The impact of urbanisation on the south-western snake-necked turtle (*Chelodina colliei*)

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By

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Declaration

I declare this thesis is my own account of my research and contains, as its main content, work that has not previously been submitted for a degree at any tertiary education institution.

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Abstract

Urbanisation is one of the most influential forms of land use change globally and continues to apply increasing pressure on biodiversity. Freshwater turtles are one of the most endangered vertebrate groups with current assessments suggesting 45-50% of all species are under threat. The habitat alteration caused by urbanisation has been identified as one of the main threats to freshwater turtle populations, but there have been relatively few studies that have quantified its impact. The status of *Chelodina colliei*, a turtle endemic to south-western Australia, in the urban wetlands of the Swan Coastal Plain (SCP) remains largely unknown. However, recent studies that have occurred in a limited number of wetlands on the SCP have suggested the populations may be experiencing declines. The overall aim of this study was to assess the impact of land use modification caused by urbanisation on *C. colliei* populations. It was predicted that increasing levels of urbanisation around wetlands would result in reduced abundances and modified population structures. Specific aims were to quantify the land use surrounding a sub-set of wetlands on the SCP, determine the abundance and population structure of *C. colliei* in those wetlands, and determine the association between land use and turtle populations. Thirty-five wetlands were sampled for *C. colliei* using modified funnel traps between October 2016 and February 2017. Surrounding land use of these wetlands was assessed and classified into different types, within three perimeters (50, 300, and 500 m from the water’s edge) using Google Earth Pro. Principal Component Analysis was used to assess if surrounding land use differed between wetlands and to classify wetlands as either natural or modified. ANOSIMs were used to assess if the categories were significantly different in all perimeters. The population structures of *C. colliei* were compared between wetland types using Mann Whitney U tests. Model averaging was used to
identify the most influential land use variables on the relative abundance of *C. colliei*. The study found that wetlands grouped dichotomously based upon surrounding land use, with natural having over 50% of the surrounding land use as bushland, whereas modified wetlands had less than 50%. *Chelodina colliei* population abundances and structures differed between those groups, with greater abundances and population viability in natural wetlands. The availability and accessibility of fringing bushland was the most important land use variable associated with the relative abundance of *C. colliei*. The positive association with bushland was likely due to it providing suitable nesting sites for *C. colliei*. This study increases our understanding of the status of *C. colliei* in urban wetlands and demonstrates, for the first time, that surrounding land use has a direct effect. It has important implications for wetland management and suggests protection and restoration of fringing bushland around urban wetlands is crucial for enhancing the viability of remnant *C. colliei* populations.
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1.0 Introduction

1.1 Urbanisation

Urbanisation is one of the main causes of land conversion globally, and is occurring at an ever-increasing rate (Pickett et al. 2011; Roe et al. 2011). It is the manifestation of an increasing proportion of the human population becoming concentrated in ever-expanding cities (Hamer and Mcdonnell 2010). Urbanisation results in the conversion of natural environments (Fig. 1.1) into human-modified systems (Fig. 1.2) (Zang et al. 2011); dominated by buildings, roads and other impervious surfaces (Hamer and Mcdonnell 2010; Stokeld et al. 2014; Ferronato et al. 2016). This process completely changes the impacted landscape, often degrading, fragmenting, isolating, or completely removing terrestrial and aquatic habitat (McKinney 2002; DeStefano and DeGraaf 2003; McKinney 2008). This negatively impacts the native flora and fauna that depend on this habitat, with reductions in biodiversity correlated with increasing levels of urbanisation (McKinney 2002; Wang et al. 2008; Rees et al. 2009; Zang et al. 2011). The composition of species present becomes dominated by generalists; usually those species with r life-history strategies (e.g. high reproductive capacities through production of more offspring, earlier sexual maturity and shorter life span) (Gadgil and Solbrig 1972; Pickett et al. 2011), as their life histories allow them to quickly colonise and dominate disturbed habitats, which are common in urbanised areas (Pickett et al. 2011). As urbanisation is relatively permanent compared to other forms of habitat loss such as logging, it is considered one of the most influential factors in current and predicted species extinctions (McKinney 2002; Conner et al. 2005; Pittman and Dorcas 2009; Goddard et al. 2010).
Figure 1.1 Natural fringing vegetation around a Swan Coastal Plain wetland

Figure 1.2 Example of an urbanised Australian suburb (Lewins 2016)
Wetlands are one of the most productive habitats on Earth, and are globally important ecosystems (Mackintosh and Davis 2013), providing habitat for a diversity of species, together with ecosystem functions and services for both terrestrial and aquatic native species, as well as humans (Semlitsch and Bodie 2003). Urbanisation and other forms of habitat modification such as draining and infilling, have led to the complete removal of a significant proportion (64-71% since 1900AD) of wetlands around the world (Wang et al. 2008; Davidson 2014). Those that have not been removed, are often modified and/or isolated within a terrestrial landscape dominated by urban development (Gibbs 2000; Bartholomaeus 2016). A large proportion of terrestrial habitat surrounding urban wetlands, as well as their flood-adapted fringing vegetation, is often degraded or removed and replaced with built environments, negatively affecting native species who require this habitat for breeding sites, shelter, provision of food, and migration (De Meester et al. 2002).

The effects of urbanisation differ from species to species (Eskew et al. 2010; Roe et al. 2011), but include influencing and altering species distribution and abundance, population structure, survival and recruitment, and causing stress and immune responses (Stokeld et al. 2014). Wildlife living in urban areas must also contend with increased mortality (Ferronato et al. 2016) through threats such as wildlife-vehicle collisions (Rees et al. 2009; Ferronato et al. 2016) and predation from introduced species like foxes, dogs and cats (Aresco 2005; De Lathouder et al. 2009). As such, a species’ ability to tolerate habitat alterations and evade urban threats largely governs whether or not they will persist in these environments (Rees et al. 2009).
1.2 Freshwater turtles

Globally, there are 317 recognised species of freshwater turtles and tortoises (Buhlmann et al. 2009). Within Australia there are 35 species, belonging to three families: Chelidae (32 species), Trionychidae (2), and Carettochelyidae (1) (Buhlmann et al. 2009). Freshwater turtles are k-strategists (rather than r-strategists favoured by urbanisation) and their life histories are characterised by long lives, high adult survival rates, delayed sexual maturity, low annual recruitment, and high nest and hatchling mortality (Congdon et al. 1993; Congdon et al. 1994; Dawson et al. 2014). A turtle’s life cycle typically has four stages, which are: egg, hatchling, juvenile and adult (Unwin 2004). All species of freshwater turtle nest on land, hatch from eggs, and are not cared for by the parents (Graham et al. 1996); they are independent from birth and feed themselves. The majority of growth occurs during the hatchling and juvenile stages, but continues throughout an individual’s life, with the length of time it takes to reach maturity varying by species (Giles et al. 2008).

Urbanisation is one of the main threats to the survival of freshwater turtles (Burgin and Ryan 2008; Pittman and Dorcas 2009; Blamires and Spencer 2013; Micheli-Campbell, Campbell et al. 2013), with species having to cope with a built environment that has slowly encroached upon their once pristine upland habitat and modified or destroyed their wetland habitat (Rees et al. 2009; Bartholomaeus 2016). In recent decades evidence has shown that this group has undergone major population declines (Hoffman et al. 2010) with current assessments suggesting that 45 – 50% of all freshwater turtle species are under threat, making this group one of the most endangered vertebrate groups worldwide (Buhlmann et al. 2009; Rhodin et al. 2010; Stokeld 2014).
1.3 The south-western snake-necked turtle (*Chelodina colliei* Gray, 1841)

*Chelodina colliei* (previously *Chelodina oblonga* see Taxonomy below) (Fig. 1.3) has multiple common names, including the long-necked turtle, in reference to the turtle’s long neck, and the oblong turtle, in reference to the shape of its shell. The shell of the species is generally a dark brown to black colour with the underside being pale whitish/yellow in colour (Cann 1998).

![Figure 1.3 Chelodina colliei](image)
1.3.1 Taxonomy

There has been some taxonomic confusion regarding the south-western snake-necked turtle and the northern snake-necked turtle (*C. oblonga*, previously *Chelodina rugosa*). While the south-western species was known as *C. oblonga*, and the northern species as *C. rugosa* for over 45 years (Kennet *et al.* 2014), as a result of a recent International Commission on Zoological Nomenclature ruling the south-western species is now known as *C. colliei*, and the northern species as *C. oblonga* (Georges and Thomson 2010, Kuchling 2010, Kennet *et al.* 2014).

1.3.2 Distribution

*Chelodina colliei* is endemic to south-west, Western Australia (Cann 1998). The range of its distribution is from the Hill River near Jurien Bay in the north, to the Fitzgerald River National Park in the south-east (Cann 1998). It has been recorded as far inland as Toodyay, Pingelly and Katanning (Department of Environment and Conservation 2009). *Chelodina colliei* was considered relatively common throughout metropolitan Perth (Guyot and Kuchling 1998), and could be found in a wide variety of wetlands, including both natural and constructed, permanent and seasonal lakes, swamps, farm dams, damplands and rivers (Guyot and Kuchling 1998; Tysoe 2005; Bartholomaeus 2016).

1.3.3 Population demographics

A natural population structure for freshwater turtles consists of a range of sexually mature age classes (Harless and Morlock 1979), a small percentage of juveniles (due to the high mortality rates of hatchling and juveniles) (Steen and Gibbs 2004; Fordham
et al. 2009; Dawson et al. 2014; Kuchling 2015), and a sex ratio at approximate parity (Georges et al. 2006; Burgin and Ryan 2008).

1.3.4 Life-cycle

A specific lifespan has not been confirmed for *C. colliei*, however many freshwater turtle species are estimated to live for approximately 30 years (Gibbons 1987). *Chelodina colliei*’s shell can attain a carapace length (CL) of over 30 cm (Cann 1998) and sexual maturity in males is reached at approximately 13 cm CL, while females are considered to be sexually mature at approximately 15 cm CL (Kuchling 1988; 1989). These sizes may take up to a decade to reach due to its slow growth rates (Giles 2009).

Figure 1.4 The use of aquatic and terrestrial habitat throughout freshwater turtle life cycles. (Aerial Image: Google Earth 2016; Symbols: Courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/)
Once maturity has been reached, *C. colliei* can reproduce annually if conditions are favourable (Tysoe 2005). *Chelodina colliei* generally mate in winter and spring (Cann 1998) within the wetland (Fig. 1.4), with the female nesting seasons occurring in spring and summer in permanent systems, but only during spring in seasonal waters (Clay 1981). Female *C. colliei* start their reproductive cycle approximately 8 – 10 months before the breeding season, but can delay if conditions are unsuitable (Tysoe 2005). Once a reproductive cycle has been started, the female cannot abort its eggs like some other species (Tysoe 2005).

Female *C. colliei* construct nests in the surrounding terrestrial habitat (Fig. 1.4), preferring open sites (Clay 1981) with soft sandy soil, usually within 500 m from the water’s edge (Bartholomaeus 2010; Bartholomaeus 2016), but have been reported to travel up to 1 km in search of nesting sites (Clay 1981). Females are able to produce one clutch per season, and an individual has the potential to produce up to 45 eggs over the two nesting seasons (Clay 1981). However, on average eight eggs are laid in spring and four in summer, with 25 eggs in one nest being the largest reported (Clay 1981; Bush *et al.* 2010).

Historically, “seasonal rain-bearing low pressure systems, falling barometric pressure and an air temperature above 17°C” (Clay 1981) has been the cue for *C. colliei* females to nest, having been seen leaving wetlands en-masse in these conditions (Clay 1981). This masse movement usually occurs in two groups, with half of the female population moving at once, and the second half of the population moving approximately two weeks later (Phoenix Environmental Sciences 2011). However,
recently this en-masse nesting movement has been increasingly rare (Bartholomaeus 2016).

Female turtles have been recorded to spend up to an hour finding a suitable nesting site, and an additional 25 – 45 minutes to complete the nesting process, which includes digging, laying and compacting (Clay 1981). When searching for an appropriate nest site, the female turtles use available vegetation cover for protection and concealment (Clay 1981). *Chelodina colliei* eggs take approximately 210 to 222 days to incubate, and hatchlings of both seasons generally emerge from the nest in August (Clay 1981). Hatchlings have been observed waiting within nests once hatched and are suspected to only emerge from the nest once conditions are suitable for departure (Bartholomaeus 2016). A delay in emergence from nests has also been observed in hatchlings of other species of freshwater turtles; with emergence usually synchronous and triggered by an environmental cue (Doody *et al.* 2001; Spencer and Janzen 2011). *Chelodina colliei* hatchlings have an average carapace length of 29 to 33 mm (Burbidge 1967). The overland movement, by hatchlings and breeding *C. colliei* females, back to the wetland render them vulnerable to predators such as foxes (Dawson *et al.* 2014).

1.3.5 *Metapopulations*

A metapopulation (Fig. 1.5) can be defined as “a set of local populations which interact via individuals moving between populations” (Hanski and Gilpin 1991). Each population is located in a suitable patch of habitat, separated from one another by a matrix of unsuitable habitat (Hanski and Gilpin 1991). The movement of individuals between these patches promotes gene exchange and maintains the genetic diversity of the sub-populations. In aquatic systems, connectivity across this matrix can be
fundamental to population sustainability due to the highly seasonal and annual variability of their attributes such as water and resource availability and quality (Bodie 2001; Roe et al. 2009).

While *C. colliei* nesting movements and migrational patterns have not specifically been tracked before, anecdotal observations of the species and studied movements of other Australian freshwater turtle species, such as *Chelodina longicollis*, suggest that some turtles, commonly less mature individuals, may choose to migrate to other wetlands in search of new resources, such as food and mates (Roe et al. 2009; Bartholomaeus 2010). For example, turtles will migrate from permanent wetlands to temporary wetlands to take advantage of their high productivity, and then back to the permanent wetlands when the temporary wetlands dry (Roe et al. 2008). Juvenile animals often emigrate to new patches, and this dispersal is a key component in maintaining genetic diversity and connectedness of populations (Gibbs and Amato 2000). Terrestrial habitat that allows for dispersal between wetlands is thus essential for the viability of individual freshwater turtle populations, as well as the whole metapopulation (Graham et al. 1996; Roe and Georges 2007; Bartholomaeus 2016).
Figure 1.5 Example of a metapopulation. Each wetland has its own population of *Chelodina colliei* and the life cycle can be completed within each wetland. The movement of individuals between them, which helps to maintain genetic diversity and population fitness, connects these populations. (Aerial Image: Google Earth 2016; Symbols: Courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/))

1.3.6 Diet

*Chelodina colliei* is the underwater apex predator within its range and is considered a generalist feeder and an opportunistic carnivore (Woldring 2001). It feeds on a variety of macroinvertebrates that changes seasonally, as well as fish, frogs and, sometimes, even small birds (Tysoe 2005; Phoenix Environmental Sciences 2011; Bartholomaeus 2016). As *C. colliei*’s diet consists of other aquatic organisms only, *C. colliei* can only feed within the aquatic environment of the wetland (Wetland Research and Management 2013).
1.3.7 Current status

*Chelodina colliei* is currently listed as ‘near threatened’ by the IUCN (Tortoise & Freshwater Turtle Specialist Group 1996), although the listing is for *C. oblonga* and is in need of updating. Prior to this study, *C. colliei*’s status in the Perth Metropolitan region had not been fully assessed. An examination of the number of *C. colliei* captured in studies performed at Shenton Park (on the Swan Coastal Plain (SCP)) over the last two decades suggests this population may be in decline. For example, 355 *C. colliei* were captured by Guyot and Kuchling (1998), 89 by Tysoe (2005), and only 20 by Hamada (2011). However, the trapping techniques and effort varied between the studies and thus the results are not directly comparable. A recent study of *C. colliei* in urban wetlands by Bartholomaeus (2016) reported the majority of wetlands surveyed had depauperate populations, and many had no signs of recruitment occurring. Although not definitive evidence, it may be indicative that *C. colliei* population numbers are dwindling in these urbanised environments. It has been suggested that *C. colliei* population decline in urban areas may be due to land use development reducing the habitat available for the species (Guyot and Kuchling 1998; Tysoe 2005; Giles *et al.* 2009). However, the full impacts of urban development on *C. colliei* populations may not become apparent for many years as they are long lived, so reductions in recruitment may go unnoticed until the population is in severe decline (Tysoe 2005).

1.4 What elements of urbanisation are likely to affect *Chelodina colliei*?

For living organisms to survive, the environment in which they live must provide the basic resource requirements, which include water, food and habitat (Semlitsch and Bodie 2003; Blamires and Spencer 2013; Bartholomaeus 2016). The dry upland habitat surrounding, and providing connection between wetlands is not usually
considered part of the wetland, but for *C. colliei* it is an integral part of their habitat (Gibbons 2003).

*Chelodina colliei* is dependent upon the terrestrial habitat surrounding wetlands for nesting sites (Fig. 1.4) and migration (Fig. 1.5) (Burbidge 1967; Clay 1981; Guyot and Kuchling 1998). Therefore, removing or reducing the quality of this habitat can have major implications for freshwater turtle populations, including reduced recruitment and increased mortality (Congdon *et al.* 1993). This can adversely affect freshwater turtle populations as their life history characteristics of delayed sexual maturity and low annual recruitment make it difficult to recover from reduced recruitment and increased rates of mortality (Congdon *et al.* 1993; Congdon *et al.* 1994). Even slight (10%) decreases in recruitment or increases in adult mortality has been shown to have severe impacts to the abundance and population viability of other turtle species (Congdon *et al.* 1993; Gibbs and Steen 2005). A mark-recapture study of *C. colliei* populations in three natural urban wetlands (i.e., Booragoon Lake, Blue Gum Lake, and Piney Lakes) over the course of nine months in 2000 and 2001 found that densities were up to 10 times higher at Piney Lakes, which is bordered by large buffer zones and suburban roads, compared to Booragoon Lake, which has a reduced buffer zone and is bordered by a highway. This suggests that the type and degree of modification of the habitat surrounding wetlands may influence the abundance and densities of *C. colliei* (Giles *et al.* 2008).

1.4.1 Habitat loss

The construction of built environments causes habitat loss and fragmentation through wetland and terrestrial ecosystem degradation (Findlay and Bourdages 2000; Andrews...
Terrestrial habitat that consists of structurally complex vegetation, consisting of a canopy, natural understory, and sandy open soils are ideal for nesting (Clay 1981). Freshwater turtles such as *C. colliei* prefer nest sites that are raised above the wetland water level to reduce the chance of flooding (Booth 2010; Micheli-Campbell, Baumgartl *et al.* 2013). The availability of terrestrial habitat that is suitable for nesting around a wetland depends on the surrounding land use (Fig. 1.6) (Stokeld *et al.* 2014). Urbanisation causes a reduction in available nesting habitat in the terrestrial landscape (Semlitsch and Bodie 2003; Conner *et al.* 2005; De Lathouder *et al.* 2009). Reduced availability of nesting sites will reduce recruitment into *C. colliei* populations, resulting in demographic alteration such as a reduced number of juveniles, which will critically affect populations in the long term (Bodie 2001; Tysoe 2005; Bartholomaeus 2016).

Urbanisation can also completely remove the aquatic wetland habitat where *C. colliei* spends the majority of its life (Bartholomaeus 2016). Urbanisation on the SCP has resulted in over 70% of wetlands either being significantly modified or completely removed before the turn of the millennium (Davis and Froend 1999; Chessman *et al.* 2002). The complete removal of wetlands would likely result in a regional reduction in turtle numbers, with those *C. colliei* inhabiting those former wetlands either having to relocate or face local extinction.
Figure 1.6 Conceptual impacts of urbanisation on *Chelodina colliei* populations. The crosses represent crucial elements to the survival of *Chelodina colliei* that are prevented by urbanisation. These include access to nest sites, which leads to a lack of recruitment into the population, and the prevention of migration between populations leading to decreases in genetic fitness. (Aerial Image: Google Earth 2016; Symbols: Courtesy of the Integration and Application Network, University of Maryland Center for Environmental Science (ian.umces.edu/symbols/)

1.4.2 Habitat fragmentation

Built environments often bisect the terrestrial habitat between wetlands resulting in habitat fragmentation. Habitat fragmentation results in remaining habitat in urbanised areas consisting of relatively small patches that are dispersed throughout the landscape (Soulé 1991; Ferronato *et al*. 2016). Due to a lack of connectivity, many species find it increasingly difficult to disperse between these patches as the distance between them increases (Soulé 1991). *Chelodina colliei* movement across terrestrial environments may be obstructed by built infrastructure, sometimes trapping them within the modified habitat (Stokeld *et al*. 2014; Bartholomaeus 2016). Increasing isolation of
populations impacts upon *C. colliei*’s access to new resources including mates, prey, and shelter, compromising metapopulations (Fig. 1.6) (Aresco 2005; Tysoe 2005; Giles *et al.* 2008; Steen *et al.* 2012; Stokeld *et al.* 2014; Hamer *et al.* 2016).

When freshwater turtles are unable to leave low quality habitat to find higher quality habitat, population viability may decline (Stokeld *et al.* 2014). Isolation of populations can increase the occurrence of inbreeding, and remove the possibility of immigrants improving a population that is at risk due to an unbalanced sex-ratio, or that has suffered an incident of unusually high mortality, thus reducing overall resilience and increasing risk of local extinction (Soulé 1991; Epps *et al.* 2007; De Lathouder *et al.* 2009). This risk is exacerbated by the inability of populations to be sustained in such small patches of habitat due to a lack of critical resources (De Lathouder *et al.* 2009; Rees *et al.* 2009; Bartholomaeus 2016).

1.4.3 *Increased presence of roads*

Urbanisation increases the presence of roads around wetlands. The type of road adjacent to the wetland can have a major influence on the effect of wildlife-vehicle collisions on *C. colliei* populations, with incidents on major roads often resulting in death and those on local roads more likely to cause injury (Tysoe 2005; Giles *et al.* 2008). The increases in mortality are often sex-specific, affecting females more than males due to their nesting movements (Guyot and Kuchling 1998; Aresco 2005; Gibbs and Steen 2005; Steen *et al.* 2006; Steen *et al.* 2012; Bartholomaeus 2016). As freshwater turtles nest around dawn and dusk, the movements of females often corresponds with peak traffic periods, thus increasing the risk of injury and mortality.
(Steen and Gibbs 2004). This sex-specific mortality can lead to male-biased sex-ratios resulting in reduced recruitment through a lack of reproducing females in the population (Steen and Gibbs 2004; Bartholomaeus 2016). Steen and Gibbs (2004) found that populations of *Chrysemys picta* and *Chelydra serpentina* surrounded by high road densities had approximately 20% more males than populations surrounded by low road densities. Bartholomaeus (2016) found that populations of *C. colliei* in anthropogenic wetlands had a significantly higher proportion of males than natural wetlands, and that juveniles were noticeably absent from anthropogenic wetlands.

The presence of roads also has implications upon hatchlings, who may need to cross roads to access a wetland, as they are unable to climb vertical kerbs (Guyot and Kuchling 1998; Tysoe 2005; Bartholomaeus 2016). Increased levels of mortality in hatchlings reduces recruitment into the population, altering the age class distribution of the population and resulting in an aging population (Marchand and Litvaitis 2004b; Steen and Gibbs 2004; Aresco 2005; Andrews *et al.* 2008; Roe *et al.* 2011). If the recruitment continues to fail, it will eventually leading to local extinction (Marchand and Litvaitis 2004b; Gibbs and Steen 2005; Steen *et al.* 2006; Andrews *et al.* 2008; Hamer *et al.* 2016). As such, the increased presence of roads associated with urban environments has the potential to impact most life history stages of *C. colliei*.

### 1.5 Current study

*Chelodina colliei* is known to continue to occupy wetlands that have had the surrounding terrestrial habitat converted to gardens, golf courses and parkland areas for recreational purposes (Guyot and Kuchling 1998; Phoenix Environmental Sciences 2011). In these environments, *C. colliei* may use residential gardens to nest, especially
if the urban properties are on higher ground (Guyot and Kuchling 1998; Giles 2001; Bartholomaeus 2010).

The findings of Bartholomaeus (2016) have highlighted that *C. colliei* populations in urban wetlands may be under increasing pressure from urbanisation, and thus the relative abundance and viability of *C. colliei* populations occupying wetlands of varying degrees of land use modification is in need of further investigation. The overall aim of the current study is to assess the impact of urbanisation on *C. colliei* populations. It is predicted that increasing levels of urbanisation around wetlands will result in reduced abundances and modified population structures. Specifically, the study aims to assess the following hypothesis:

*Relative abundance of turtles and their population structure will be affected by the degree of surrounding land use modification*

To investigate this hypothesis, the thesis has four aims.

1. Classify wetlands based on the degree of modification to surrounding land use.

2. Compare turtle abundance and population structure in wetlands with differing degrees of land use modification.

3. Investigate relationships between land use type and turtle population characteristics (abundance and structure).
4. Assess whether urban *C. colliei* populations may be declining by comparing my results with previous studies.

Additionally, the study has the following subsidiary aims:

1. Assess whether less permanent marking techniques such as paint and nail polish are effective for mark-recapture studies of freshwater turtle populations.
2. Assess the efficiency of two commonly used trap types: fyke nets and modified funnel traps.
2.0 Materials and Methods

2.1 Study region

Perth, the capital of Western Australia (WA), with an estimated population of 2.06 million, it is the fourth-most populous city in Australia (Australian Bureau of Statistics 2017). Perth is located on the SCP in the south-western region of WA, a recognised biodiversity hotspot (Myers et al. 2000). The SCP is bordered by the Indian Ocean to the west and the Darling Scarp to the east, extending for almost 100 km to cover an area of approximately 42,000 km². It has a Mediterranean climate with cool, wet winters and hot, dry summers (Tapper and Tapper 1996).

A variety of shallow (<3 m deep) wetlands exist on the SCP, primarily in inter-dunal swales (Horwitz et al. 2009), and include water bodies that are permanently, seasonally or episodically flooded (Chessman et al. 2002). These wetlands are expressions of the groundwater table (Chester et al. 2013), and are influenced by annual rainfall, of which nearly two thirds falls in the winter months (June – August) (Australian Bureau of Meteorology, 2017). In the 160 years since European settlement over 70% of the SCP wetlands have been destroyed, usually as a result of activities associated with urbanisation (such as draining and infilling) and agriculture (Chessman et al. 2002). The remaining natural wetlands include Ramsar sites (Ramsar 2017) or Wetlands of National Importance, and many are found within regional parks, state forest, recreation and/or nature conservation reserves (Department of Parks and Wildlife 2017). As the last representatives of a coastal wetland system that was previously more widespread and connected they have significant value (Horwitz et al. 2009; Chester et al. 2013). However, these wetlands are still under threat
from continuing urbanisation (land clearing, groundwater extraction) as well as climate change. Many of these wetlands that were once perennial are now seasonal, and some that were seasonal have remained dry for over a decade (Chester et al. 2013).

Natural wetlands are not the only water bodies found on the SCP. There are a large and increasing number of ‘anthropogenic’ wetlands. These include sites that were likely once natural wetlands that have been significantly modified to meet human needs, and also newly created lakes. Many of these artificial wetlands were created within old claypits or quarries, or created for the disposal of stormwater, the irrigation of playing fields or to add aesthetic features to public open space (Chester et al. 2013). As these wetlands are primarily managed for human purposes many have artificially maintained water levels, and contain conspicuous artificial structures such as vertical limestone, concrete or timber walls, plastic or concrete liners, aeration fountains, and art sculptures (Lundholm and Richardson 2010). Further modifications include the removal of native vegetation and replacement with lawn or exotic specimen trees (Chester et al. 2013).

2.2 Wetland selection

Thirty-five wetlands located upon the SCP within the Perth metropolitan region (30 km radius from the city) were chosen for this study (Fig. 2.1). The wetlands were chosen based on the known presence of *C. colliei*, having information available from previous research, and to represent as many types and sizes of wetland as possible.
Figure 2.1 Aerial Image of Perth, Western Australia showing locations of all 35 wetlands sampled between 27th October 2016 and 21st February 2017. (Google Earth Pro 2017).
2.3 Habitat mapping

The land use surrounding each wetland was assessed within perimeters 50, 300 and 500 m radius from the water’s edge. The 50 m perimeter was chosen to assess the quality and area of vegetation in the interface between the wetlands water and the surrounding terrestrial habitat. The 300 m perimeter captures the area in which turtles have been most frequently observed (Bartholomaeus 2016), and the 500 m perimeter captures the furthest distance that turtles have been observed (Bartholomaeus 2016). The area within these perimeters was assessed using Google Earth Pro aerial imagery and classified into one of seven land use categories (Table 2.1). Residential and industrial areas contain roads, so the road category was strictly used for a road that was not part of or contained within a residential or industrial area. Areas of each land use category within each perimeter were calculated using the polygon function in Google Earth Pro to create polygons of each land use type (Fig 2.2). As each wetland is unique, the size of the area contained within each perimeter varied based upon the size of the wetland. Thus, land use was calculated as an area (m$^2$) and then converted to a percentage of the total area within the perimeter.

<table>
<thead>
<tr>
<th>Land use</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bushland</td>
<td>Areas of natural bushland – contain trees, shrubs and open sand</td>
</tr>
<tr>
<td>Rural</td>
<td>Areas of grasslands or agricultural land uses</td>
</tr>
<tr>
<td>Open Water</td>
<td>Open water in neighboring wetlands and other water bodies</td>
</tr>
<tr>
<td>Lawn</td>
<td>Areas of grass that were regularly maintained – mowed, fertilised.</td>
</tr>
<tr>
<td>Road</td>
<td>Roads that were not within or adjacent to suburban or industrial areas</td>
</tr>
<tr>
<td>Industrial</td>
<td>Industrial areas – includes hospitals and shopping centres</td>
</tr>
<tr>
<td>Residential</td>
<td>Residential housing and small parks within suburbia</td>
</tr>
</tbody>
</table>
Figure 2.2 Example of land use mapping using Google Earth Pro. The four perimeters in red are the water perimeter, 50, 300, and 500 m from the water. The light green polygons represent lawn, dark green represents bushland, and white represents residential. (Google Earth Pro 2017).

A measure of the proportion of each wetlands perimeter (including the internal perimeter where islands were present) that allowed turtles direct access to suitable nesting habitat was assessed. This was defined as access to bushland without being required to transition through any other type of land use (Bush perimeter). In some instances the proportion of accessibility to bushland was modified. For example, accessibility was slightly increased in several wetlands where bushland that was not abutting but in reasonable proximity (<150 m) to the wetland via lawn was assumed to be accessible to turtles. Additional variables included the length (m) of each wetland’s
perimeter, a measure of the proportion of a wetland’s perimeter that was in direct contact with lawn, and whether an island was present within a wetland.

2.4 Wetland classification

A principal component analysis was used to classify the wetlands as either natural or modified. The wetlands were also categorised as permanently (n=17) or seasonally inundated (n=18), and placed into five size classes (Table 2.2) based upon their surface area dimensions (water surface area) attained via Google Earth Pro (Google 2017, imagery dated 15th November 2015).

2.5 Sampling of *Chelodina colliei*

To assess the turtle population of each wetland, mark-recapture was conducted. Each wetland was sampled at least twice. Wetlands where a total of 25 or more individuals were captured in the first two sampling occasions were sampled a third time. There was approximately one month between trapping sessions at each wetland. The trapping was conducted from October 2016 to February 2017. This period was chosen to avoid the lower temperatures of winter and early spring where the success rate of catching Chelonians is reduced (Chessman 1988, Rowe and Moll 1991), but also to avoid seasonal wetlands drying out before trapping was completed. Two wetlands (North Lake and Perry Lakes) could not be sampled for the third session due to low water level.

A trapping session consisted of one overnight period, with traps set in the afternoon, left overnight and checked the following morning. Modified funnel traps (Kuchling
2003) (Fig 2.3), baited with tinned sardines in vegetable oil where used. The tinned sardines were ‘cracked’ but the lid was not removed. This allowed the oil and ‘sardine scent’ to disperse through the water but did not allow any turtles to consume the bait, thus the bait signal remained active through to when the traps where checked and removed. The number of traps placed at each wetland was based upon the size of the wetland (Table 2.2). The traps were set in clusters of three, with a minimum of 25 m between traps where possible. Trap clusters were evenly spread around wetlands where possible and where access allowed. If access was restricted the traps were placed in clusters of three at accessible locations.

**Table 2.2** Wetland size classification showing the range of area (m²) for each size class, the number of wetlands, how many are natural and modified, and how many are permanent and seasonal within each size, and the number of modified funnel traps used per sampling occasion.

<table>
<thead>
<tr>
<th>Size Category</th>
<th>Smallest area (m²)</th>
<th>Largest area (m²)</th>
<th>Frequency</th>
<th>Natural/Modified</th>
<th>Permanent/Seasonal</th>
<th>Funnel Traps</th>
</tr>
</thead>
<tbody>
<tr>
<td>X Small</td>
<td>1</td>
<td>5500</td>
<td>6</td>
<td>2 / 4</td>
<td>3 / 3</td>
<td>3</td>
</tr>
<tr>
<td>Small</td>
<td>5501</td>
<td>20000</td>
<td>8</td>
<td>2 / 6</td>
<td>5 / 3</td>
<td>6</td>
</tr>
<tr>
<td>Medium</td>
<td>20001</td>
<td>100000</td>
<td>8</td>
<td>4 / 4</td>
<td>3 / 5</td>
<td>9</td>
</tr>
<tr>
<td>Large</td>
<td>100001</td>
<td>600000</td>
<td>8</td>
<td>5 / 3</td>
<td>3 / 5</td>
<td>12</td>
</tr>
<tr>
<td>X Large</td>
<td>600001</td>
<td>4350000</td>
<td>5</td>
<td>4 / 1</td>
<td>3 / 2</td>
<td>15</td>
</tr>
</tbody>
</table>

All traps were labeled with a tag stating the trap was part of a study by Murdoch University, and had researcher contact details, Department of Fisheries, Department of Parks and Wildlife, and Murdoch University Animal Ethics permit numbers on the reverse side to reduce the chance of traps being removed by members of the public.
2.6 Processing

Following removal of traps from the wetland, each turtle was placed into a clean calico bag to reduce stress while awaiting processing. Each turtle captured was weighed using Kern HDB 5KN5 hanging electronic scales. Using 30 cm vernier calipers the 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. Turtles were classified into age classes based upon their carapace length (Table 2.3). To mark the turtles for identification, small “v” shaped notches were filed on the marginal scute/s following the numbering system shown in Figure 2.4. The sex of each turtle was identified in the field based on tail length (Burbidge 1967). Once processed, turtles were released within 5m of the location they were trapped.

Table 2.3 Carapace length (cm) range of male and female *Chelodina colliei* for each age class. (Kuchling 1988; 1989; Giles 2012).

<table>
<thead>
<tr>
<th>Age Class</th>
<th>Male</th>
<th>Female</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hatchling</td>
<td>&lt; 4</td>
<td>&lt; 4</td>
</tr>
<tr>
<td>Juvenile</td>
<td>4 – 10.99</td>
<td>4 – 12.99</td>
</tr>
<tr>
<td>Sub-Adult</td>
<td>11 – 12.99</td>
<td>13 – 14.99</td>
</tr>
<tr>
<td>Adult</td>
<td>&gt;13</td>
<td>&gt;15</td>
</tr>
</tbody>
</table>
Figure 2.3 Modified funnel trap (Kuchling 2003) set in a wetland. N.B. There are two *Chelodina colliei* in this trap.
Figure 2.4 Numbering system used to mark and identify turtles. Notches are circled in red. The individual in the photo is number 95.
2.7 Trial of paint and nail polish as a marking technique

As a condition of the animal ethics approval for the project, a comparison of the potential effectiveness of alternate, less invasive marking techniques was trialed. All turtles captured during the first trapping session at each wetland were also marked with a dab of quick drying, water resistant, non-toxic nail polish and a dot of Sharpie non-toxic, permanent oil-based paint pen on the same marginal scutes that were filed. Any recaptures caught during the second and third trapping sessions were checked for the presence of these marks.

2.8 Trap comparison (fyke vs modified funnel trap)

Bartholomaeus (2016) found trapping efficiency to be a key limitation to accurately determine turtle numbers. This research investigated the limitations of trapping by comparing fyke and modified funnel traps to determine if trapping efficiency could be improved. In order to compare the potential effectiveness of alternate trapping methods, fyke traps (Fig. 2.5) were set at three wetlands. Each net was 11.2 m wide (including two 5 m wings and one 1.2 m wide mouth), had a depth of 0.8 m and a length of 5 m, and all comprised of 2 mm woven mesh.

A ratio of three modified funnel traps to one fyke trap was used. This ratio was based on catch per unit effort (CPUE) calculated as the average setup time per trap and the coverage achieved per trap. Baiting of the modified funnel traps was as described above. Fyke traps were baited with one can of tinned sardines placed in the end of the funnel section, opened in the same way as described above. The tail of the trap was tied to natural vegetation or a metal stake above the water level and a float was also
placed inside the trap to ensure there was available access to oxygen for any trapped turtles. The traps were set in groups using the 1:3 ratio.

Figure 2.5 Fyke net used to compare turtle captures with the modified funnel traps

2.9 Data analysis

Aim 1: Classify wetlands based on the degree of modification to surrounding land use

Principal Component Analysis (PCA) ordinations using the percentages of each land use for each site were undertaken for each of the 50, 300, and 500 m perimeters to determine if these variables differed between wetlands in all of the perimeters considered. The results were used to classify wetlands as natural (surrounding land use primarily natural bushland) or modified (surrounding land use primarily urban). A Euclidean distance resemblance matrix was calculated for each of the 50, 300, and 500
m perimeters, and ANOSIMs were then undertaken on all perimeters to determine if any of the differences that were observed between wetlands classified as natural and modified were significant.

A PCA was undertaken to determine if the variables the proportion of the wetlands perimeter that was bordered by bushland or by lawn, the wetland’s surface area, and the length of the wetland perimeter for each site were different for natural and modified wetlands. A resemblance matrix was calculated, and an ANOSIM was undertaken to determine if any differences observed between natural and modified wetlands were significant. The above analyses were conducted using PRIMER-E (v6).

A Mann Whitney U test was used to test if the proportion of a wetlands perimeter that was in direct contact with bushland differed between natural and modified wetlands (undertaken in R software V3.3.2).

Aim 2: Compare *Chelodina colliei* abundances and population structures in wetlands with differing degrees of land use modification

Absolute abundance population estimates were calculated for all wetlands using the Lincoln-Peterson capture-recapture model with Chapman’s modification to remove bias.

Formula: $N = (n_1 + 1)(n_2 + 1) - 1 / (m_2 + 1)$

Where $N$ = population size, $n_1$ = No. of marked animals in the population, $n_2$ = sample size, and $m_2$ = No. of marked animals in the sample.
Relative abundance was calculated using catch per unit effort (CPUE). As trap effort differed between sample sites, CPUE was calculated for each sample site using the following formula:

$$\text{CPUE} = \frac{T}{(TC \times TH_1) + (TC \times TS_n)}.$$  

Where $T =$ total turtles captured, $TC =$ trap clusters, $TH_1 =$ trap hours session 1, and $TH_n =$ trap hours session n.

This allowed for standardization of turtle captures into a relative abundance to enable an accurate comparison between wetlands.

Mann Whitney U tests were used to test for differences in CPUE, and carapace lengths of turtles in natural and modified wetlands. Chi-square tests of association were used to identify patterns of association between juvenile, and sub-adult turtle presence/absence based on whether a wetland was natural or modified. Chi-square tests were used to check if the sex ratio of turtles differed from an expected 1:1 at each wetland (Georges et al. 2006; Burgin and Ryan 2008). Chi-square tests were used to analyse sex ratios in natural and modified wetlands (all tests conducted using R software V3.3.2).

**Aim 3: Investigate relationships between land use type and turtle population characteristics (abundance and structure)**

To investigate relationships between wetlands surrounding land use characteristics and turtle population characteristics, generalised linear models (GLMs) were performed using RStudio version 3.3.2. To avoid including correlated land use and other spatial variables in the GLMs, bivariate correlations were first calculated using Pearson’s
correlation coefficient. Variables with greater than 0.7 correlation scores were removed from subsequent analysis.

A multimodel inference approach was used to determine the environmental variables that best explained the variation in turtle population characteristics. Two sets of models were created with the dependent variables set as, 1) the CPUE calculated using the above formula, and, 2) the proportion of the population that was immature (i.e., juvenile and sub-adult). The predictor variables for both set of models included the following continuous variables: the percentage of a wetlands perimeter that had direct access to bushland, the percentages of lawn, road, and bushland within the 0-300 m perimeter, the percentages of residential, rural, open water, and road in the 0-50 m perimeter, and the water surface area. The following categorical factors were also included as predictor variables: whether the wetland was natural or modified, and whether the wetland was seasonal or permanent. This model was fitted using the R package MuMIn (Barton, 2013), the global model was used to generate models using all possible combinations of predictor variables. A subset of models were generated that were within four Akaike Information Criterion (AIC) values of the best model (Akaike 1974) value. By summing the Akaike likelihood weights across all models within the top-ranked set, relative importance values for each environmental predictor variable were determined. The importance values represent the probability of each variable appearing in the AIC-best model (Burnham and Anderson, 2010). MuMIn was used to perform model averaging on the subset of models (Burnham and Anderson, 2010).
Chi-square tests of association were performed comparing the presence/absence of non-adult turtles in wetlands when grouped based on the proportion of a wetlands perimeter that had direct access to bushland, as well as the percentage of bushland within the 300 m perimeter (analyses performed in R v3.3.2).
3.0 Results

3.1 Classify wetlands based upon surrounding land use

3.1.1 Natural and modified wetland classification

The PCA comparing the land use within the 50 m perimeter of the water’s edge showed two distinct clusters of wetlands (Fig. 3.1). PC1 explained 87.3% of the variation in the data, and was the driving force behind the clusters. Clay (1981) studied the nesting behaviors of C. colliei at Thomsons Lake (a natural wetland) and observed that the mean distance C. colliei would travel to nest varied between spring (86.56 + 10.00) and summer (25.38 + 2.41). Based on Clay’s (1981) findings and the clear division of surrounding land use within 50 m at wetlands, the PC1 axis was used to classify the wetlands into two groups. The wetlands in the cluster to the left of the 0 point on the x-axis (Fig. 3.1) were defined by a large amount of bushland within the 50 m perimeter and classified as natural wetlands (Table 3.1). Those wetlands to the right were defined by large amounts of lawn or residential land use within that zone (Fig. 3.1) and classified as modified wetlands (Table 3.1). These groupings explained 93.9% of the variation in surrounding land use (ANOSIM, p < 0.01, global R = 0.939).
Figure 3.1 PCA ordination showing the similarities and differences between sites and land use within a 50 m perimeter from the water’s edge. PC1 explained 87.3% of the variation and PC2 explained 9.8%, giving a total of 97.1% of variation explained.
Table 3.1 Classification of natural and modified wetlands based on PCA of land use within 50 m of the water’s edge. Wetland numbers in this table correlate with numbers in the PCA.

<table>
<thead>
<tr>
<th>Modified Wetlands</th>
<th>Natural Wetlands</th>
</tr>
</thead>
<tbody>
<tr>
<td>2. Bodkin Park</td>
<td>20. Big Carine Swamp</td>
</tr>
<tr>
<td>4. Emu Lake</td>
<td>22. Booragoon Lake</td>
</tr>
<tr>
<td>5. Frederick Baldwin Park</td>
<td>23. Chelodina Wetland</td>
</tr>
<tr>
<td>6. Harmony Lake</td>
<td>24. Kogolup Lake</td>
</tr>
<tr>
<td>7. Jackadder Lake</td>
<td>25. Lake Claremont</td>
</tr>
<tr>
<td>9. Juett Park</td>
<td>27. Lake Goollelal</td>
</tr>
<tr>
<td>10. Lake Coogee</td>
<td>28. Lake Jandabup</td>
</tr>
<tr>
<td>11. Lake Gwelup</td>
<td>29. Lake Joondalup</td>
</tr>
<tr>
<td>12. Lake Monger</td>
<td>30. Little Rush Lake</td>
</tr>
<tr>
<td>13. Lucken Reserve</td>
<td>31. North Lake</td>
</tr>
<tr>
<td>15. Melaleuca Swamp</td>
<td>33. South Lake</td>
</tr>
<tr>
<td>16. Neil McDougall Park</td>
<td>34. Thomsons Lake</td>
</tr>
<tr>
<td>17. Piney Lakes Ornamental</td>
<td>35. Yangebup Lake</td>
</tr>
<tr>
<td>18. Tomato Lake</td>
<td></td>
</tr>
</tbody>
</table>

3.1.2 Land use within the 50 m perimeter

Land use in the 50 m perimeter around natural wetlands was dominated by bushland, with all having >60% (mean = 89.8% ±3.2% S.E.) (Table 3.2), and eight having bushland covering 100% of this zone (Fig. 3.2). Lawns dominated the 50 m perimeter of modified wetlands with all having >16.2% (mean = 72.6% ±5.9% S.E.) (Table 3.2), and three having 100% lawn cover (Fig. 3.2). Residential land use was also prevalent.
within the 50 m perimeter, being present at 16 wetlands, 13 of which were modified.

The mean percentage of residential land use was higher at modified wetlands (13.9% ±4.8% S.E.) than natural wetlands (1.4% ±1.1% S.E.) (Table 3.2).

Table 3.2 Mean ± S.E. (range) percentage of each land use of each perimeter by wetland type

<table>
<thead>
<tr>
<th>Land use</th>
<th>Modified</th>
<th>Natural</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0 – 50 m perimeter</td>
<td></td>
</tr>
<tr>
<td>Bushland</td>
<td>10.0 ±3.3 (0.0 – 47.9)</td>
<td>89.8 ±3.2 (63.4 – 100.0)</td>
</tr>
<tr>
<td>Industrial</td>
<td>0.0 ±0.0 (0.0 – 0.0)</td>
<td>0.0 ±0.0 (0.0 – 0.0)</td>
</tr>
<tr>
<td>Lawn</td>
<td>72.6 ±5.9 (16.2 – 100.0)</td>
<td>7.1 ±2.5 (0.0 – 36.6)</td>
</tr>
<tr>
<td>Open Water</td>
<td>0.7 ±0.5 (0.0 – 8.6)</td>
<td>0.0 ±0.0 (0.0 – 0.0)</td>
</tr>
<tr>
<td>Residential</td>
<td>13.9 ±4.8 (0.0 – 79.6)</td>
<td>1.4 ±1.1 (0.0 – 18.0)</td>
</tr>
<tr>
<td>Road</td>
<td>0.2 ±0.2 (0.0 – 3.6)</td>
<td>0.1 ±0.1 (0.0 – 1.6)</td>
</tr>
<tr>
<td>Rural</td>
<td>2.6 ±2.6 (0.0 – 46.5)</td>
<td>1.5 ±1.5 (0.0 – 25.0)</td>
</tr>
<tr>
<td></td>
<td>0 – 300 m perimeter</td>
<td></td>
</tr>
<tr>
<td>Bushland</td>
<td>6.5 ±2.4 (0.0 – 30.4)</td>
<td>55.4 ±7.9 (4.1 – 100.0)</td>
</tr>
<tr>
<td>Industrial</td>
<td>2.3 ±1.7 (0.0 – 30.1)</td>
<td>1.4 ±0.9 (0.0 – 13.4)</td>
</tr>
<tr>
<td>Lawn</td>
<td>23.0 ±3.0 (3.1 – 47.2)</td>
<td>13.1 ±4.0 (0.0 – 45.6)</td>
</tr>
<tr>
<td>Open Water</td>
<td>2.9 ±1.9 (0.0 – 35.0)</td>
<td>0.0 ±0.0 (0.0 – 0.6)</td>
</tr>
<tr>
<td>Residential</td>
<td>60.6 ±6.2 (4.2 – 87.6)</td>
<td>28.1 ±5.8 (0.0 – 76.7)</td>
</tr>
<tr>
<td>Road</td>
<td>0.5 ±0.5 (0.0 – 8.9)</td>
<td>0.7 ±0.3 (0.0 – 4.0)</td>
</tr>
<tr>
<td>Rural</td>
<td>4.2 ±4.2 (0.0 – 76.4)</td>
<td>1.3 ±1.3 (0.0 – 22.0)</td>
</tr>
<tr>
<td></td>
<td>0 – 500 m perimeter</td>
<td></td>
</tr>
<tr>
<td>Bushland</td>
<td>8.0 ±2.6 (0.0 – 38.4)</td>
<td>45.4 ±7.6 (1.6 – 100.0)</td>
</tr>
<tr>
<td>Industrial</td>
<td>3.5 ±2.3 (0.0 – 35.9)</td>
<td>3.4 ±1.9 (0.0 – 30.5)</td>
</tr>
<tr>
<td>Lawn</td>
<td>15.0 ±1.8 (2.0 – 26.5)</td>
<td>10.6 ±3.1 (0.0 – 36.8)</td>
</tr>
<tr>
<td>Open Water</td>
<td>2.8 ±2.1 (0.0 – 37.8)</td>
<td>0.3 ±0.1 (0.0 – 2.2)</td>
</tr>
<tr>
<td>Residential</td>
<td>67.8 ±6.1 (2.4 – 93.4)</td>
<td>38.4 ±6.5 (0.0 – 86.6)</td>
</tr>
<tr>
<td>Road</td>
<td>0.3 ±0.3 (0.0 – 4.7)</td>
<td>0.6 ±0.2 (0.0 – 2.0)</td>
</tr>
<tr>
<td>Rural</td>
<td>2.6 ±2.6 (0.0 – 47.5)</td>
<td>1.4 ±1.0 (0.0 – 15.4)</td>
</tr>
</tbody>
</table>
Modified wetlands were a mixture of created wetlands and natural wetlands that were so highly modified that their surrounding land use was indistinguishable from artificial wetlands. For the purpose of this study, natural wetlands were defined as those that have greater than 50% of the surrounding land use within the 50 m perimeter as bushland, and modified wetlands as those that have less than 50% bushland. While this divide is useful as a guide for future reference, in reality the natural wetlands in this study had a mean of 89.8% ±3.2% S.E. bushland within the 50 m perimeter, while the modified wetlands had a mean of 10.0% ±3.3% S.E. (Table 3.2). Modified wetlands were therefore typified by highly modified land use (lawn, residential etc) within the 50 m perimeter (Table 3.2).
3.1.3 Land use within the 300 m perimeter

Greater similarity between natural and modified wetlands occurred within the 300 m perimeter, the result of increased residential land use in this perimeter for both wetland categories. There was less bushland around natural wetlands and lawn around modified wetlands. These results indicate greater land use change with increasing distance from the wetlands, with an increasing presence of built environments.

The PCA comparing land use within the 300 m perimeter showed two distinct groups of natural wetlands to the left of the PCA, however, the remaining natural wetlands showed greater variation in land use, with some grouped among the main modified wetland cluster to the right (Fig. 3.3). Despite this, an ANOSIM revealed that land use surrounding natural and modified wetlands classified using land use in the 50 m perimeter was significantly different in the 300 m perimeter (p <0.01, global R = 0.427). However, the increased similarity of land use between wetland types in the 300 m perimeter was reflected in a lower global R compared to that of the 50 m perimeter (global R = 0.939).
Figure 3.3 PCA ordination showing the similarities and differences between sites and land use within the 300 m perimeter. PC1 explained 74.6% of the variation and PC2 explained 13.6%, giving a total of 88.2% variation explained.

Similar to the land use within the 50 m perimeter upon which the classification of the two wetland types were based, there was a contrast between land use surrounding the two groups of wetlands, but it was less distinct (Fig. 3.4). The residential category was the most prevalent land use in the 300 m perimeter; present at 31 wetlands and covering over 50% of the available area at 15 wetlands, 13 of which were modified (Fig 3.4). Residential land use was present at 94.4% of wetlands in the modified category (Fig 3.4), covering a mean of 60.6% ±6.2% S.E. of available land within the 300 m perimeter, an increase of ~45% compared to the 50 m perimeter (Table 3.2). Residential land use was the second most prevalent land use at natural wetlands within the 300 m perimeter (Table 3.2), present at 82.4% of the natural wetlands, an almost five-fold increase compared to the 50 m perimeter (Fig. 3.4). The mean cover of
residential land use increased by ~25% to 28.1% ±5.8% S.E. within the 300 m perimeter at natural wetlands, indicating increasing levels of modification to these wetlands, but it was still much less than surrounding modified wetlands (60.6% ±6.2% S.E) (Table 3.2).

The increased presence of residential land use within the 300 m perimeter (compared to the 50 m) perimeter at natural wetlands reduced the dominance of bushland at these wetlands, with the mean cover decreasing by ~30% to 55.4% ±7.9% S.E. (Table 3.2). However, bushland cover was still higher than at modified wetlands where bushland was only present at 66.7% of the wetlands (Fig. 3.4) and the mean cover was only 6.5% ±2.4% S.E. (Table 3.2). The natural wetlands could be grouped into three types based on bushland presence in the 300 m perimeter. There were those that had over 85% bushland (Fig. 3.4), which were those in the left most cluster in the PCA (Fig. 3.3), a group with approximately 65% bushland (Fig. 3.4) which formed the 2nd cluster on the left of the PCA (Fig. 3.3) and the remaining natural wetlands which had less than 30% bushland (Fig. 3.4), which were intermingled within the modified cluster in the PCA (Fig. 3.3).

The increased presence of residential land use within the 300 m (compared to the 50 m) perimeter at modified wetlands removed the dominance of lawn at these wetlands, with the mean cover decreasing by ~50% to 23.0% ±3.0% S.E. (Table 3.2). However, lawn cover was still higher than at natural wetlands, where lawn was only present at just over half of the natural wetlands (Fig. 3.4), with a mean cover of 13.1% ±4.0% S.E. (Table 3.2).
3.1.4 Land use within the 500 m perimeter

The trends of decreasing bushland cover around natural wetlands, decreasing lawn cover around modified wetlands, and increasing residential coverage at both wetland types continued in the 500 m perimeter, and indicates that modification of surrounding land use increases with increasing distance from the wetlands.

The PCA comparing land use within the 500 m perimeter showed a similar trend to the 300 m perimeter, with natural wetlands being able to be grouped into three distinct clusters (Left, middle, and right) (Fig. 3.5). These three clusters were almost identical to those from the 300 m perimeter, with Little Rush Lake and South Lake moving from the left most cluster to the middle cluster being the only difference. The majority
of modified wetlands were clustered together (Fig. 3.5) suggesting land use type distributions were very similar. An ANOSIM revealed that land use surrounding natural and modified wetlands was still significantly different in this outer perimeter (p <0.01, global R = 0.311). The continuing trend of lower global R compared to the previous perimeter suggested that with increased distance from the wetland, land use became increasing similar between natural and modified wetlands.

Figure 3.5 PCA ordination showing the similarities and differences between sites and land use within the 500 m perimeter. PC1 explained 80.3% of the variation and PC2 explained 10.5%, giving a total of 90.8% variation explained.

While the contrast between surrounding land use at natural and modified wetlands was obvious at some wetlands, the differences between many wetlands were indiscernible (Fig. 3.6). The trend of increasing proportions of residential land use continued from the 300 m perimeter. Residential land use was present at all modified wetlands and covered an increased mean of 67.8% ±6.1% S.E. of available land (Table 3.2). Similarly, it was present at all but two natural wetlands, covering an increased 38.4%
±6.5% S.E. of the available area (Table 3.2). While increases were seen in both wetland types, the average coverage of residential land use remained lower at natural wetlands compared to modified wetlands (Table 3.2).

The mean cover of bushland continued to decrease at natural wetlands (45.4% ±7.6% S.E.) compared to the 300 m perimeter (Table 3.2). The three clusters of natural wetlands in the PCA again were largely based upon the amount of bushland. The three wetlands to the far left of the PCA (Fig. 3.5) had above 85% bushland (Fig. 3.6), the cluster in the middle of the PCA (Fig. 3.5) was wetlands with approximately 50 to 60% bushland (Fig. 3.6), and those natural wetlands among the modified cluster (Fig. 3.5) had less than 25% bushland within the 500 m perimeter (Fig. 3.6). Bushland was present at 72% of modified wetlands (Fig. 3.6), and there was a minor increase in the mean coverage (8.0% ±2.6% S.E. up from 6.5% ±2.4% S.E.) compared to the 300 m perimeter (Table 3.2).
3.1.5 Percentage of the wetland perimeter that had direct access to suitable nesting habitat

The initial values of bushland perimeter (i.e. suitable nesting habitat) indicated that there were seven wetlands that had no direct access to suitable nesting habitat, all of which were modified (Fig. 3.7). All of the modified wetlands studied had less than 50% of their perimeter with access to bushland, whereas all natural wetlands studied except Lake Claremont had over 50% (Fig. 3.7). Of the natural wetlands, nine had direct access to bushland from the entirety of the wetland perimeter.
Figure 3.7 The percentage of each wetland’s perimeter that had direct access to bushland (i.e. suitable nesting habitat). These values include the perimeter of an island if there was one present in the wetland.

While these measurements of the perimeter allowed a comparison of the accessibility of bushland between natural and modified wetlands from an aerial perspective, these values were refined based on ground-truthing to reflect realistic accessibility by turtles. Juett Park, Melaleuca Swamp, and Piney Lakes Ornamental percentages increased to account for areas of bushland not directly adjoining to their perimeters, but readily accessed by the species via crossing relatively short distances (<150 m) over lawn.

Following these adjustments there were only four wetlands that had no direct access to suitable nesting habitat, all of which were modified (Fig. 3.8). Nine wetlands, all natural, had access to bushland habitat from the entirety of the wetlands perimeter.
(Fig. 3.8). Half of the 18 modified wetlands had less than 25% of their perimeter classified as accessible bushland (Fig. 3.8). There was a significant difference between the percentage of a wetlands perimeter that had access to bushland between natural and modified wetlands (Mann Whitney U test, p<0.001). Natural wetlands had a higher mean percentage of their perimeter with access to bushland (87.94% ±4.3% S.E) than modified wetlands (20.92% ±3.6% S.E).

Figure 3.8 The percentage of each wetlands perimeter that had direct access to bushland.

A PCA comparing wetland surface area, perimeter length, and the ground-truthed percentage of the perimeter that had access to bushland or lawn showed a distinct difference between natural and modified wetlands (Fig. 3.9). The PCA indicated modified wetlands had a higher percentage of their perimeter bordered by lawn,
whereas natural wetlands had a larger percentage bordered by bushland. An ANOSIM revealed that the surface area of the wetland, the length of their perimeter, and the percentages of their perimeters that were bordered by either bushland or lawn of natural and modified wetlands were significantly different ($p < 0.01$, global $R = 0.88$). The high global $R$ indicated that the grouping of natural and modified wetlands explained 88.0% of the variation in the data.

![Figure 3.9](image.png)

**Figure 3.9** A PCA ordination showing the similarities between sites and their surface area, perimeter length, and the amount of their perimeter that connected with bushland or lawn. PC1 explained 81.1% of the variation and PC2 explained 15.4%, giving a total of 96.4% variation explained.
3.2 Compare turtle abundance and population structure in wetlands with different land use modification

3.2.1 Total turtles captured

Over the four-month trapping period 1398 turtles were captured. At natural wetlands 960 turtles were caught, whereas only 438 turtles were captured at modified wetlands. Blue Gum Lake (natural) had the highest number of turtle captures (225, 214 not including recaptures), followed by Yangebup Lake (natural) and Lake Monger (modified) with 140 (139) and 93 (91) respectively (Fig. 3.10). An additional eight wetlands had turtle captures of >50 individuals, five of which were natural wetlands (Fig. 3.10). Broadwater Garden (modified) was the only wetland with no turtles captured. However, 18 wetlands, 11 of which were modified, had less than 25 individuals captured (Fig. 3.10).
Figure 3.10 Total number (including recaptures) of *Chelodina colliei* caught at each wetland. All wetlands sampled twice. *wetlands sampled 3 times. ** wetlands sampled once.
3.2.2 Turtle population estimates

Of the 35 wetlands sampled, seven had recaptures; with only 21 individual turtles being recaptured (Fig. 3.10). The highest population estimate, and five of the top six population estimates were for natural wetlands, while the five lowest population estimates were all for modified wetlands (Fig. 3.11). However, due to the assumptions of the Lincoln-Peterson capture-recapture model not being met, these population estimates are likely to be inaccurate and were not used in further analysis.

Figure 3.11 *Chelodina colliei* population estimates using the Lincoln-Peterson capture-recapture model with Chapman’s modification to remove bias. All wetlands sampled twice. * indicates wetlands sampled three times. ^ indicates wetlands where re-captures were recorded.
3.2.3 Catch per unit effort

The six highest CPUE (turtles / 3 modified funnel traps / hour) were recorded at natural wetlands (Fig. 3.12). The mean CPUE was higher in natural wetlands (0.413±0.115 S.E) than modified wetlands (0.176±0.033 S.E), but there was not a significant difference (Mann Whitney U test, p = 0.06697), due to the high variability of turtle CPUE across wetlands.

![Figure 3.12 Catch Per Unit Effort of Chelodina colliei in natural and modified wetlands.](image)

CPUE is commonly used in studies of fish (e.g., Salthaug and Godø 2001; Marchal et al. 2002), and has also been used for freshwater turtles (Hamer et al. 2016). To demonstrate that the CPUE values were a reasonable proxy for relative abundance in this study the CPUE value and population estimate for each wetland were compared in a scatterplot (Fig. 3.12). However, the correlation is likely to be affected by the
expected inaccuracy of the population estimates. Despite the likely inaccuracy of the population estimates, some correlation was still observed between population estimates and CPUE for each wetland ($R^2 = 0.2889$, $p < 0.001$) (Fig. 3.13).

**Figure 3.13** Comparison of population estimates and catch per unit effort values calculated with each wetland’s *Chelodina colliei* turtle data.

### 3.2.4 Size distribution

Natural wetlands had a wider range of carapace sizes, particularly the smaller, below-maturity sizes, and a higher frequency of turtles in all sizes up to 20 cm carapace length (turtles that are <5 cm (female) and <7 cm (male) above maturity length (Kuchling 1999)) (Fig. 3.14). This is indicative that recruitment was reduced in modified wetlands. Frequency of turtles in carapace lengths greater than 21 cm was relatively even between wetland types (Fig. 3.14). The mean carapace length at natural (17.5 cm ±0.1 cm S.E) and modified (19.9 cm ±0.1 cm S.E) wetlands was significantly different, with the natural wetlands mean carapace length being slightly smaller (Mann Whitney U test, $p < 0.001$).
Distributions at the majority of modified wetlands were indicative of an aging population, with a mean carapace length of 19.9 cm (Fig. 3.15). Twelve of the 17 modified wetlands had only adult turtle captures (Fig. 3.16). The majority of natural wetlands had juveniles and sub-adult turtles present (Fig. 3.16). However, Kogolup Lake, Lake Gillon, and Thomsons Lake appeared to be showing signs of aging, with no sub-adults or juveniles captured in these wetlands (Fig. 3.15). Splitting the distributions by gender supplies additional evidence of aging populations at the majority of modified wetlands, with average female carapace lengths being above 19 cm, and males above 17 cm; four centimetres greater than the respective maturity sizes of each gender (Fig. 3.16). The exceptions are Juett Park, Melaleuca Swamp and Piney
Lakes Ornamental where average carapace lengths were the three lowest of modified wetlands, and all recorded juvenile turtles (Fig. 3.16).

**Figure 3.15** Distribution of carapace length of *Chelodina colliei* by wetland. Dashed lines indicate size of sexual maturity for males (13 cm) and females (15 cm) respectively.
There was no significant difference between natural (41%) and modified wetlands (24%) when considering only juvenile presence (Fig. 3.17) ($\chi^2(1) = 0.71065, p = 0.399$), however, there was a significant difference between natural (65%) and modified wetlands (12%) when considering only sub-adult presence (Fig. 3.17) ($\chi^2(1) = 8.5833, p < 0.01$). There was a significant difference between natural and modified wetlands when considering presence of all age classes other than adults, with a higher percentage of natural wetlands (82%) containing hatchlings, juveniles and sub-adults in their populations compared to modified wetlands (29% (excluding Broadwater Gardens)) (Fig. 3.17) ($\chi^2(1) = 8.4092, p < 0.001$).
3.2.5 Sex ratio

Both genders were present in all but four of the wetlands; however, only a single turtle was caught in two wetlands (Lucken Reserve and Thomsons Lake) (Fig. 3.18). Kogolup Lake (7 turtles) and Thomsons Lake both only had males present; Lucken Reserve and Mabel Talbot Park (5 turtles) only had females present (Fig. 3.18). The ratio of males to females was skewed towards males at four modified wetlands and 11 natural wetlands, with two modified and five natural wetlands having skews equal to or greater than 2:1 (Table 3.3). The ratio was skewed towards females at 11 modified wetlands and five natural wetlands, with six modified and two natural wetlands having skews equal to or greater than 2:1 (Table 3.3). The ratio was relatively even at two modified and one natural wetlands (Table 3.3). The sex ratio was significantly
different (p <0.05) from the expected 1:1 at nine wetlands (Table 3.3). However, at 13 of the wetlands the test may not be accurate as the sample size was too small (Table 3.3).

**Figure 3.18** Proportion of each gender in each wetlands *Chelodina colliei* population.
There was a significant difference between the proportion of males and females at natural and modified wetlands ($\chi^2(1) = 12.369$, $p < 0.001$). There was a larger proportion of females in modified wetlands (55%) compared to natural wetlands (44%) (Fig. 3.19).

**Figure 3.19** Proportion of each gender of *Chelodina colliei* present in natural and modified wetlands.
Table 3.3 The ratio of male to female *Chelodina colliei* captured from each study site in Perth.

*indicates a significant difference (*P* < 0.05) from the expected 1:1. ^small sample size – test may not be accurate.

<table>
<thead>
<tr>
<th>Ratio of M:F</th>
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<td>Neil McDougall Park</td>
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<td><strong>Natural Wetlands</strong></td>
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<tr>
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<td>Yangebup Lake</td>
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3.3 Comparison with previous research

Previous research and surveys on *C. colliei* had occurred at 23 of the wetlands sampled in this study (Table 3.4). Seventeen of the wetlands had only been sampled on one occasion before this study, with an additional three having been sampled on two occasions before this study (Table 3.4). Lake Joondalup and Jualbup Lake were the most studied wetlands, both having been sampled on five occasions prior (Table 3.4). The earliest study occurred at Thomsons Lake in 1975/76 (Clay, 1981); the most recent being Bartholomaeus (2016), where 11 of the wetlands in this study were sampled between 2012 and 2013. At 17 of the wetlands, less turtles were caught in this study compared to the most recent study before, while six recorded higher captures (Table 3.4). However, due to the trapping method and effort varying between studies these results are not comparable statistically, but do serve as an indication of trends that may be occurring. The size range of turtles captured at each wetland does vary between studies, but in most cases the difference is not great (Table 3.4). The sex ratios recorded at each wetland, differ between studies, sometimes quite considerably (Table 3.4). For example, Guyot and Kuchling (1998) recorded a ratio of 1:3.67 at Jualbup Lake, while this study recorded 1:0.2 (Table 3.4).
Table 3.4 Comparison of results from previous studies and surveys completed at wetlands sampled in this study. The aims of the studies as well as trapping methods and effort varied between studies and as such this is not statistically comparable.

* only mean and standard error reported ^ measurements only provided for 5 turtles

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<th>Wetland</th>
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<td>56</td>
<td>10 – 23</td>
<td>1:2.07</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>6</td>
<td>9.8 – 18.6</td>
<td>1:0.95</td>
</tr>
<tr>
<td>Kogolup Lake</td>
<td>Hamada (2011)</td>
<td>2011</td>
<td>8</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>7</td>
<td>16.1 – 19.8</td>
<td>1:0</td>
</tr>
<tr>
<td>Little Rush Lake</td>
<td>Bartholomaeus (2016)</td>
<td>2012/13</td>
<td>102</td>
<td>9 – 24</td>
<td>1:4.21</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>43</td>
<td>12.1 – 23.0</td>
<td>1:1.3</td>
</tr>
<tr>
<td>Lucken Reserve</td>
<td>Bartholomaeus (2016)</td>
<td>2012/13</td>
<td>5</td>
<td>20 – 25</td>
<td>1:1</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>1</td>
<td>20.5</td>
<td>0.1</td>
</tr>
<tr>
<td>Mabel Talbot</td>
<td>Hamada (2011)</td>
<td>2011</td>
<td>19</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>5</td>
<td>15.2 – 23.2</td>
<td>0.5</td>
</tr>
<tr>
<td>North Lake</td>
<td>Phoenix Environmental Services (2011)</td>
<td>2011</td>
<td>0</td>
<td>-</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>44</td>
<td>4.1 – 21.3</td>
<td>1:0.5</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>41</td>
<td>13.8 – 19.9</td>
<td>1:0.5</td>
</tr>
<tr>
<td>Pinney Lake Ornamental</td>
<td>Bartholomaeus (2016)</td>
<td>2012/13</td>
<td>26</td>
<td>10 – 23</td>
<td>1:0.92</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>9</td>
<td>10.3 – 19.5</td>
<td>1:2</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>14</td>
<td>14.0 – 19.6</td>
<td>1:0.6</td>
</tr>
<tr>
<td>Thomsons</td>
<td>Clay (1981)</td>
<td>1975/76</td>
<td>76</td>
<td>11.3 – 22.8^</td>
<td>1:2.86</td>
</tr>
<tr>
<td></td>
<td>as above Santoro (2017)</td>
<td>1976/77</td>
<td>27</td>
<td>as above</td>
<td>1:0.7</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>1</td>
<td>18.1</td>
<td>1:0</td>
</tr>
<tr>
<td>Yangebup</td>
<td>Hamada (2011)</td>
<td>2011</td>
<td>28</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Santoro (2017)</td>
<td>2016/17</td>
<td>140</td>
<td>12 – 23.7</td>
<td>1:1</td>
</tr>
</tbody>
</table>
3.4 Investigate relationships between land use type and turtle populations (relative abundance and structure).

3.4.1 Factors explaining variations in CPUE and proportion of juveniles

Correlations between land uses

Before general linear modeling (GLM) could be undertaken, correlations between variables needed to be identified (Fig. 3.20). All of the land use variables within the 500 m perimeter correlated with the respective land use variable within the 300 m perimeter. As such, the 500 m land use values were removed from analysis. Surface area highly correlated with perimeter length, thus only surface area was included in the GLM. The accessibility of bushland (Bush Perimeter) was predicted to be more influential over turtle presence and abundance than accessibility of lawn (Clay 1981), so Lawn perimeter was excluded due to its high correlation with Bush Perimeter (Fig. 3.20). The variables ‘percentage of lawn within the 50 m perimeter’, and ‘percentage of bushland within the 50 m perimeter’ highly correlated with Bush perimeter so were excluded from the GLM (Fig. 3.20). The percentage of industrial land use within the 300 m perimeter correlated with the percentage of road land use within the 50 m perimeter so was excluded from the GLM (Fig. 3.20). The percentage of rural land use within the 300 m perimeter correlated with the percentage of rural within the 50 m perimeter so was excluded from the GLM (Fig. 3.20).
Figure 3.20 Heat map showing correlations between variables considered for the generalised linear models.

Generalised linear models

The model selection procedure of the generalised linear models that best described the relationship between the CPUE of turtles and the surrounding land use of the wetlands revealed the best fitting model included the variables: percentage of wetland perimeter in contact with bushland, and the percentage of bushland within the 0-300 m perimeter (Table 3.5). The model predicts that as the percentage of bushland within a 300 m perimeter increases, CPUE slightly decreases, and as the percentage of a wetlands perimeter that is bordered by bushland increases, CPUE of turtles will increase (Table 3.6).
Table 3.5 Summary information and model comparisons for the best generalised linear models within 4 AIC (GLMs, using Akaike information criterion) relating to catch per unit effort of *Chelodina colliei* in sampled Swan Coastal Plain wetlands.

<table>
<thead>
<tr>
<th>Model</th>
<th>Df</th>
<th>AIC</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bush Perimeter + Bushland300</td>
<td>4</td>
<td>19.4</td>
<td>0.14</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Not Adult Presence</td>
<td>5</td>
<td>20.2</td>
<td>0.09</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Road50</td>
<td>5</td>
<td>21.1</td>
<td>0.06</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Island Presence</td>
<td>5</td>
<td>21.2</td>
<td>0.06</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Wetland Type</td>
<td>5</td>
<td>21.6</td>
<td>0.05</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Wetland Hydro-regime</td>
<td>5</td>
<td>21.8</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Surface Area</td>
<td>5</td>
<td>21.9</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Residential50</td>
<td>5</td>
<td>21.9</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Road300</td>
<td>5</td>
<td>21.9</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Rural50</td>
<td>5</td>
<td>22.0</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Lawn300</td>
<td>5</td>
<td>22.1</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Open Water50</td>
<td>5</td>
<td>22.1</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Open Water300</td>
<td>5</td>
<td>22.1</td>
<td>0.04</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Island Presence + Non Adult Presence</td>
<td>6</td>
<td>22.4</td>
<td>0.03</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Road300</td>
<td>6</td>
<td>22.7</td>
<td>0.03</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Road50</td>
<td>6</td>
<td>22.8</td>
<td>0.03</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Wetland Hydro-regime</td>
<td>6</td>
<td>22.8</td>
<td>0.03</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Surface Area</td>
<td>6</td>
<td>22.9</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Island Presence + Road50</td>
<td>6</td>
<td>22.9</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Lawn300</td>
<td>6</td>
<td>22.9</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Wetland Type + Non Adult Presence</td>
<td>6</td>
<td>23.0</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Rural50</td>
<td>6</td>
<td>23.1</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Open Water300</td>
<td>6</td>
<td>23.1</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Residential50</td>
<td>6</td>
<td>23.1</td>
<td>0.02</td>
</tr>
<tr>
<td>Bush Perimeter + Bushland300 + Non Adult Presence + Open Water50</td>
<td>6</td>
<td>23.1</td>
<td>0.02</td>
</tr>
</tbody>
</table>

Not Adult = Proportion of the turtle population that was not mature  
Island = Island was present in wetland  
Hydro-regime = Wetland was seasonal  
Wetland Type = Wetland was ‘natural’
The influence of environmental variables on the CPUE of *C. colliei* in the wetlands sampled is shown in Table 3.6. The percentage of bushland within the 300 m perimeter, and the percentage of a wetlands perimeter that is bordered by bushland ranked as the equal most important predictor variables, and were included in 100% of the best fitting models (Table 3.6). The presence of non-adult turtles ranked as the third most important predictor variable and was only included in 36% of the best fitting models (Table 3.6). All remaining predictor variables were included in equal to or less than 11% of the best fitting models (Table 3.6). Interestingly, a wetlands hydro-regime, size, or whether an island was present were not significant predictors.

<table>
<thead>
<tr>
<th>Environmental Variable</th>
<th>Importance</th>
<th>Estimate</th>
<th>SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bush Perimeter</td>
<td>1.00</td>
<td>9.263e-03</td>
<td>2.557e-03</td>
</tr>
<tr>
<td>Bushland300</td>
<td>1.00</td>
<td>-9.339e-03</td>
<td>2.542e-03</td>
</tr>
<tr>
<td>Not Adult Presence</td>
<td>0.36</td>
<td>1.478e-01</td>
<td>1.161e-01</td>
</tr>
<tr>
<td>Island Presence (Yes)</td>
<td>0.11</td>
<td>-1.048e-01</td>
<td>1.180e-01</td>
</tr>
<tr>
<td>Road50</td>
<td>0.11</td>
<td>7.064e-02</td>
<td>8.173e-02</td>
</tr>
<tr>
<td>Wetland Type</td>
<td>0.07</td>
<td>1.343e-01</td>
<td>2.355e-01</td>
</tr>
<tr>
<td>Wetland Hydro-regime</td>
<td>0.07</td>
<td>5.970e-02</td>
<td>1.129e-01</td>
</tr>
<tr>
<td>Road300</td>
<td>0.07</td>
<td>1.443e-02</td>
<td>3.028e-02</td>
</tr>
<tr>
<td>Surface Area</td>
<td>0.07</td>
<td>-3.120e-08</td>
<td>6.736e-08</td>
</tr>
<tr>
<td>Residential50</td>
<td>0.06</td>
<td>1.159e-03</td>
<td>3.565e-03</td>
</tr>
<tr>
<td>Lawn300</td>
<td>0.06</td>
<td>-1.199e-03</td>
<td>3.968e-03</td>
</tr>
<tr>
<td>Rural50</td>
<td>0.06</td>
<td>-1.570e-03</td>
<td>5.847e-03</td>
</tr>
<tr>
<td>Open Water300</td>
<td>0.06</td>
<td>1.263e-03</td>
<td>9.082e-03</td>
</tr>
<tr>
<td>Open Water50</td>
<td>0.06</td>
<td>-2.962e-03</td>
<td>3.403e-02</td>
</tr>
</tbody>
</table>
A series of generalised linear models were compared to determine which combination of variables best explained the variation in the proportion of non-adults of *C. colliei* populations at the wetlands sampled. The models explaining the proportion of non-adults all returned non-significant results (p > 0.05). The generalised linear model that best described the relationship between the non-adult proportion of turtles and the wetlands surrounding land use only included the variable ‘the percentage of bushland within the 0-300 m perimeter’. The second and third best models included the factor wetland hydro-regime and continuous variables: percentage of bushland in the 300 m perimeter and surface area, and only continuous variables percentage of bushland in the 300 m perimeter and surface area, respectively. The percentage of bushland within the 300 m perimeter was ranked as the most important variable and included in 76% of the models, followed by surface area (47%) and wetland hydro-regime (36%).

**3.4.2 Differences in relative turtle abundance and population structure based on classification of land use**

A higher proportion of wetlands with more than 50% of their perimeter bordered by bushland had non-adult turtles present compared to wetlands with less than 50%. There was a significant difference ($\chi^2(1) = 8.1177, p < 0.01$) in the presence of non-adult turtles in wetlands based upon the amount of the wetlands perimeter was bordered by bushland (Table 3.7).
Table 3.7 Presence and absence of non-adult *Chelodina colliei* in wetlands dependent on the amount of the wetlands perimeter that is bordered by bushland.

<table>
<thead>
<tr>
<th>Percent of perimeter</th>
<th>Present</th>
<th>Absent</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 50%</td>
<td>8</td>
<td>15</td>
</tr>
<tr>
<td>51 – 100%</td>
<td>11</td>
<td>1</td>
</tr>
</tbody>
</table>

A higher proportion of wetlands with more than 50% bushland within their 300 m perimeter had non-adult turtles present compared to wetlands with less than 50%, however there was not a significant difference ($\chi^2(1) = 2.4206, p = 0.1197$) (Table 3.8). While not significantly different, the higher proportion of wetlands with >50% bushland in the 300 m perimeter containing non-adult turtles suggests that recruitment into these populations was more likely to be occurring compared to wetlands with less than 50% bushland.

Table 3.8 Presence and absence of non-adult *Chelodina colliei* in wetlands dependent on the amount of the bushland within each wetlands 300 m perimeter.

<table>
<thead>
<tr>
<th>Percent of bushland</th>
<th>Present</th>
<th>Absent</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 50%</td>
<td>11</td>
<td>14</td>
</tr>
<tr>
<td>51 – 100%</td>
<td>8</td>
<td>2</td>
</tr>
</tbody>
</table>

3.5 Trial of paint and nail polish as a marking technique

Of the 21 recaptures, 13 were from the first session at each wetland, when turtles were also marked with the paint pen and nail polish. Only one of these 13 turtles had retained the paint and nail polish when recaptured, a 7.7% retention rate. This was significantly different from the required 100% retention rate ($\chi^2(1) = \text{Inf, } p < 0.0001$).
3.6 Comparison of fyke and modified funnel traps

A total of 50 turtles were captured in the fyke traps, whereas 63 were caught in the modified funnel traps. There was not a significant difference between the number of turtles caught in each trap type ($\chi^2(1) = 1.4956$, $p = 0.2214$). As there was no significant difference observed, the author chose to use modified funnel traps for the remainder of the research, due to ease of use, and lack of bycatch.
4.0 Discussion

This is the first study to investigate the influence of surrounding land use on the abundance and structure of populations of *C. colliei*. Sampling of thirty-five wetlands revealed that *C. colliei* population abundance and structure varied widely among wetlands, an observation that is consistent with previous studies on *C. colliei* (Guyot and Kuchling 1998; Giles 2001; Tysoe 2005; Bartholomaeus 2016). The presence of, and more importantly, access to bushland was found to be the most influential factor on *C. colliei* abundance and population structure. Importantly, the species clearly favoured wetlands with large percentages of their perimeter bordered by bushland. As discussed below the study has implications for conservation of the species on the Swan Coastal Plain, as well as freshwater turtle species globally.

4.1 Impact of surrounding land use modification on *Chelodina colliei* populations

4.1.1 *Chelodina colliei* response to reduced accessible bushland

This study found that the relative abundance of *C. colliei* in urban wetlands on the SCP was positively associated with the presence and accessibility of bushland in the surrounding land use, which was also a key factor in the classification of the wetlands as natural or modified. A positive relationship was found between CPUE and Bush Perimeter was observed, whereas a slight negative relationship was observed between CPUE and Bushland300. As Bush Perimeter and Bushland50 were highly correlated, Bushland50 is highly likely to be positively related to CPUE. However, the land use assessment in the current study was based on the land use that was present around the wetlands in 2015 (date of the imagery on Google Earth Pro), and did not take into account historic land use.
Many of the ‘modified’ wetlands where relatively large abundances of turtles were recorded were historically natural wetlands that have had the majority of their bushland removed within the half-century prior to this study. For example, Jackadder Lake, Lake Gwelup, Tomato Lake, Neil McDougall Park, and Lake Monger recorded the sixth to eleventh highest CPUE for the current study, respectively. Jackadder Lake, Lake Gwelup and Tomato Lake had their surrounding areas developed in the 1970s, during which time the Mitchell Freeway was constructed alongside Lake Monger (City of Vincent 2012; City of Belmont 2015; City of Stirling 2015; City of Stirling 2016). The lake at Neil McDougall Park was created in 1966 from a swamp, and has become increasingly modified since (City of South Perth 2015). However, while still relatively abundant, the turtle populations in these modified wetlands consisted entirely of adult turtles. Therefore, as *C. colliei* is long-lived, it is likely that these are the remnants of what were once flourishing turtle populations and that the lack of recruitment can be attributed to the paucity of suitable nesting habitat.

Due to freshwater turtle’s long-lived nature, the effects of urbanisation on their populations can take decades to become apparent, with studies similar to the current study finding no relationship between turtle abundance and land use variables (Stokeld *et al.* 2014; Bartholomaeus 2016). The fact that a significant relationship was identified between relative abundance and the presence and accessibility of bushland in the current study indicates that the removal of bushland is already affecting urban *C. colliei* populations. Further support for this is evidenced by the lack of juveniles and sub-adults in a large proportion of wetlands with less than 50% of their perimeter having access to bushland. Previous research has suggested that *C. colliei* is tolerant of wide-ranging environmental conditions, including water quality (pH, dissolved
oxygen, conductivity, turbidity and nutrient (phosphorus and nitrogen) concentrations) (Bartholomaeus 2016). *Chelodina colliei’s* continuing presence (sometimes in high numbers) in wetlands where a large proportion, if not all, suitable nesting habitat has been removed, is an indication that these modified wetlands are capable of sustaining adult turtles but a lack of recruitment is reducing the size of those *C. colliei* populations.

The terrestrial habitat surrounding a wetland plays an essential role in the life cycle of *C. colliei*, as it is where nesting occurs (Clay 1981). If the terrestrial habitat surrounding a wetland does not provide suitable, and accessible nesting locations, recruitment into the population will be significantly reduced (Bodie 2001), and the population of *C. colliei* will not persist over the long term. De Lathouder et al. (2009) found that populations of freshwater turtles in urban environments in Brisbane were suffering from reduced reproductive success. Ferronato et al. (2016) observed higher mortality rates in *Chelodina longicollis* populations in suburban areas compared to those in a nature reserve, and suggested it was due to wildlife-vehicle mortality associated with terrestrial nesting movements. Steen et al. (2012) suggested that freshwater turtle population persistence is likely to be increased by managing and protecting nesting sites that are close to wetlands, to prevent excess mortality in female turtles traveling further to nest. This study provides further evidence that urbanisation is negatively affecting freshwater turtle populations through its impact on nesting and recruitment, and provides support for ensuring remnant terrestrial vegetation around wetlands is not removed or degraded.
4.1.2 Provision of habitat requirements by wetland type

Given the likely importance of the land use surrounding wetlands in the lifecycle of *C. colliei*, the current study analysed land use at increasing perimeters away from the wetlands, and quantified the changes in land use brought about by urbanisation that are common on the SCP. This revealed a dichotomy of wetlands that were classified into two broad types: natural and modified. The wetlands classified as natural in this study were characterised by high accessibility to bushland and high percentages of bushland within the 50 m perimeter. Large proportions of the original vegetation were retained around these wetlands, and thus, ground cover was generally open sandy soil, and the understory and canopy was largely intact, therefore providing most, if not all of the nesting and recruitment requirements for *C. colliei*. The parkland habitat dominating the 50 m perimeter of modified wetlands consisted of a ground cover of almost entirely lawn, largely reducing and sometimes completely removing the availability of nesting sites, due to *C. colliei*’s preference of open sandy soils (Clay 1981). The largely absent understory and sparse canopy typical of parkland habitat also provides no protection from predators for *C. colliei* compromising nesting success and hatchling survival. Thus, the land use around modified wetlands within the 50 m perimeter would significantly impede *C. colliei* nesting and recruitment.

Residential land use became more dominant in the 300 and 500 m perimeters of modified wetlands as well as some natural wetlands, and presents an entirely different challenge to *C. colliei*. Groundcover in residential areas is characterised by large proportions of impervious surface such as concrete or bitumen, and large areas of space are totally replaced with buildings. Besides residential gardens (Guyot and Kuchling 1998; Bartholomaeus 2010), this environment provides *C. colliei* with
limited nesting sites and introduces additional threats such as vehicle-related mortality (Spencer and Thompson 2003, Steen and Gibbs 2004, Spencer and Thompson 2005, Steen et al. 2006, Beaudry et al. 2008, Beaudry et al. 2010). However, as the frequency of turtle sightings decreases with increased distance from the wetland (Bartholomaeus 2016), the land use at more distant perimeters did not have significant impact upon *C. colliei* nesting success in natural wetlands, compared to the effects of intact bushland (see below).

The presence of islands, the size of a wetland, and hydro-regime (permanency) were all not significant predictors of turtle CPUE or juvenile presence. It is possible that turtles avoid nesting on islands as their natural instincts may suggest that inundation of nests may occur. Clay (1981) observed that the average height of *C. colliei* nests during the summer season was 8m above, and during the spring season 16m above the wetlands water level. While the size of a wetland was not a significant predictor of CPUE (a proxy for density) in the current study, it is likely that if all *C. colliei* requirements were met, that larger wetlands would be able to support larger *C. colliei* populations. Wetland hydro-regime was expected to be a predictor as freshwater turtles are known to migrate from permanent to seasonal wetlands as the latter are more productive when inundated (Roe et al. 2008). However, given that the majority of wetland on the SCP are now isolated, it is likely that *C. colliei* may not be able to successfully migrate between wetlands explaining its lack of influence and impeding the mixing of the metapopulation.
4.2 Differences in *Chelodina colliei* populations between wetland types

This study found that *C. colliei* population structures were significantly different between natural and modified wetlands, with natural wetlands having more natural age class distributions. While relative abundances (CPUE) were not found to be significantly different between wetland types due to the high variability across wetlands, natural wetlands had approximately two times as many turtles captured in them as modified wetlands, and a higher mean CPUE. Similarly, Sirois (2011) observed that a population of bog turtles (*Glyptemys muhlenbergii*) in an area where habitat had been reduced and degraded suffered declines in survival rates and population sizes compared to a population within high quality habitat.

4.2.1 Age class

Juvenile and sub-adult age classes were generally absent from modified wetlands; but were present in the majority of natural wetlands, which is consistent with the findings of Bartholomaeus (2016). Natural wetlands had a far greater presence of turtles in the early maturity size ranges, from 14 cm through to 20 cm, whereas more mature individuals were similarly abundant in both wetland types. Hamer *et al.* (2016) found that *C. longicollis* populations in Melbourne were similarly affected by increasing road density, with greater abundances of smaller sized turtles in wetlands with lower road densities in the surrounding terrestrial environment. A study on an *Emmys marmorata* population within an urban catchment in California found that recruitment was not occurring, and suggested it was due to the extreme difficulty of finding a suitable nesting site (Spinks *et al.* 2003). Spinks *et al.* (2003) speculated that the success of nesting in urban environments was impacted by increased moisture levels of terrestrial environments around urban wetlands from irrigation; that may be causing eggs to
crack due to increased internal pressure. The author is unsure if this could be affecting *C. colliei* as the species eggs are leathery, not hard shelled. However, the general absence of juveniles and sub-adults, combined with far fewer individuals in the early maturity size ranges in modified wetlands indicates the removal of bushland has significantly reduced recruitment into these wetlands.

While modified wetlands generally did not provide *C. colliei* populations with adequate nesting habitat, there were clear exceptions. Juveniles and sub-adults were present in the *C. colliei* populations at Juett Park, Melaleuca Swamp, and Piney Lakes Ornamental. These wetlands all have one commonality, i.e., relatively easy access to an adjacent area of natural bushland via movement over less than 150 m of lawn. The presence of juveniles and sub-adults in all of these populations is evidence that recruitment is possible in cases where bushland is accessible by traversing lawn.

A study on *C. longicollis* along a gradient from urban to nature reserve observed that recruitment was occurring at similar rates across all sites, and that abundance was higher in urban and rural environments than in the nature reserve (Ferronato et al. 2017). However, *C. longicollis* has demonstrated high propensity for overland travel and is known to travel up to 6 km between sites (Roe et al. 2009) so it is likely that some of the *C. longicollis* turtles in urban environments were able to reach nesting areas (Ferronato et al. 2017). The furthest distance *C. colliei* are commonly observed from wetlands is 500 m (Bartholomaeus 2016), thus, the reduction in connectivity and nesting sites in urban environments is likely to have a much greater impact upon *C. colliei*. Ferronato et al. (2017) also noted that a fence constructed around the nature
reserve at the time a drought broke, resulted in far greater turtles outside the fence suggesting that these turtles may have left during the drought conditions and were prevented from returning once the drought broke, which may have affected the observed abundance within the nature reserve. This illustrates the importance of migration as well as nesting in maintaining population numbers. In this study, variable catch numbers occurred between two neighboring wetlands (Chelodina Wetland and Melaleuca Swamp), which may have been due to seasonal migration of turtles taking advantage of different conditions in seasonally and perennially flooded wetlands. The results of this study and those of similar studies suggest that freshwater turtle populations are able to survive in wetlands in urban environments as long as access to nearby wetlands through bushland is not impeded, and thus management of urban wetlands should ensure the provision of accessible natural vegetation in the surrounding terrestrial environment.

The remaining modified wetlands, with the exception of Lake Gwelup, all exhibited clear signs of aging turtle populations with the average size of females captured being above 20 cm CL, and males above 17 cm CL, both well above maturity size. While natural wetlands generally had better age class distributions than modified wetlands, there were natural wetlands that were missing juvenile and sub-adult age classes. Kogolup Lake, Lake Gillon, and Thomson Lake had only adult turtles captured in this study. As nesting site availability is not likely to be an issue at these wetlands, other factors such as predation or the wetlands hydro-regimes may be factors influencing the C. colliei populations at these wetlands.
4.2.2 Sex ratio

Modified wetlands had a higher proportion of females, which was unexpected as populations of turtles in urbanised areas generally show a male-dominated skew due to increased female mortality, as they are commonly killed on roads when leaving the wetlands to nest (Steen and Gibbs 2004, Aresco 2005, Gibbs and Steen 2005, Steen et al. 2006). While the proportion was in favour of females at modified wetlands, the majority of the modified wetlands had low turtle captures, and thus skews are easily affected by the random chance of capture. For example, with the exception of Lake Gwelup, the modified wetlands where high captures occurred all had sex ratios of approximate parity. Only five of the natural wetland sex ratios were equal to or greater than 2:1 in favour of males (excluding Thomsons Lake where only 1 turtle was caught). Previous studies on C. colliei have shown variable sex ratios among populations, with two finding male skews, six female skews and eleven with general equivalence of sex ratios (Clay 1981, Guyot and Kuchling 1998, Tysoe 2005, Giles et al. 2008, McKeown 2010, Phoenix Environmental Services 2011, Dawson 2012, Giles 2012, Bartholomaeus 2016). The current findings show similar variability of C. colliei population sex ratios as previous research, however the majority of populations both within this study and in previous research do not significantly differ from approximate parity, lending support to the notion that natural C. colliei sex ratios would generally be equivalent. Further research into the cause of the variability between C. colliei populations is recommended.
4.3 Additional threats associated with urbanisation

Many of the wetlands, both natural and modified, had relatively small abundances of turtles, with fewer than 20 turtles captured. Along with the influence of bushland on recruitment, the differences in abundance could also be attributed to a variety of other factors, such as the presence of predators, invasive species, poaching, translocation, artificial barriers and climate change. Most of these factors may be highly variable depending on the wetland and therefore specific consideration of individual wetlands is warranted.

4.3.1 Predation and human interaction

The lack of juveniles and sub-adults at Lake Gwelup and Tomato Lake may be explained by predation. Both wetlands have walking paths within close proximity to the water and the author noted significant presence of humans and large numbers of dogs, sometimes not leashed, at both the wetlands. During nesting season it is highly likely that there would be many encounters between humans and/or dogs with female turtles. Clay (1981) noticed turtles were easily disturbed when first leaving the water to nest and would often retreat easily. Thus, it is likely that many nesting attempts at these wetlands may be abandoned due to encounters with humans or dogs. The author also observed dug up nests with predated eggs on two of the three sampling sessions at Lake Gwelup. While the egg remains were too small to be turtle eggs, this is an indication that there is considerable predation pressure that may also be having effects on the C. colliei population.
4.3.2 Invasive vegetation

Very few turtles were also recorded at South Lake during this study; however, more individuals were captured compared to Bartholomaeus (2016), who captured 11 turtles. The turtles captured in this study were smaller (14 - 20 cm) than those caught by Bartholomaeus (19 – 25 cm) (2016). Bartholomaeus noted that from when her study commenced to when it finished *Typha orientalis* went from a small stand to completely dominating the lake. During sampling for the current study, the lake perimeter was still almost completely choked by *T. orientalis*, but the center of the lake was largely free from the plant. The smaller size of the turtles caught in this study suggests that it is possible that some successful recruitment had occurred just prior to, or around the start time of Bartholomaeus’ (2016) study. However the absence of individuals smaller than 14 cm suggests that recruitment since Bartholomaeus’ (2016) study may have been affected by the presence of *T. orientalis* blocking exit from the wetland from almost the entirety of the perimeter.

4.3.3 Poaching and translocation

Bartholomaeus (2016) previously sampled eleven of the wetlands sampled in this study in 2012/13. While the trapping effort differed between studies, many of the wetlands recorded similar numbers of turtles caught. Three of the wetlands (Little Rush Lake, Piney Lake Ornamental and Juett Park) had significantly fewer turtles caught in the current study. Of considerable interest is the massive declines in numbers caught at Piney Lakes Ornamental and Juett Park compared to Bartholomaeus (2016). When Juett Park was sampled by Bartholomaeus (2016), 56 individuals were caught, compared to six caught during this study. Piney Lakes Ornamental sampling during this study captured nine individuals compared to the 26 caught by Bartholomaeus
The author noted that there were signs posted at the gazebo at Juett Park regarding illegal poaching of turtles occurring at the lakes, which may help explain the large declines in a short period of time. The majority of captures at Juett Park in this study were smaller than 15 cm, and the entirety of captures at both Juett Park and Piney Lakes Ornamental were below 20 cm, suggesting that these turtles may have been small enough to avoid capture by poachers. However, it is also likely that these individuals just avoided capture as the lakes appeared empty due to their now depauperate state.

Poaching was also suspected to have occurred at Chelodina Wetland, after the first two sampling sessions returned no and one turtle captures respectively. When Bartholomaeus (2016) sampled the population in 2012/13 she captured 18 individuals. This wetland was completely surrounded by natural vegetation for at least 100m in all directions, so it is unlikely that this population would disappear in five years due to natural reasons. The third sampling session resulted in eleven new individuals being captured. These turtles may have migrated from the nearby Melaleuca Swamp due to it drying out (as discussed above). The turtles caught however, where all within the same size range of the turtles caught by Bartholomaeus (2016). Therefore it is possible this is one metapopulation with connectivity and migration between Chelodina Wetland and Melaleuca Swamp. Microchipping and radio-tracking of these turtles would be valuable to determine the population dynamics between these two systems.

Jualbup Lake is an example of the additional pressures urban environments place on C. colliei. Jualbup Lake was first surveyed by Guyot and Kuchling (1998), and 355 individuals were caught, with a female dominated sex ratio (1:3.67). The most recent
study prior to the current study, an aquatic fauna survey, captured only 10 individuals: all adults and an equal sex ratio (Wetland Research and Management 2013). There were two notable events between the occurrences of these studies that may help explain the significant decline in turtle numbers. Firstly, ~500 turtles were relocated to other permanent SCP wetlands between 2002 and 2004 due to concern about the lake drying and turtles not aestivating (Wetland Research and Management 2013). Secondly, 132 turtles died during the March 2010 hail storms (City of Subiaco 2010). During the current study, all 17 individuals captured were adults, and the sex ratio dropped to a low of 0.2 females per 1 male. Recruitment into this population is unlikely to occur as the wetland is almost completely ringed by a limestone wall of approximately 1m in height, apart from a re-vegetated section to the east of the wetland. The remainder of the perimeter has only a few thin wooden planks as possible exits for female turtles looking to nest. As the wetland is completely surrounded by lawn and then suburban roads, it is likely that any females that manage to find the small wooden planks to exit the wetland to nest, would be unable to find suitable nesting sites. Further, the heavily male biased sex ratio indicates that females trying to nest may be experiencing high rates of mortality, either through wildlife-vehicle collisions or predation.

4.3.4 Climate change
Climate change may be exacerbating other anthropogenic stressors of *C. colliei* populations. The south-west of Western Australia has experienced a dramatic climatic shift since the 1970’s resulting in major rainfall reductions (Smith and Power 2015). Perry Lakes Reserve historically consisted of two water bodies that were previously surveyed by Guyot and Kuchling (1998), with 1041 turtles being captured in the two
lakes over a 6-month period. In the time since Guyot and Kuchling (1998), both lakes have become increasingly shallow, with the larger lake (where the majority of turtles (724) were captured) being permanently dry since 2007 (Town of Cambridge 2015). As there are no other water bodies nearby, it is reasonable to assume that the turtles from the dry lake would have migrated to the lake studied in the current study. However, the aquatic habitat of this smaller wetland may have been too small to support such large numbers of turtles as only 41 individuals were captured during the current study. The population consisted entirely of adults, with the exception of one individual, and was male dominated (2:1). It is possible that a large number of turtles may have tried to migrate elsewhere, however, considering the lack of available wetlands nearby, it is likely if these attempts were made, they were unsuccessful. The terrestrial habitat surrounding the wetland is suitable for nesting, but the lack of juveniles and sub-adult suggest there may be other factors limiting recruitment into this population. By reducing water levels and periods of inundation on SCP wetlands, climate change may therefore influence the migration patterns of *C. colliei* and reduce the overall amount of suitable habitats available to complete their life-cycle.

4.3.5 Cumulative effects

Obviously, the impacts discussed above may occur simultaneously in certain wetlands. Thomsons Lake and the adjacent Kogolup Lake provide examples of the likely cumulative effects of predation, climate change, and human interference on *C. colliei*. Both Thomsons Lake and Kogolup Lake are seasonal wetlands, thus, turtles inhabiting these wetlands either migrate or aestivate (enter a prolonged state of torpor) during summer. In the recent past, there has been a problem with foxes around Thomsons
Lake (Department of Conservation and Land Management 2005), and thus Kogolup Lake. To prevent access to Thomsons Lake once management of the foxes had begun, a fence was erected around the reserve (Department of Conservation and Land Management 2005). Foxes are a known predator of *C. colliei* and have been found to be the major predator of turtle nests (Dawson *et al.* 2014).

All of the individuals caught at Kogolup Lake (7) and Thomsons Lake (1) during this study were adult males. Clay (1981) previously surveyed Thomsons Lake over a four-year period, during which time 103 turtles were captured. This apparent drop in turtle abundance, and the presence of only one male at Thomsons Lake, and the similar situation at Kogolup Lake is likely to be a result of the combined effects of fox predation, mortality through dehydration and prevention of migration. It is likely that before management of foxes begun, a large proportion of the nesting females and eggs may have been predated upon, severely reducing the female *C. colliei* population over time. The reduction of females in the population reduces the chance of successful mating and thus, recruitment. Further reductions to the population are likely to have occurred to turtles during aestivation as they are easy targets for the foxes.

Those individuals that attempted migration are likely to have died of dehydration as the fence completely surrounding Thomsons Lake is likely to prevent turtle migration during the summer. The author noted that the aquatic environment of Thomsons Lake was completely choked by vegetation (*Myriophyllum* sp.) and suspects that turtles would not be able to swim efficiently or hunt effectively in this wetland, making it increasingly likely that any turtles living in this wetland would migrate and seek habitat elsewhere. There are also anecdotal (Department of Parks and Wildlife) reports
of turtle corpses around Thomsons Lake. Roe and Georges (2008) found that 45% of
*C. longicollis* who attempted aestivation died while awaiting re-flooding of a seasonal
wetland, whereas only 18% of those that migrated to permanent wetlands died.
Similarly, a large road between Kogolup Lake and Yangebup Lake would mean turtles
trying to migrate to the more permanent Yangebup Lake are likely to experience high
casualties when trying to cross that road. These human creations are likely to be
significant factors in the depauperate *C. colliei* populations at these wetlands. The
combined effects of these factors make it likely that both these populations are
experiencing extinction debt (Kuussaari *et al.* 2009).

### 4.4 Trial of alternative marking techniques

The trial of alternative marking techniques, paint pen and nail polish, did not prove to
be successful in the current study, both recording a dismal 7% retention rate. These
results correlate with the Ecological Census Techniques handbook (Blomberg and
Shine 1996) that reports paint to only last several days, thus only being useful for very
short-term studies. Further, Norton *et al.* (2014) recommends that more permanent
marking techniques are used alongside paint. The author suggests that in *C. colliei’s*
case, notching is the superior method, and the extra stress caused to the individuals
due to having to clean the shell, and wait for the paint and nail polish to dry before
releasing is not acceptable due to the extremely low retention rate.
4.5 Trap comparison

While there was no significant difference in the number of *C. colliei* caught between trap types, the author suggests that modified funnel traps were superior. They were far lighter, easier to transport, and easier to set up. Being able to set up and place three funnel traps in approximately the same time as one fyke provides further advantages in placement, in that a wider range of habitat types can be accounted for. Further, fyke traps recorded a large abundance of by-catch such as tadpoles and small freshwater fish, whereas no by-catch was recorded in the modified funnel traps. Thus, the author would recommend modified funnel traps over fyke nets for these reasons.

4.6 Study limitations and future research

The current study represents the largest number of sites sampled as part of an investigation into *C. colliei*. Nonetheless, the study would still have benefited by increasing the number of wetlands that were sampled. The variation in the number of sampling sessions at each wetland may have caused some bias if the CPUE was influenced by trap shyness of *C. colliei*. Including more wetlands in the sample size, and having a consistent number of sampling sessions across all sites could strengthen the findings of the current study.

No information exists on the effective trap radius for *C. colliei*. To account for this, wetlands were placed into size classes and the trap effort standardised across the wetlands to keep the bias consistent among the wetlands. Nonetheless, there was still potential for there being some overlap of trap attractiveness and/or CPUE being influenced by intra-wetland variation in densities of *C. colliei* due to habitat
preferences. Future research could be focused upon designating an effective radius for the modified funnel trap (Kuchling 2003), along with determining the influence of habitat preferences of *C. colliei* within wetlands to inform future population assessments.

An additional limitation of the current study was the lack of recaptures. Initially, I intended to use mark-recapture to calculate a population estimate of turtle abundance. The Lincoln-Peterson model method of population estimation relies on three assumptions: population closure, equal catchability, and zero mark loss (Williams *et al.* 2002), only two of which were met by this study. The population closure assumption assumes the population is closed to additions through births and immigrants, and deletions through deaths and emigrants (Williams *et al.* 2002). Due to the short time frame between sampling sessions in this study it is likely that this assumption was met. The equal catchability assumption assumes that all individuals have an equal chance to be captured (Williams *et al.* 2002). The low recaptures recorded in this study suggest that *C. colliei* may experience “trap shyness”, which decreases the chance of recapturing an individual after it has previously been captured. Thus, this assumption was likely to have been violated in this study. The zero mark loss assumption assumes marks placed on captured individuals will not be lost between sampling sessions (Williams *et al.* 2002). The marking procedure used in this study ensured that marks were not lost. Thus, this assumption was met. The Lincoln-Peterson population estimation model is also considered unsatisfactory in small samples (Williams *et al.* 2002). Therefore, due to the absence of recaptures at the majority of wetlands, the population estimates provided for the wetlands in the current study (Fig. 3.11) are likely to be unreliable. The low recapture rates at the wetlands
that did record recaptures mean the estimates will have very large standard errors and confidence intervals. The low recapture rates are also likely to cause the model to greatly over-estimate population sizes. For example, a comparison of the population estimates of Blue Gum Lake, Lake Monger and Lake Claremont, where more than one recapture occurred have lower estimates compared with those where one or less recaptures occurred such as Lake Gwelup, Bibra Lake, Lake Joondalup and Yangebup Lake (Fig. 3.11). Thus, the population estimates shown in Fig. 3.11 are unreliable and were not used in modeling the influence of surrounding land use on *C. colliei* populations.

The rates of recaptures of *C. colliei* has also been variable in past studies. Giles (2001) recorded a 13% recapture rate at Blue Gum Lake, and 7% at both Piney Lake (Natural) and Booragoon Lake, whereas Bartholomaeus (2016) recorded 0% at over half of the wetlands studied and a variation of between 2% and 30% at the remaining wetlands. The studies by Phoenix Environmental Services (2011) and Giles (2010) both reported no recaptures, with all other studies not mentioning recaptures. Further research is needed on trap shyness by the species to develop methods for estimating turtle population sizes. As it appears trap shyness is an issue when sampling *C. colliei*, subsequent studies could use a more intensive method of trapping that focused on one session with large numbers of traps.

Further research is also required to specifically identify habitat preferences within wetlands, as well as to better determine preferred nesting habitats in addition to the apparent preference to bushland identified in the current study. This could be achieved using radio and acoustic tracking of *C. colliei* individuals in a subset of
natural and modified wetlands. This would help to identify the ideal habitat for *C. colliei* nesting and could help in the design of the new nesting habitats.

While the effect of invasive species such a *T. orientalis* on *C. colliei* access to bushland and nesting sites is currently unknown, future research could look into the degree of impediment (and if the thickness of the stand is a factor) species such as *T. orientalis* cause, and whether *C. colliei* are likely to search for the small passages that are usually all that is left to access the terrestrial habitat. Additionally, future research could investigate how long species such as *T. orientalis* are present before they start to impact upon other aquatic flora and fauna.

### 4.7 Management implications

The effect of surrounding land use on relative abundance of *C. colliei* revealed in the current study has important management implications. The average carapace length of turtles in modified wetlands was 20 cm, and assuming an average growth rate of 0.5 cm a year once mature (Wetland Research and Management 2013), these turtles are at least 20 years old. Therefore, should those populations continue to have a lack of recruitment their abundances will continue to decline. Given the approximate 30-year life span of *C. colliei* (Gibbons 1987), these turtle populations have approximately 10 years left before they are locally extinct should no recruitment occur. Further, *C. colliei* populations in modified wetlands generally experience higher anthropogenically caused mortality rates due to increased predation and wildlife-vehicle incidents (Ferronato *et al.* 2016), thus, likely increasing their rate of decline.
The results of the current study indicated that accessibility to fringing bushland was the most important land use factor influencing the relative abundance of *C. colliei* in urban wetlands. Importantly, it appears as though *C. colliei* nesting and recruitment can be supported by as little at 3500 m² of bushland (the area of bushland available within the 50 m perimeter of Juett Park), as long as it is accessible. Revegetation to ensure wetlands have higher than 50% of their perimeter bordered by bushland is recommended, however, where this is not possible, even smaller percentages may prove beneficial in enabling recruitment. It is also recommended that these areas be dispersed around the wetland so that turtles have increased chance of finding suitable habitat from multiple exit points. In wetlands managed for primarily anthropogenic benefit such as passive recreation, these could form alternate spokes from the wetland’s perimeter of bushland and lawn.

To reduce impacts from predation, it is also recommended that nesting areas be fenced to limit access to the area by foxes and dogs. Fences could also be used to reduce the impacts of wildlife-vehicle mortality at wetlands where roads are within close proximity to the wetland. For example, Blue Gum Lake was one of the smallest wetlands, but it had the largest number of turtles captured. The land use around this wetland is highly urbanised, however, it has an area of bushland bordering two sides, while the other two sides have roads nearby. Small fences exist between the wetland and those roads and it is likely these fences have reduced turtle mortality from wildlife-vehicle incidents, and also have re-directed the turtles back into the bushland habitat, where they are likely to successfully nest.
The added benefits of recreating fringing/riparian vegetation around wetlands are providing nutrient filtering and sediment and bank stabilisation, leading to lower nutrient levels, water turbidity and sediment build up. This would reduce the need for retainer walls around wetlands and increase water quality that will lead to fewer algal blooms and pest species such as mosquitos. The reduced presence of high maintenance grass surrounding wetland perimeters will also reduce the requirement for large amounts of watering, fertiliser application and mowing, reducing costs for local councils. The reduced usage of fertiliser will also lead to less nutrients such as nitrogen and phosphorus entering the water, again leading to higher water quality and fewer algal blooms.
5.0 Conclusions

This project has increased our understanding of the effects of urbanisation on C. colliei and strongly suggests that land use change has impacted relative abundance and population structures of the species in urban wetlands. The study revealed that C. colliei relative abundance was related to the availability of fringing bushland due to its suitability for nesting sites. Importantly, C. colliei currently persists in highly modified wetlands and therefore the recovery of currently declining and aging populations is not only possible, but may be relatively straightforward. Currently, the majority of the C. colliei populations in modified wetlands consist of aging, mature individuals. Therefore, the creation of new areas of native bushland bordering the highly modified wetlands would likely result in successful nesting and increase recruitment into those populations. For natural wetlands it is crucial that remnant bushland is not removed and that degraded habitat is revegetated and restored. Our actions over the next few years are likely to determine the future of this species on the SCP. Continued monitoring of C. colliei populations in urban wetlands is also necessary to quantify changes in the populations in response to management actions and ongoing environmental change.
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