, and increase aquatic and terrestrial biodiversity through the provision of valuable habitat for wildlife (Lovell & Sullivan, 2006; O'Toole, Robson, & Chambers, 2016). Thus, together with helping to conserve freshwater turtles, the protection and restoration of riparian vegetation would also provide a wide range of other ecological benefits to wetlands.

At present, wetlands in Western Australia are managed by a variety of government agencies, including the relevant local councils, the Department of Biodiversity, Conservation, and Attractions (Ramsar wetlands), and the Department of Water and Environmental Regulation (waterways and estuaries) (Department of Biodiversity, Conservation and Attractions, 2019). However, there are many non-government organizations, such as local wetland advocacy groups, that regularly conduct rehabilitation and restoration activities in those habitats. Increasing the resources available to these groups and coordination among government agencies is likely to enhance the conservation of remaining wetlands in this region.

Urban development and its associated land use change around wetlands is likely to be having an adverse negative influence on *C. colliei* populations, yet population recovery in many cases may be relatively straightforward, as the restoration of fringing vegetation around wetlands is a well-established management strategy. Other freshwater turtle species surviving in urbanized regions may also be candidates for this type of restoration. Understanding the factors influencing population viability together with continuing the monitoring of freshwater turtles in urban wetlands is necessary to identify and quantify changes in populations, and to enable effective management strategies to be developed.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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