

**Impacts of Urbanisation on the Eastern Long-Necked
Turtle *Chelodina longicollis* (Testudines: Chelidae)**

By

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Abstract

Anthropogenic habitat fragmentation threatens the viability of many wildlife species, and has contributed substantially to the current reptile extinction crisis. Urbanisation is one of the most damaging forms of anthropogenic habitat fragmentation. Semi-aquatic reptiles are vulnerable to urbanisation because they often use terrestrial habitats in addition to their wetlands thus exposing themselves to urban hazards such as traffic. This study examined the impact of a moderate level of urbanisation on the eastern long-necked turtle (*Chelodina longicollis*) in south-eastern Australia, a species which exhibits a complex use of both aquatic and terrestrial habitats. I used radio-telemetry to compare the behaviour and survivorship of *C. longicollis* in an urban site (high road density and traffic volume) with a neighbouring nature reserve. Mark-recapture methods were used to assess sex ratios, size frequency distributions, relative abundance, growth rates, survivorship, movements and injury incidence of *C. longicollis* in wetlands surrounded by varying land-uses and road densities.

Turtles in the urban site moved greater distances (4.3 ± 0.7 km) than turtles in the nature reserve (1.9 ± 0.2 km) and some estimates of home range sizes were also larger for the urban turtles. Patterns of wetland use did not differ between sites for the turtles, with individuals at both sites associating with several different wetlands. Males used larger home ranges, moved greater distances and used more wetlands than females at both sites, though frequency of movements between wetlands did not differ between sexes. At least four turtles moved from the nature reserve into the urban area, however there was no indication that movements occurred in the opposite direction.

Terrestrial aestivation behaviour was substantially modified in the urban area, with turtles from the nature reserve spending an average of 108.3 ± 33.5 days in terrestrial habitat, and urban turtles spending no time in terrestrial habitat except when moving between wetlands. More females (72.7%) spent time in terrestrial habitat than males (28.6%), and for longer periods (females 171.0 ± 36.0 days; males 3.4 ± 2.7 days).

Relative abundance and several other measures of population structure did not differ between the urban and reserve sites, nor were they affected by potentially inimical urban landscape modifications such as road density in the surrounding landscape. The proportion of surrounding grassland and water pH were the only variables that could

explain some of the variation in abundance of turtles among wetlands. Some of the variation in proportion of adults among wetlands was explained by water depth, proportion of surrounding grassland and urban green space (e.g. golf courses), but sex ratios were not influenced by any wetland or habitat variable. Adult turtles in the urban site were larger and grew faster than turtles in the nature reserve. Size frequency distributions of juveniles were similar for the two sites. Survivorship estimates and incidence of injuries did not differ significantly between the urban area and the nature reserve.

This study demonstrated that *C. longicollis* was not detrimentally affected by a moderate level of urbanisation at this site. A high level of inter-wetland connectivity via aquatic drainage lines and under-road culverts in the urban area probably allowed this mobile species to avoid traffic and other urban hazards. Wetland management strategies that do not focus on maintaining linkages between wetlands would be inappropriate for conserving this species in an urban environment. Where such linkages are present, *C. longicollis* may be benefited by a moderate level of urbanisation owing to increased primary productivity and deep, permanent wetlands often present in urban landscapes. This study highlights the need to consider species-specific responses and landscape characteristics when determining the impact of landscape modification on semi-aquatic reptiles.

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Chapter 1

General Introduction

1.1. Anthropogenic Habitat Fragmentation

Habitat fragmentation is one of the most threatening processes driving species extinction (Meffe and Carol 1994). Human activities such as urbanisation, agriculture, and forestry are among the most significant causes of habitat fragmentation (Adams et al. 2006), and often lead to a complete restructuring of vegetation and a changed species composition (Shochat et al. 2006). Habitat fragmentation affects wildlife through complex interactions between deterministic and stochastic processes (Adams et al. 2006), and can influence species behaviour, reproduction, dispersal capabilities, and viability. Although some species benefit from fragmentation processes, most species experience local or regional declines that can sometimes result in their extinction (Lindenmayer and Fischer 2006).

Habitat loss and habitat isolation are two fragmentation processes that profoundly influence species persistence in fragmented landscapes (Lindenmayer and Fischer 2006). Habitat loss typically results in smaller patches of habitat, which can constrain wildlife population size because food, shelter and breeding sites become limited (MacNally and Bennett 1997). Declining populations in these patches become increasingly susceptible to stochastic processes that may accelerate population decline resulting in extinction (Gilpin and Soule 1986). Habitat isolation results in an increase in the average distance between habitat patches, a decrease in connectivity between habitat patches, and an increase in the edge:size ratio of each habitat patch (Lindenmayer and Burgman 2005).

A large body of theory exists on effects of habitat isolation (Hanski 1994; Hanski and Gilpin 1997), but these effects are often species or landscape-specific, and sometimes require a case-by-case examination (Harrison et al. 1988; Thomas et al. 1992). Habitat isolation can be detrimental to a population if movement between patches is severely limited, but as long as some inter-patch movement continues, some of the impacts can be mitigated. A metapopulation is a group of populations that occur on patches of suitable habitat and are separated from one another by a matrix of unsuitable habitat, but are connected by inter-patch dispersal (Hanski and Gilpin 1991, Thrall et al. 2001). As

long as wildlife movements between habitat patches continue, within-patch extinctions will be less likely to occur (Opdam 1993). The ability of a species to move between patches is influenced by the distance between suitable patches, the resistance of the intervening landscape (Bennett 2003), and also on species-specific characteristics such as mobility, mode of travel and behaviour (Lindenmayer and Burgman 2006).

1.1.1. Urbanisation

Urbanisation is arguably the most destructive form of anthropogenic fragmentation (Jellinek et al. 2004; Garden et al. 2006; Shochat et al. 2006). The increasing human demand for residential, commercial, industrial and recreational space means that landscapes around that world are becoming urbanised at an increasing rate (Germain and Wakeling 2001; Adams et al. 2006; Bradley and Altizer 2006). Urbanisation exposes wildlife to a number of challenges in addition to those presented by fragmentation *per se*, including increased spread of disease (Bradley and Altizer 2006), pollution (Anderson 1965), increased predation rates (Jokimaki and Huhta 2000) and additional threats such as roads and traffic (Ashley and Robinson 1996; Marchland and Litvaitis 2004; Steen and Gibbs 2004). Although biodiversity is usually reduced in urban environments, simplified habitat structures (Shochat et al. 2006), increased resource availability (Marzluff 2001), and altered trophic interactions (Faeth et al. 2005) increase the abundance of certain urban dwelling species, regardless of the degree to which habitats are fragmented (Savard et al. 2000).

One of the most destructive components of urbanised landscapes is the associated network of roads and traffic that can negatively affect wildlife populations through both increasing habitat fragmentation effects and by other processes (Forman et al. 2003). Road construction may lead directly to loss and degradation of wildlife habitat (Forman et al. 2003), and habitat quality on areas adjacent to roads may be reduced as a result of vehicle pollution (Jaeger et al. 2005), traffic noise (Reijnen et al. 1996), or edge effects (Garcia et al. 2007). Roads can impede critical movements of wildlife between habitat patches, by either creating behavioural barriers to movement (Shine et al. 2004; Andrews and Gibbons 2005), or through increasing mortality (Dodd et al. 1989; Haxton 2000; Baldwin et al. 2004). Roads can also affect wildlife populations by facilitating the spread of diseases and aiding the dispersal of exotic species into urban areas (Saunders et al. 1991; Lonsdale and Lane 1994), or by increasing human access to wildlife habitats

(Young 1994). Although most species can persist in the presence of at least some roads, reduced population sizes, increased risk of extinction, and reduced re-colonisation rates have all been demonstrated for species living in the presence of roads (Forman et al. 2003).

Species persistence in urban areas is largely determined by their ability to avoid urban dangers, such as roads (Koenig et al. 2001). Unless highly mobile species are able to modify their behaviour to become less mobile in urban areas, they can be potentially vulnerable to additional mortality due to the increased likelihood that they will encounter traffic and other urban hazards (Bonnet et al. 1999; Carr and Fahrig 2001; Roe et al. 2006). However, species that actively avoid roads will be more impacted by isolation effects as they will be prevented from moving between habitat patches and maintaining metapopulation dynamics (Forman et al. 2003). Determining the impacts that urbanisation will have on wildlife species requires a thorough understanding of the interactions between the life history and behavioural characteristics of a species and properties of the landscape in which it occurs. This information is essential for directing management actions aimed at halting wildlife species decline in urban environments.

1.1.2. *Reptiles*

Reptile species are variable in their ability to cope with habitat loss and fragmentation (Diaz et al. 2000; Kjoos and Litvaitis 2001). Although the limited dispersal ability of many reptiles makes them particularly vulnerable to isolation effects (Diaz et al. 2000; MacNally and Brown 2001), low energy and space requirements allow some reptiles to persist in small patches of habitat (Diaz et al. 2000). A number of studies have suggested that reptiles are able to cope with fragmentation better than any other vertebrate group (Dickman 1987; Burkey 1995), though most recent research suggests that reptiles can be particularly sensitive to habitat fragmentation effects (Sarre 1995; Smith et al. 1996; Boudjemadi et al. 1999; Driscoll 2004).

Reptiles can be sensitive to both habitat loss and habitat isolation. For example abundance, survivorship and recruitment of the Florida Scrub Lizard (*Sceloporus woodi*: Phrynosomatidae) were found to be positively associated with patch size (Hokit and Branch 2003), and the lizard *Psammmodromus algirus* (Lacertidae) was found more readily and at higher abundance in reserve habitat than in remnant patches of habitat

embedded in an agricultural matrix (Diaz et al. 2000). The isolation of a population of the Red-footed Amazonian Tortoise (*Geochelone carbonaria*: Testudinidae) resulted in an altered age structure and population density, as well as reduced body growth rates compared with the original non-isolated population (Aponte et al. 2003).

Reptile responses to urbanisation are among the least understood of all vertebrate classes (Germaine and Wakeling 2000). Recent studies have shown that reptiles respond to urbanisation in a variety of ways, largely depending on their life history characteristics and properties of the landscape. For example, blue-tongued lizards (*Tiliqua scincoides*: Scincidae) show a number of ecological characteristics that allow them to persist in urban areas, including strong site fidelity (remaining in 'safe' locations), road avoidance behaviour, and ready use of 'artificial' shelter sites and prey species found in most gardens (Koenig et al. 2001). Fast growth rates and production of large litters also benefit this species in urban environments. The abundance and species richness of lizards in Arizona, USA, peaked in sites that were of medium levels of urbanisation (Germaine and Wakeling 2001). This was attributed to increased primary productivity as a result of watering and fertilising of urban gardens. Beyond a moderate level of urbanisation, lizard species richness and abundance declined rapidly (Germaine and Wakeling 2001).

Reptiles can be particularly vulnerable to the effects of roads because many species expose themselves to traffic by migrating along and across roads to find suitable foraging, breeding, nesting and over-wintering sites (Ashley and Robinson 1996; Burke and Gibbons 1995; Buhlmann and Gibbons 2001; Joyal et al. 2001; Semlitsch and Bodie 2003). Some reptile species even show preference for roadside habitat (Driscoll 2004; Aresco 2005) and may even use roads for thermoregulation (Ashley and Robinson 1996; Rosen and Lowe 1994), again putting themselves at increased risk of mortality from vehicles. In addition, many reptile species are characteristically slow moving and tend to not be as aware of danger presented by vehicles as homeothermic species (Ashley and Robinson 1996; Andrews and Gibbons 2005; Roe et al. 2006).

Despite the vulnerability of reptiles to fragmentation and loss of habitat (Driscoll 2004), habitat fragmentation literature is extremely taxonomically biased towards birds and mammals (Kjoss and Litvaitis 2001; MacNally and Brown 2001). A review of 134 fragmentation papers published in the journals *Conservation Biology*, *Landscape*

Ecology, and *Ecological Applications*, found that reptiles and amphibians only contributed to 4% of studies (McGarigal and Cushman 2002). Population responses of reptiles to anthropogenic habitat fragmentation and urbanisation are therefore generally poorly understood (Germaine and Wakeling 2001). Given the rapid degree to which landscapes are being modified, more effort is needed to investigate the impacts of fragmentation on reptile behaviour, population dynamics, persistence, and viability. This information is essential for directing management actions aimed at mitigating reptile species decline.

1.2. Background to the Aims

A major challenge for managers of wildlife in urban landscapes and adjacent areas is to minimise threats to wildlife while not interrupting the natural processes of movement, habitat use, and behaviour important for maintaining viable populations. Wetland fauna are frequently ignored in wildlife management decisions because they are inconspicuous and are often believed to rarely venture away from aquatic habitat, yet many wetland species such as semi-aquatic reptiles rely on both aquatic and terrestrial habitat to fulfil their life history requirements. Existing wetland management strategies can be inadequate for protecting such species in urban environments because they often focus on managing individual wetlands but not on maintaining terrestrial linkages between the wetlands (Amezaga et al. 2002). Consequently, semi-aquatic reptiles are often exposed to the threats presented by road networks and other elements of urban landscapes.

Freshwater turtles are one group of semi-aquatic reptile that are at risk of encountering roads and other urban dangers when they use terrestrial habitats for nesting, aestivation and foraging (Burke and Gibbons 1995; Ashley and Robinson 1996; Buhlmann and Gibbons 2001; Joyal et al. 2001; Semlitsch and Bodie 2003). Several species are also vulnerable to urban threats when moving overland between wetlands for dispersal, escape from drying wetlands, or seasonal migrations (Roe and Georges 2007). Life-history characteristics of freshwater turtles such as low annual recruitment, high adult survival rates, delayed sexual maturity, and slow population growth rates (Congdon et al. 1993; 1994) make them particularly vulnerable to additional mortality as populations have difficulty recovering from the additional loss of adults (Baldwin et al. 2004; Gibbs and Shriver 2002; Steen and Gibbs 2004; Aresco 2005). Freshwater turtle species are therefore highly vulnerable to urbanisation effects.

Most studies have reported that urbanisation has a negative impact on freshwater turtles (Dodd et al. 1989; Mitchell 1988; Haxton 2000; Hoff and Marlow 2002; Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Conner et al. 2005; Budischak et al. 2006; Aresco 2005; Gibbs and Steen 2005; Steen et al. 2006), although a limited number of species over a small geographic area (i.e., eastern North America) have been examined despite the large number of studies. Moreover, few studies have investigated turtle behaviour in response to urban threats, such as roads, and while some studies have investigated population-level effects of urbanisation, few have studied turtle populations in landscapes with varying levels of urbanisation. Further investigations are required to determine interactions between landscape properties and the individual and population level responses of freshwater turtles to urbanisation. This information is critical for identifying appropriate management actions for freshwater turtles in urban environments.

1.2.1. *The Eastern Long-Necked Turtle (Chelodina longicollis)*

The eastern long-necked turtle, *Chelodina longicollis* (Shaw 1794), is a common Australian species of freshwater turtle in the family Chelidae. Its range extends from the Adelaide region through the Murray-Darling drainage system of Victoria and New South Wales, and as far north as Charters Towers in Queensland (Cann 1998). Within this range the species inhabits a wide range of water bodies including permanent freshwater lakes and rivers, ephemeral ponds, swamps, wetlands and billabongs (Chessman 1988). *C. longicollis* can be distinguished from other *Chelodina* species by black margins on the plastral seams (Ehmann 1992; Swan 1995), and an extension of the anterior lobe of the plastron to the midline of the carapacial marginal scutes (Cann 1998). This species grows to a maximum carapace length of 275 mm (Parmenter 1985; Ernst and Barbour 1989; Ehmann 1992; Cogger 2000) but I extend this maximum to 279 mm (see Chapter 2; Table 7). It is sexually dimorphic, with females being larger than males. Males mature at 6-7 years (Parmenter 1976) at a size of 145 mm (Kennett and Georges 1990), and females mature later at 9-11 years (Parmenter 1976), at a size of 165 mm (Kennett and Georges 1990). *C. longicollis* is an opportunistic carnivore that uses its long neck in a strike and gape action (Legler and Georges 1993), and obtains its food from a wide variety of sources, including plankton, nekton, benthic macro-organisms, carrion and terrestrial organisms that fall upon the water (Georges et al.

1986). It employs a sit-and-wait ambush strategy as well as actively searching for prey (Parmenter 1976). Nesting occurs between October and November (Kennett and Georges 1990), where females deposit between 13 and 24 eggs into a shallow nest chamber (Vestjens 1969). Females usually lay a single clutch, but have the potential to lay up to three clutches in a season (Parmenter 1976; Chessman 1978; Kennett and Georges 1990).

Although much is known about the behaviour and ecology of *C. longicollis* in relatively natural settings, very little is known about the impacts of urbanisation on this species. *C. longicollis* regularly uses terrestrial habitats for aestivation, inter-wetland movements and nesting, and often undertakes extensive over-land migrations (Roe and Georges 2007) particularly following rain (Goode 1967; Cann 1978). This species is therefore at risk of encountering urban dangers such as roads and traffic. The ability of *C. longicollis* to undertake long distance migrations can be attributed to a range of physiological adaptations that enable it to survive in hot and dry conditions during terrestrial travel. These include an ability to reduce dehydration by low cutaneous water loss and possibly by large amounts of water carried in the cloacal bursae (Chessman 1984), an ability to maintain a large head-body temperature gradient (Webb and Johnson 1972), and the ability to aestivate terrestrially (Chessman 1983; Roe and Georges 2007 in press). *C. longicollis* shows some navigational capacity with odour detection, solar guidance, and familiarity with visual landmarks (Stott 1987; Graham et al. 1996). Although some studies have found terrestrial movements of *C. longicollis* to be restricted to daylight hours (Graham et al. 1996), others have not (Chessman 1978; Stott 1987). Owing to its complex interaction with wetland and terrestrial habitats, *C. longicollis* provides an ideal opportunity to examine the consequences of terrestrial habitat alteration for a semi-aquatic reptile.

1.3. Aims and Objectives

The aim of this project was to investigate the ecology of the eastern long-necked turtle (*Chelodina longicollis*) both on and off reserve, and to examine the potential for anthropogenic habitat modification to affect its behaviour, survivorship, and population structure. This study addressed the following broad questions:

- 1) Is *C. longicollis* detrimentally affected by a moderate level of urbanisation?

- 2) Are management strategies focussed on managing wetlands as isolated units, and not on maintaining linkages between wetlands, suitable for conserving *C. longicollis* in urban environments?
- 3) Is it important to consider management at a landscape scale when conserving urban populations of this species?

A further aim of this project was to identify management actions to help conserve *C. longicollis* and other semi-aquatic reptile species in urban environments, and to provide wildlife managers and land planners with critical ecological information to aid them with making informed management decisions.

1.4. Structure of this Thesis

Above, I have provided introduction to what is known and what is poorly known of the impact on and responses of reptiles to urbanisation and road infrastructure, to set the context for the aims, objectives and significance of my project. In Chapters 2 and 3, I outline the specific studies undertaken to meet the above aims, and further explore the significance of my project in light of current understanding of anthropogenic habitat modification and reptiles. Chapter 2 is a radio-telemetry study that examines impacts of urbanisation on behaviour and survivorship of *C. longicollis* in an urban area compared with an adjacent nature reserve. Chapter 3 uses mark-recapture methods to examine the effects of urbanisation on *C. longicollis* at a population level, by comparing sex ratios, size frequency distributions, relative abundance, growth rates, survivorship, movements and injury incidence in wetlands surrounded by varying land-uses and road densities. My final chapter, Chapter 4, is a synopsis, which provides integration of the results that are discussed separately in Chapters 2 and 3, and re-examines the issue of urbanisation and reptiles in light of the findings from this study.

Chapter 2

Behaviour and Survivorship of the Eastern Long-necked Turtle (*Chelodina longicollis*) in an Urban Landscape and an Adjacent Nature Reserve

2.1. Introduction

Loss and fragmentation of habitat through anthropogenic landscape modification is a major cause of wildlife extinction worldwide (Soule 1983; Meffe and Carol 1994). Urbanisation is arguably the most damaging, persistent and rapidly expanding form of habitat fragmentation (Adams et al. 2006), often completely restructuring landscapes and leading to dramatically changed species composition (Shochat et al. 2006). Wildlife species are exposed to a number of challenges in urban areas in addition to those presented by fragmentation *per se*, such as noise, toxins and diseases (Adams et al. 2006; Shochat et al. 2006). High resource availability in urban areas can also support high population densities of particular wildlife species (Marzluff 2001), regardless of the degree to which habitats are fragmented (Savard et al. 2000). Increased road networks and higher traffic flows associated with urban areas tend to magnify the effects of habitat fragmentation on wildlife populations living in these areas, as roads not only further degrade habitat and impede critical movements of wildlife travelling between habitats, but can also result in additional mortality of wildlife individuals (Mitchell and Clemens 2000; Forman et al. 2003). The magnitude of the effect that a road will have on a wildlife population largely depends on the road avoidance behaviour of the species and the population sensitivity to road effects (Jaeger et al. 2005).

The current global crisis in extinctions of amphibians and reptiles has been largely attributed to habitat fragmentation (Gibbons et al. 2000; Hokit and Branch 2003) and many reptile species living in urban areas are vulnerable to both fragmentation processes (Driscoll 2004) and road effects (Rosen and Lowe 1994; Ashley and Robinson 1996; Marchland and Litvaitis 2004; Steen and Gibbs 2004). Many reptile species expose themselves to traffic and other threats in the urban environment through terrestrial activities such as nesting, aestivation, hibernation, migration and dispersal (Ashley and Robinson 1996; Burke and Gibbons 1995; Buhlmann and Gibbons 2001; Joyal et al. 2001; Semlitsch and Bodie 2003). Some species show preference for

roadside habitat (Driscoll 2004; Aresco 2005; Garcia 2007), or may even use roads for thermoregulation (Ashley and Robinson 1996; Rosen and Lowe 1994). Reptile species that readily cross roads, or are attracted to roads, are especially vulnerable to road mortality because reptiles are characteristically slow and tend to be unaware of the danger presented by vehicles (Ashley and Robinson 1996; Andrews and Gibbons 2005; Roe et al. 2006). Contrastingly, reptile species that actively avoid roads are vulnerable to habitat isolation effects, as road avoidance may prevent them from carrying out critical terrestrial behaviours or may disrupt metapopulation dynamics.

Along with other species of aquatic and semi-aquatic reptiles, freshwater turtles are one group that are vulnerable to road effects because they often leave the wetland and use terrestrial habitats, increasing their chances of encountering roads and traffic (Aresco 2005; Gibbs and Steen 2005). In addition, freshwater turtles display a number of life-history characteristics such as low annual recruitment, high adult survival rates, delayed sexual maturity and slow population growth rates (Congdon et al. 1993; 1994). Populations may thus have difficulty recovering from additional loss of adults through road mortality (Baldwin et al. 2004; Gibbs and Shriver 2002; Steen and Gibbs 2004; Aresco 2005). Existing management strategies are often inadequate for protecting species such as freshwater turtles living in urban areas, as such strategies tend to focus on managing the individual wetland and not on maintaining linkages between wetlands (Amezaga et al. 2002). The availability and quality of terrestrial habitat in urban environments is therefore often ignored. Freshwater turtles are thus exposed to additional risks of overland travel in urban areas, such as those presented by roads.

Although a number of studies have demonstrated high levels of road mortality for freshwater turtles (Hoff and Marlow 2002; Goodman et al. 1994; Haxton 2000; Steen and Gibbs 2004) and differential road mortality of nesting females (Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Aresco 2005; Gibbs and Steen 2005), there are few studies that have investigated turtle behaviours in urban settings with high road densities. To my knowledge there are also no studies that have used proper controls by simultaneously examining such behaviours in more pristine habitats. Identifying the type of threat that roads pose to wildlife is essential when developing appropriate mitigating actions. This can only be assessed with a thorough knowledge of turtle behaviour in response to roads and other urban hazards.

In this study I use radio-telemetry to compare the behaviour and survivorship of eastern long-necked turtles (*Chelodina longicollis*) in an urban site with a high road density and traffic volume and a neighbouring nature reserve with no roads. I expect that *C. longicollis* would be vulnerable to urban effects because they are known to make regular use of terrestrial habitats (Roe and Georges 2007), thereby putting themselves at risk of encountering road traffic and other hazards. The objective of the study was to determine the type of threat that urbanisation presents to *C. longicollis* and to identify management actions that could mitigate these impacts. Specifically, I addressed the following questions:

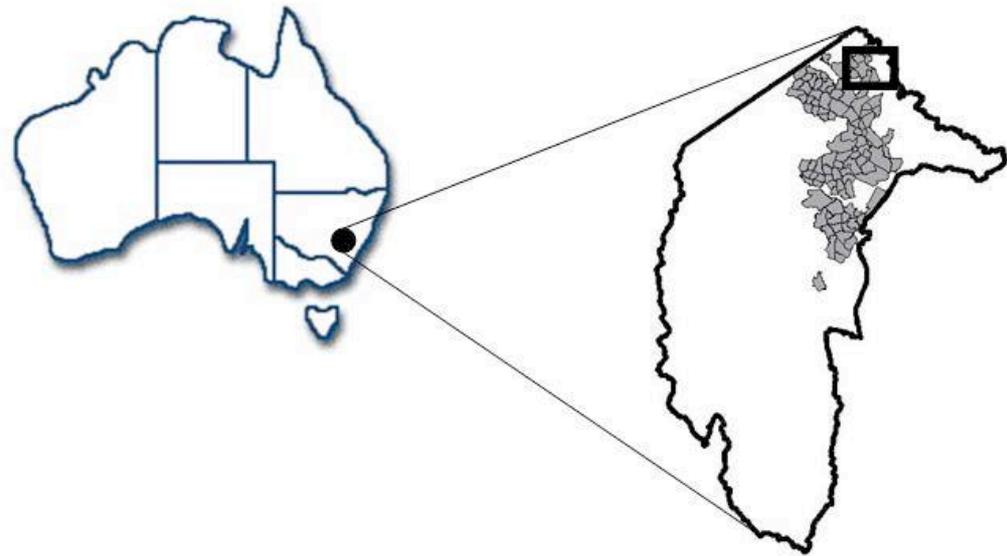
- 1) Does *C. longicollis* become less mobile and less terrestrial when in the presence of roads and other urban hazards?
- 2) Is survivorship of *C. longicollis* lower in the urban site compared with the nature reserve, owing to increased road mortality in the urban site?

This study is the first to compare freshwater turtle behaviour between a reserve and an urban area and will increase knowledge on the potential impacts of urbanisation to reptiles. Information gained from this study may help identify appropriate conservation strategies for wetlands in urban areas and will increase understanding of anthropogenic habitat fragmentation in the context of wildlife conservation.

2.2. Methods

2.2.1. Study Area

I studied eastern long-necked turtles (*Chelodina longicollis*) between September 2006 and November 2007 in Mulligans Flat Nature Reserve and adjacent suburbs in the Gungahlin area of the Australian Capital Territory (ACT), south-eastern Australia (Fig. 1). The Gungahlin area was used for livestock grazing since at least the 1950s (National Capital Development Commission 1988) and was developed as Canberra's fourth satellite town in 1975 (ACT Government 1994). Although the first residents moved into Gungahlin in the early 1990's, development of its outer suburbs was still occurring over the duration of this study. Gungahlin is characterised by large areas of residential housing, high road densities and areas of open space (Fig. 1). There are two large permanent ponds in Gungahlin: Gungahlin pond, constructed in 1989 is located south-



Enlarged section of Northern ACT (Gungahlin)

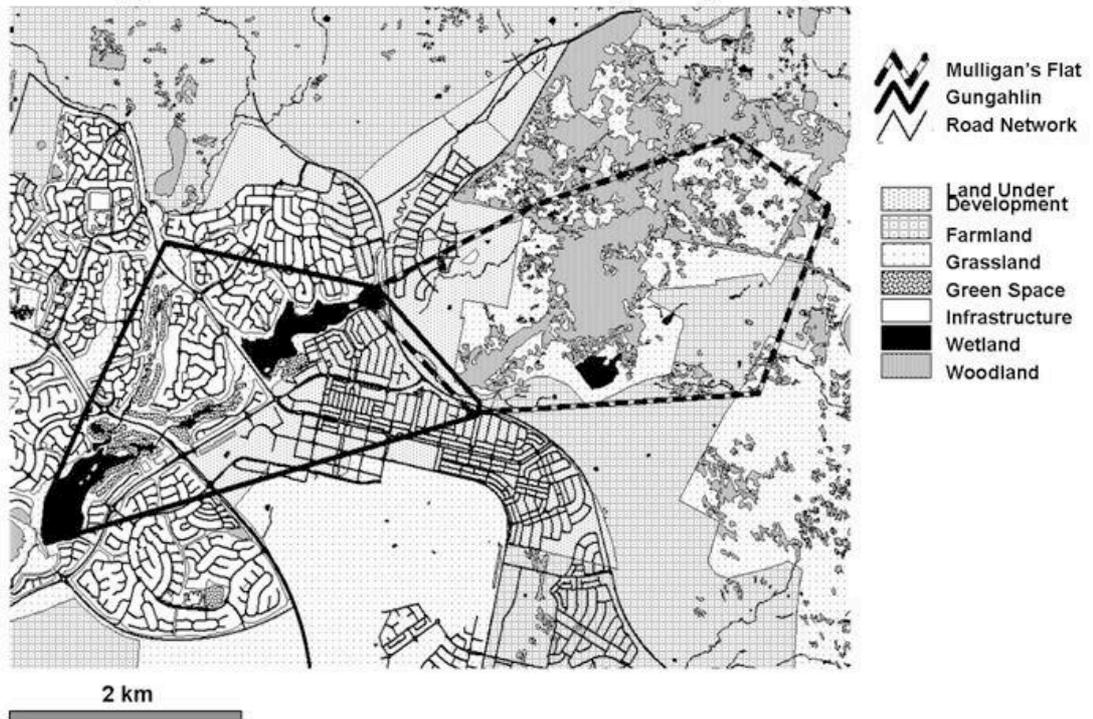


Figure 1. The study site at Gungahlin, Australian Capital Territory, Australia. Bottom picture shows Mulligans Flat Nature Reserve (dashed lines) and Gungahlin (solid line) with their habitat composition and road network. The study sites were delineated by drawing a polygon around the outermost radio-locations of turtles captured from their respective sites.

west of the site; Yerrabi Pond, constructed in 1994 is located north-east of the site (Fig. 1). A number of smaller urban ponds are located within the site, most of which are permanent and constructed as stormwater drainage reservoirs or as golf course ponds. Wetland plant species in Gungahlin are visually dominated by the sedges *Typha orientalis*, *Eleocharis acuta*, *Scirpus validus*, and *Phragmites australis*, and the waterweeds *Potamogeton tricarinatus* and *Vallisneria gigantean*.

Mulligans Flat Nature Reserve was designated a protected area in 1995 (Manning et al. 2007) and now contains the largest and best preserved area of native vegetation in the Gungahlin area (Lepschi 1993). The reserve is bordered in the south-west by expanding urban development and by farmland (primarily sheep and cattle grazing) on the rest of its borders. Habitats on the reserve consist of woodlands and grasslands recently used as rangeland (Lepschi 1993) and a number of small ponds that were originally used for stock (Lepschi 1993). Some ponds are deep and permanent and others are ephemeral (i.e., periodically or sporadically wet). Wetland plant species in Mulligans Flat are visually dominated by the sedges *Typha orientalis* and *Eleocharis acuta* and the waterweeds *Potamogeton tricarinatus* and *Myriophyllum crispatum*.

2.2.2. Habitat Mapping

Habitat boundaries were delineated using a Geographic Information System (ArcView 3.1, Environmental Systems Research 1992) by digitized aerial photographs (ACT Planning and Land Authority 2006). Boundaries were verified by ground-truthing and corrected where appropriate. Habitat was classified as grassland, woodland, farmland (agricultural and grazed land), infrastructure (man-made structures including buildings and car-parks), green space (managed green areas including golf-courses and recreational ovals), land under development and wetland (dams, ponds, lakes and creeks). Because many wetlands have fluctuating water levels, the wetland edge was defined at the high water level mark. The two study sites were delineated by drawing a polygon surrounding the outermost locations of the radio-tracked turtles at the end of the radio-tracking period (see Fig. 1). The area (ha) and proportion of each habitat type and the road density (ACT Planning and Land Authority 2007) was calculated for both sites using the X-tools function in ArcView GIS.

2.2.3. Capture and Radio-Transmitter Attachment

Turtles were captured in wetlands distributed across both study sites either by hand or by using baited crab traps, most of which were fitted with a 'snorkel' that allowed turtles to surface for air. To ensure systematic sampling turtles were only captured from wetlands that were of similar size between the two study sites. A total of 36 adult turtles were fitted with radio-transmitters: 17 turtles (11 F; 6 M) were from Gungahlin and 19 turtles (11 F; 8 M) were from Mulligans Flat (24 Hollohil Systems, Ltd. and 7 Sirtrack Ltd.). Transmitters were mounted on aluminium plates that were secured to the carapace with bolts or plastic cable-ties attached through holes drilled into the posterior marginal scutes. Before attaching transmitters turtle masses were measured on a top-loading balance (+ 0.5 g) and their straight-line carapace length (CL) and midline plastron length (PL) (+ 0.01 mm) was measured using vernier callipers. Sex was determined by examining the plastron curvature (see Kennett and Georges 1990). Size data (mean initial PL and mean mass) of the turtles selected for radio-tracking are presented in Table 1. Transmitters ranged from 1.4 % to 6.4 % of the turtles body mass.

Table 1. Initial plastron length and mass of radio-tracked turtles from Gungahlin and Mulligans Flat Nature Reserve. Values are presented as Mean \pm 1 SE (minimum value – maximum value)

	n	Plastron length (mm)	Mass (g)
Gungahlin			
M	6	152.7 \pm 3.3 (139.6 - 158.9)	729.8 \pm 44.9 (559 – 843)
F	11	171.9 \pm 5.2 (152.8 - 196.3)	1091.5 \pm 109.0 (718 – 1779)
Mulligans Flat			
M	8	141.6 \pm 2.2 (133.2 - 149.9)	585.9 \pm 29.8 (470 – 734)
F	11	172.6 \pm 5.4 (144.6 - 201.7)	1068.6 \pm 87.5 (605 – 1516)

2.2.4. Data Collection

Turtles were located twice per week during the active season (September to March) and once per week in the inactive season (April to August). Although turtles were not located every day, I am confident that the majority of movements were captured as such movements usually occur as isolated events separated by at least a few days (J. Roe pers comm.). Where possible, coordinate positions were determined using a hand-held GPS unit (GPS III Plus, Garmin, Olathe, Kansas, USA; error of 1-7 m) held directly above the turtle, or more commonly by plotting a location obtained via triangulation on an aerial photograph. Location coordinates were then plotted and digitized on habitat maps using ArcView GIS and each location was classified as being in a wetland or being in terrestrial habitat.

At the conclusion of the study turtles were classified as either having survived or having died. Several variables were used to describe movements and use of space for each radio-tracked individual. Both minimum convex polygon (MCP) (Jennrich and Turner 1969) and Kernel density techniques (Worton 1989) were used to estimate the size of total area use, while only the kernel methods were used to define intensively-used 'core' areas (Wray et al. 1992). For kernel density analysis, the fixed kernel method and the least squares cross validation method were used to select a bandwidth for the smoothing parameter h and the 95 and 50% isopleths were used to estimate the size of home range and activity centres, respectively. Linear range length, which is defined as the straight-line distance between the two most widely spaced locations (Plummer et al. 1997) was also determined. The number of wetlands that a turtle visited and the number of times that a turtle moved between wetlands was calculated. Movement distances were estimated as the sum of the minimum straight-line distances between sequential radio locations. Total (cumulative) distance moved by each turtle, and the distance moved in terrestrial habitat (calculated as the minimum straight line distance between two wetlands, or from one wetland to a terrestrial location) was measured. Movements that occurred between two wetlands that were connected by a drainage line that contained water were not included as terrestrial movements. Many turtles were located while making their way along a creek line and I am confident that this is the most common path taken by turtles between wetlands connected along drainage lines. Estimations of

area usage and movement distances were performed with the Animal Movements extension for ArcView GIS.

Terrestrial duration (number of consecutive days a turtle spent in terrestrial habitats without returning to a wetland) and the proportion of radio-tracked time that turtles spent in terrestrial habitat were calculated. The distance between a terrestrial location and its nearest wetland was also measured. Turtles were not considered to be using terrestrial habitat if they were simply moving between two wetlands and I therefore only included data comprising a minimum of two consecutive terrestrial locations recorded. To investigate whether turtles chose terrestrial aestivation sites randomly, several micro-habitat variables were compared between each turtle aestivation site and five randomly selected sites. These random sites were selected using the Random Point Generator function of ArcView GIS, constrained to a 500m radius of the wetland, which is the furthest distance from water that *C. longicollis* is known to aestivate (Roe and Georges 2007). Canopy openness (%) was determined using hemispherical photography and gap light analysis (Frazer et al. 1999; Doody *et al.* 2006). Hemispherical (180°) photographs were taken with a Nikon Coolpix® 995 digital camera with a Nikon FC-E8® fisheye converter lens held directly above the ground pointing upwards. The resultant photographic records were scanned and the digital images analysed using the program Gap Light Analyzer Version 2.0 (Frazer et al. 1999). Average litter depth (mm) was determined by placing a 1m quadrat over each site and averaging the litter depths taken from four different locations within the quadrat (taking care not to disturb the aestivating turtle). Minimum distance to nearest protective structure (m) was the minimum straight line distance to the nearest tree, shrub, log or stump.

2.2.5. Data Analyses

Statistical analyses were performed with SPSS Version 14.0 (SPSS 2006) and SAS Version 8.2 (SAS Institute 2001). The assumptions of normality and homogeneity of variances were examined where appropriate and when data failed to meet these assumptions the data was transformed to approximate normal distributions and equal variances. If the data still deviated from these assumptions non-parametric tests were used. Statistical significance was accepted at the $\alpha = 0.05$ level unless specified otherwise. Means are reported with their standard errors unless otherwise stated.

Fisher's Exact test was used to determine whether survivorship differed between the two sites. Analysis of covariance (ANCOVA) was used to investigate differences in movement and area-use estimates. The response variables for each analysis were MCP, 95% and 50% kernel density estimates and range length. Site and sex were the discrete factors and PL was the covariate. All movement and area use variables (except for range-lengths) were \log_{10} transformed prior to analysis. To investigate differences in total movements between sites and between sexes I also used an ANCOVA with log-transformed total movements as the response variable, site and sex as the main factors and PL as the covariate. Differences in terrestrial movements between sites and sexes were measured using Mann-Whitney U tests. Analysis of variance (ANOVA) was used to investigate differences in wetland use patterns, with the response variables for each analysis being number of wetlands used and frequency of movements between wetlands, and the discrete factors being site and sex. Mann-Whitney U tests were used to investigate differences in time spent in terrestrial habitat between sites and sex. Firstly I combined sexes and tested for a difference between sites and then separated sexes and tested for a difference between sites for each sex. I then separated sites and tested for a difference between sexes at Mulligans Flat (Gungahlin had no turtles that spent time in terrestrial habitat). The series of Mann-Whitney U tests were treated as connected, and level of significance was adjusted using the Dunn-Sidak correction to maintain an experiment-wide error rate of 0.05. The adjusted α level was $\alpha \leq 0.0125$. ANOVA was used to investigate differences in micro-habitat variables between turtle aestivation sites and sites that were randomly located, with canopy cover (%), average litter depth (mm) and distance to nearest structure (m) as the response variables and site and type (turtle or random) as the main effects. Distance to nearest structure (m) was \log_{10} transformed prior to analysis.

2.3. Results

2.3.1. *Habitat Composition*

Gungahlin comprised an area of 539.8 ha, over-lapping slightly with Mulligans Flat along its north-east edge (Fig. 1). Mulligans Flat was a slightly larger site, comprising an area of 615.4 ha. Habitat Composition varied greatly between the two sites (Table 2; Fig. 1) with Gungahlin consisting of higher proportions of urban landscape features

such as infrastructure and Mulligans Flat Nature Reserve containing higher proportions of natural habitat such as grassland and woodland. Average road density was 18.9 times higher in Gungahlin (12.75 km / km²) than in Mulligans Flat (0.64 km / km²).

2.3.2. *Survivorship*

The 36 turtles were radio-tracked for an average of 320.3 ± 19.7 consecutive days, for which I obtained 51 ± 3 locations. Survivorship did not differ between sites (Fisher's Exact Test: $df = 1$; $p = 0.59$), with 82.4 % of turtles surviving in Gungahlin and 89.5 % of turtles surviving in Mulligans Flat. One turtle from each of the two sites died of an unknown cause (however these deaths were both immediately following an unseasonable cold spell) and one turtle was killed by a vehicle in Gungahlin. One turtle from each site died after only 16 and 34 radio-tracked days. These individuals were omitted from all of the further analyses on movements, spatial ecology and habitat use.

2.3.3. Spatial Ecology

There were no significant relationships for PL in any of the ANCOVAs (ANCOVA: $F \leq 0.580$; $df = 1, 29$; $p \geq 0.452$) and therefore it was removed as a covariate and the analyses were repeated as ANOVAs. Radio-tracked turtles in Gungahlin used larger total areas than turtles in Mulligans Flat, but this difference was only significant using the 95% kernel estimate. Males used larger total areas than females at both sites (ANOVA: MCP: $F_{\text{site}} = 2.921$; $df = 1, 30$; $p = 0.098$; $F_{\text{sex}} = 7.532$; $df = 1, 30$; $p \leq 0.01$; $F_{\text{site*sex}} = 0.805$; $df = 1, 30$; $p = 0.377$; 95% kernel density, $F_{\text{site}} = 5.473$; $df = 1, 30$; $p < 0.05$; $F_{\text{sex}} = 15.182$; $df = 1, 30$; $p < 0.01$; Table 3; Fig. 2). Gungahlin turtles used larger core activity centres than Mulligans Flat turtles and males used larger core areas than females (50% kernel density: $F_{\text{site}} = 4.200$; $df = 1, 30$; $p < 0.05$, $F_{\text{sex}} = 12.557$; $df = 1, 30$; $p < 0.01$; $F_{\text{site*sex}} = 0.024$; $df = 1, 30$; $P = 0.879$; Table 3). Linear range lengths did not differ between sites, but were larger for males than females (range length: $F_{\text{site}} = 1.206$; $df = 1, 30$; $p = 0.281$; $F_{\text{sex}} = 7.481$; $df = 1, 30$; $p \leq 0.01$; $F_{\text{site*sex}} = 0.110$; $df = 1, 30$; $p = 0.743$; Table 3). Gungahlin turtles moved greater overall distances than did Mulligans Flat turtles and males travelled significantly larger total distances than did females at both sites (ANOVA: $F_{\text{site}} = 14.736$; $df = 1, 30$; $p \leq 0.01$; $F_{\text{sex}} = 15.788$; $df = 1, 30$; $p < 0.001$; $F_{\text{site*sex}} = 1.051$; $df = 1, 30$; $p = 0.313$; Table 4). Mulligans flat turtles travelled greater distances overland than did Gungahlin turtles (Mann-Whitney U test: $Z = -2.981$; $df = 32$; $p < 0.01$; Table 4). Terrestrial distance travelled was similar between sexes (Mann-Whitney U test: $Z = -0.730$; $df = 32$; $p = 0.465$; Table 4).

Table 2. Habitat composition of Gungahlin and Mulligans Flat Nature Reserve. Values are percentages.

	Grassland	Woodland	Infra-structure	Land Under Development	Farmland	Green Space	Wetland
Gungahlin	21.0	0.6	28.9	34.3	0	4.8	10.4
Mulligans Flat	37.1	28.6	0.2	14.1	17.5	0	2.5

Table 3. Space use patterns for radio-tracked male and female *C. longicollis* in Gungahlin and Mulligans Flat Nature Reserve. Values are presented as means \pm 1 SE (minimum value – maximum value)

	n	Total and Core Area Usage			
		MCP (ha)	Range length (km)	95% Kernel density (ha)	50% Kernel density (ha)
Gungahlin					
M	6	39.6 \pm 8.7 (3.7 – 59.9)	1.7 \pm 0.3 (0.4 – 2.3)	83.1 \pm 22.4 (2.6 – 135.2)	13.6 \pm 4.6 (0.5 – 31.2)
F	10	14.2 \pm 4.7 (0.3 – 37.2)	0.8 \pm 0.2 (0.0 – 1.9)	11.7 \pm 3.2 (0.4 – 27.8)	2.3 \pm 0.6 (0.0 – 5.2)
Combined	16	23.7 \pm 5.3 (0.3 – 59.9)	1.2 \pm 0.2 (0.0 – 2.3)	38.5 \pm 12.1 (0.4 – 135.2)	6.5 \pm 2.2 (0.0 – 31.2)
Mulligans Flat					
M	7	12.3 \pm 3.7 (1.6 – 24.8)	1.2 \pm 0.2 (0.2 – 2.0)	39.1 \pm 19.0 (1.0 – 139.2)	7.5 \pm 3.8 (0.2 – 28.4)
F	11	6.4 \pm 2.7 (0.7 – 31.8)	0.6 \pm 0.2 (0.2 – 1.9)	2.9 \pm 0.6 (0.6 – 6.7)	0.6 \pm 0.1 (0.1 – 1.3)
Combined	18	8.7 \pm 2.2 (0.7 – 31.8)	0.9 \pm 0.1 (0.2 – 2.0)	17.0 \pm 8.2 (0.6 – 139.2)	3.3 \pm 1.6 (0.1 – 28.4)

Table 4. Movement patterns for radio-tracked male and female *C. longicollis* in Gungahlin and Mulligans Flat Nature Reserve. Values are means \pm 1 SE (minimum value – maximum value)

	n	Movement Patterns	
		Total distance moved (km)	Terrestrial distance moved (km)
Gungahlin			
M	6	6.4 \pm 1.1 (2.9 – 9.3)	0.3 \pm 0.2 (0 – 1.2)
F	10	3.0 \pm 0.6 (0.7 – 6.6)	0.4 \pm 0.2 (0 – 2.0)
Combined	16	4.3 \pm 0.7 (0.7 – 9.3)	0.4 \pm 0.1 (0 – 2.0)
Mulligans Flat			
M	7	2.4 \pm 0.2 (1.9 – 3.4)	1.4 \pm 0.3 (0 – 2.5)
F	11	1.5 \pm 0.2 (0.7 – 2.9)	0.9 \pm 0.2 (0 – 1.8)
Combined	18	1.9 \pm 0.2 (0.7 – 3.4)	1.1 \pm 0.2 (0 – 2.5)

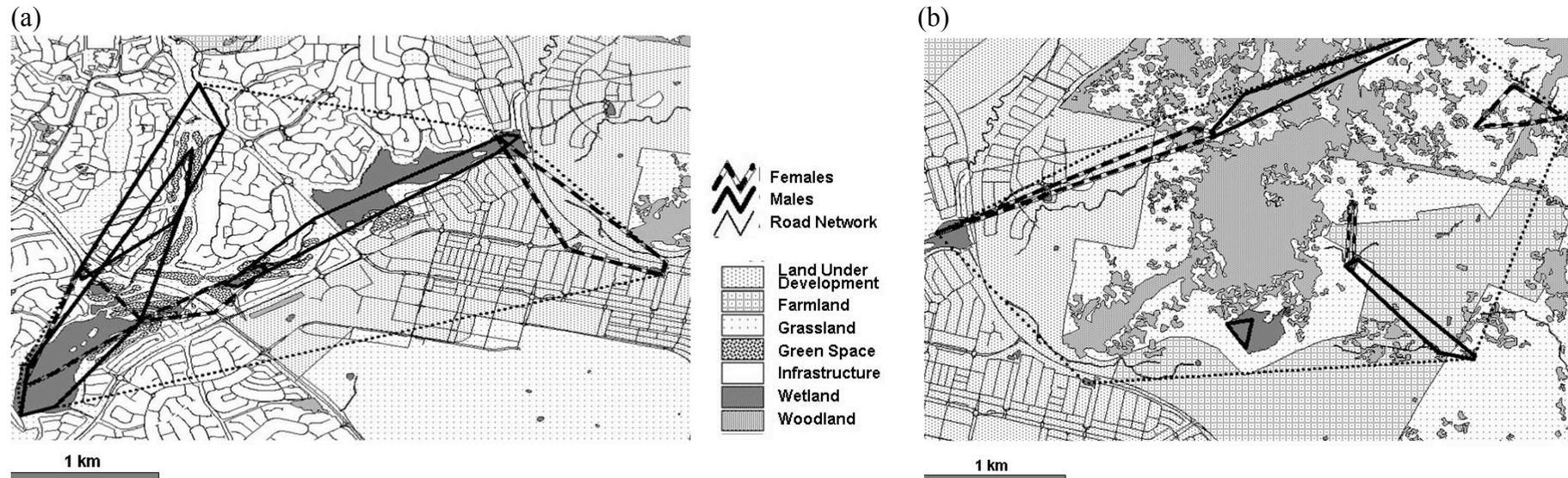


Figure 2. Habitat composition and total area use estimates for *C. longicollis* in Gungahlin and Mulligans Flat Nature Reserve. Home ranges (MCP) are shown for 3 male turtles (solid lines) and 3 female turtles (dashed lines) from each site: (a) Gungahlin and (b) Mulligans Flat Nature Reserve

2.3.4. Wetland Use Patterns

Radio-tracked turtles in both sites used a similar number of wetlands and moved between them a similar number of times (Table 5). Males used a greater number of wetlands than did females at both sites, though the frequency of inter-wetland movements did not differ between sexes (ANOVA: number of wetlands: $F_{\text{site}} = 0.66$; $df = 1, 30$; $p = 0.422$; $F_{\text{sex}} = 4.70$; $df = 1, 30$; $p < 0.05$; $F_{\text{site*sex}} = 0.56$; $df = 1, 30$; $p = 0.461$; frequency of wetland shifts: $F_{\text{site}} = 1.23$; $df = 1, 30$; $p = 0.277$; $F_{\text{sex}} = 1.62$; $df = 1, 30$; $p = 0.213$; $F_{\text{site*sex}} = 0.28$; $df = 1, 30$; $p = 0.602$; Table 5).

2.3.5. Terrestrial Habitat Use

Terrestrial habitat use included periods of brief to more extended refuge out of wetlands associated with wetland drying, but use of terrestrial habitats differed markedly between sites (Mann-Whitney U test: $Z = -3.427$; $df = 32$; $p < 0.01$). Mulligans Flat turtles spent 108.3 ± 33.5 days (range = 0 – 281; $n = 18$) in terrestrial habitat (28 ± 7.6 % of total radio-tracked time) whereas Gungahlin turtles spent no time in terrestrial habitats other than during movement between wetlands. The difference between the sites was attributed primarily to the behaviour of female turtles (Females: Mann-Whitney U test: $Z = -3.223$; $df = 20$; $p < 0.01$; Males: Mann-Whitney U test: $Z = -1.363$; $df = 12$; $p = 0.173$). The majority (72.7 %) of females in Mulligans Flat used terrestrial habitat for 45.1 ± 9.3 % (range = 0 - 77.2 %; $n = 11$) of the total time that they were radio-tracked, remaining terrestrial for 171.0 ± 36.0 (range = 0 - 281) consecutive days without returning to wetlands. In contrast, only 28.6 % of Mulligans Flat males used terrestrial habitat, staying there for 3.4 ± 2.7 (range = 0 – 19, $n = 7$) consecutive days which was only 1 ± 0.8 % (range = 0 - 5.56) of their total radio-tracked time. The difference in behaviour between males and females at Mulligans Flat was only marginally significant after applying the Dunn-Sidak adjustment $\alpha \leq 0.0125$ (Mann-Whitney U test: $Z = -2.369$; $df = 27$; $p < 0.02$).

Table 5. Wetland use patterns for radio-tracked *C. longicollis* in Gungahlin and Mulligans Flat Nature Reserve. Values are presented as Mean \pm 1 SE (Minimum value – Maximum value).

	n	Number of Wetlands Used	Number of Wetland Shifts
Mulligans Flat			
Males	7	2.9 \pm 0.4 (1 – 4)	2.0 \pm 0.4 (0 – 3)
Females	11	2.3 \pm 0.3 (1 – 4)	1.5 \pm 0.4 (0 – 4)
Combined	18	2.5 \pm 0.3 (1 – 4)	1.7 \pm 0.3 (0 – 4)
Gungahlin			
Males	6	3.5 \pm 0.6 (1 – 5)	3.0 \pm 0.8 (0 – 6)
Females	10	2.3 \pm 0.4 (1 – 5)	1.9 \pm 0.7 (0 – 6)
Combined	16	2.8 \pm 0.4 (1 – 5)	2.3 \pm 0.5 (0 – 6)

Microhabitat Analysis of Terrestrial Aestivation Sites

Aestivation sites were located 151.7 ± 44.6 m (range = 37.5 - 394.0; $n = 8$) from the nearest wetland and were in woodland habitat where the turtle buried into the leaf-litter until they were either completely covered or only a small part of the carapace was left exposed. Turtles chose aestivation sites non-randomly with respect to several micro-habitat variables (Table 6). Canopy openness was significantly lower in aestivation sites (38.9 ± 2.1 %, range = 32 - 48 %) than in randomly selected sites (66.8 ± 2.7 %, range = 35.7 - 87.7 %; $n = 40$) (ANOVA: $F_{\text{site}} = 0.501$; $df = 1, 32$; $p = 0.827$; $F_{\text{type}} = 19.318$; $df = 1, 32$; $p < 0.001$; $F_{\text{site*type}} = 0.962$; $df = 1, 32$; $p = 0.962$). Aestivation sites were also chosen with a significantly deeper mean litter depth (10.7 ± 1.3 mm, range = 5.5 - 16.2 mm) than in random locations (1.3 ± 0.3 mm, range = 0 - 5.8). However, litter depth also varied between the eight aestivation sites, with a significant site \times type interaction occurring between the two (ANOVA: $F_{\text{site}} = 3.318$; $df = 1, 32$; $p < 0.01$; $F_{\text{type}} = 158.178$; $df = 1, 32$; $p < 0.001$; $F_{\text{site*type}} = 2.971$; $df = 1, 32$; $p < 0.02$). Site had no significant effect on litter depth between random locations ($F = 0.31$; $df = 7, 32$; $p = 0.945$) whereas a pronounced effect was evident for aestivation sites ($F = 13.85$; $df = 7, 32$; $p < 0.01$). Aestivation sites were closer to protective structures such as logs, trees and stumps than were random sites (ANOVA: structure: $F_{\text{site}} = 0.523$; $df = 1, 32$; $p = 0.811$; $F_{\text{type}} = 7.601$; $df = 1, 32$; $p \leq 0.01$; $F_{\text{site*type}} = 0.228$; $df = 1, 32$; $p = 0.976$). The distance to nearest structure for aestivation sites was 0.9 ± 2.0 m (range = 0.1 - 2.0 m), but for random sites was 9.1 ± 1.8 m (range = 0.4 - 62).

Table 6. Microhabitat analysis of terrestrial aestivation sites for female turtles at Mulligans Flat Nature Reserve and associated five random locations. Type is turtle ID and associated five random locations. Random values are presented as means \pm 1 SE (minimum value – maximum value)

Type	Terrestrial Duration		Microhabitat Variable			
	Days	% of total time radio-tracked	Distance to nearest wetland (m)	% Canopy Openness	Mean litter depth (mm)	Distance to nearest structure (m)
1	281	67.6	394.0	40.9	8.8	1.1
Random 1			302.4 \pm 49.5 (220.0 – 492.0)	69.4 \pm 9.2 (41.7 – 86.3)	1.0 \pm 0.6 (0.0 – 3.1)	7.7 \pm 2.9 (0.7 – 14.9)
2	207	60.5	167.0	47.9	6.6	0.8
Random 2			344.0 \pm 64.6 (120.5 – 478.0)	66.4 \pm 7.0 (43.8 – 79.8)	2.1 \pm 1.3 (0.0 – 5.8)	6.2 \pm 2.2 (0.5 – 13.9)
3	280	67.3	54.0	40.3	14.3	0.8
Random 3			390.0 \pm 29.5 (296.0 – 457.0)	60.9 \pm 5.3 (50.5 – 80.3)	1.8 \pm 0.8 (0 – 4.4)	5.2 \pm 2.6 (0.8 – 15.2)
4	144	34.6	48.5	45.2	5.5	0.7
Random 4			345.8 \pm 56.9 (138.0 – 481.0)	74.1 \pm 7.2 (49.5 – 87.2)	0.9 \pm 0.5 (0.0 – 2.5)	10.3 \pm 1.7 (5.1 – 15.7)
5	264	77.2	179.0	39.6	11.8	0.1
Random 5			353.3 \pm 16.9 (308.0 – 406.0)	78.0 \pm 7.1 (49.8 – 87.3)	1.2 \pm 1.2 (0 – 5.8)	18.4 \pm 3.3 (1.7 – 18.2)
6	274	68.5	262.0	32.0	13.0	0.3
Random 6			225.3 \pm 66.8 (66.0 – 420.0)	76.2 \pm 8.1 (44.3 – 87.7)	0.8 \pm 0.4 (0.0 – 1.8)	15.1 \pm 5.5 (1.0 – 29.3)
7	212	57.6	37.0	32.8	16.2	1.1
Random 7			373.6 \pm 32.5 (259.0 – 457.0)	52.0 \pm 6.2 (35.7 – 70.9)	1.8 \pm 0.9 (0.0 – 5.1)	4.4 \pm 3.0 (0.4 – 16.4)
8	263	63.22	71.5	32.2	9.8	2.0
Random 8			297.5 \pm 66.1 (80.0 – 435.0)	57.7 \pm 8.0 (42.6 – 84.3)	1.3 \pm 0.9 (0.0 – 4.8)	15.2 \pm 11.8 (0.5 – 62)

2.4. Discussion

This study demonstrates that movement distances and size of area used are larger for *C. longicollis* in an urban area compared with an adjacent nature reserve. Thus, it appears that *C. longicollis* does not change its behaviour to become less mobile to avoid urban dangers, but surprisingly becomes even more mobile in an urban landscape. There is debate over whether minimum convex polygon (MCP) or kernel density techniques are more appropriate for describing use of space (Row and Blouin-Demers 2006, Nilsen et al. 2007; in press), so the fact that some estimates detected significant differences between sites while others did not leads me to interpret the area use result with some caution. Terrestrial aestivation was also found to differ dramatically between the two sites, demonstrating that urbanisation can also influence aestivation behaviour for this species. Surprisingly, mortality rates were not higher in the urban site than in the nature reserve as predicted, despite the fact that *C. longicollis* was highly mobile in the urban landscape. This study indicates that the most likely threat faced by *C. longicollis* in Gungahlin is direct road mortality rather than isolation effects, though road mortality rates were not especially high. I suggest that the structure of the Gungahlin landscape facilitates turtle movements between wetlands, so that urban dangers such as roads can be avoided. In urban landscapes where such features are not present, *C. longicollis* may be heavily impacted by road mortality and its ability to persist in urban environments would be threatened. This study highlights the importance of maintaining habitat linkages between wetlands in urban environments, so that freshwater turtles and other aquatic and semi-aquatic fauna will be protected from roads and other urban hazards during inter-wetland movements.

2.4.1. Spatial Ecology and Survival

Although few studies have investigated freshwater turtle behaviour in urban areas, movement extent and direction can be restricted in urban environments for some species. For example, Blanding's turtles (*Emydoidea blandingii*) avoid moving into developed areas in suburban Chicago, USA (Rubin et al. 2001), resulting in isolated populations restricted to forest preserves. Similarly eastern box turtles (*Terrapene carolina carolina*) in agricultural and suburban sites moved less compared to individuals in continuous forest (Iglay et al. 2007). For such species, urbanisation is likely to lead to habitat isolation that may negatively impact populations through

inbreeding and other processes (Fahrig and Merriam 1994). However, there are also many studies that demonstrate high levels of road mortality in turtles, suggesting that individuals of many species are not altering their behaviour to avoid roads (Hoff and Marlow 2002; Goodman et al. 1994; Haxton 2000; Steen and Gibbs 2004). Despite the high road densities and other inimical habitats that characterized the urban landscape in this study, *C. longicollis* did not become less mobile in an urban area. In fact, movement distances and home range sizes were more extensive in urban habitats than in the nature reserve (Table 3, 4; Fig. 2). This is the first study to simultaneously compare the behaviours of turtles in the urban landscape to a control group in a nearby natural area. I found that the urban landscape is not likely to have an isolating effect on the turtles, but rather that direct mortality while crossing roads is likely to be the more formidable threat to *C. longicollis*.

A number of studies have demonstrated highly mobile species to be vulnerable to additional road mortality because of the increased risk of encountering roads and traffic (Bonnet et al. 1999; Carr and Fahrig 2001; Roe et al. 2006). For example simulated movements of the northern water snake (*Nerodia sipedon*: Colubridae) and the copperbelly water snake (*Nerodia erythrogaster neglecta*) on maps of wetlands and road networks suggested that roads may be more of a significant source of mortality for *N. e. neglecta*, the species with higher mobility (Roe et al. 2006). Similarly, traffic volume was found to have a significant negative effect on population density of highly mobile leopard frogs (*Rana pipiens*), whereas densities of the less mobile green frogs (*Rana clamitans*) were not affected (Carr and Fahrig 2001). In contrast with such studies, mortality rates of *C. longicollis* were surprisingly low in the urban site given their extensive movements and presumed exposure to threats. In fact, survivorship of urban turtles was similar to those in the nature reserve and both were within the expectations for *C. longicollis* (Shine and Iverson 1995; Roe 2008) and other freshwater turtles (Frazer et al. 1991; Congdon et al. 1993; Shine and Iverson 1995; Spencer and Thompson 2005).

The continued ability of turtles to move about in the urban landscape, without the expected mortality consequences may be attributable to the structure of the Gungahlin landscape (Fig. 3). Gungahlin is characterised by two large permanent ponds and a number of artificial wetlands, many of which are connected along natural or artificial drainage lines that often remain flooded. The large home range sizes, core activity areas

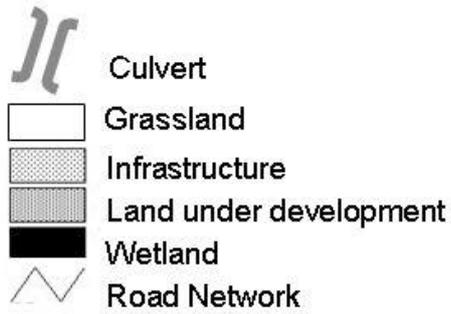
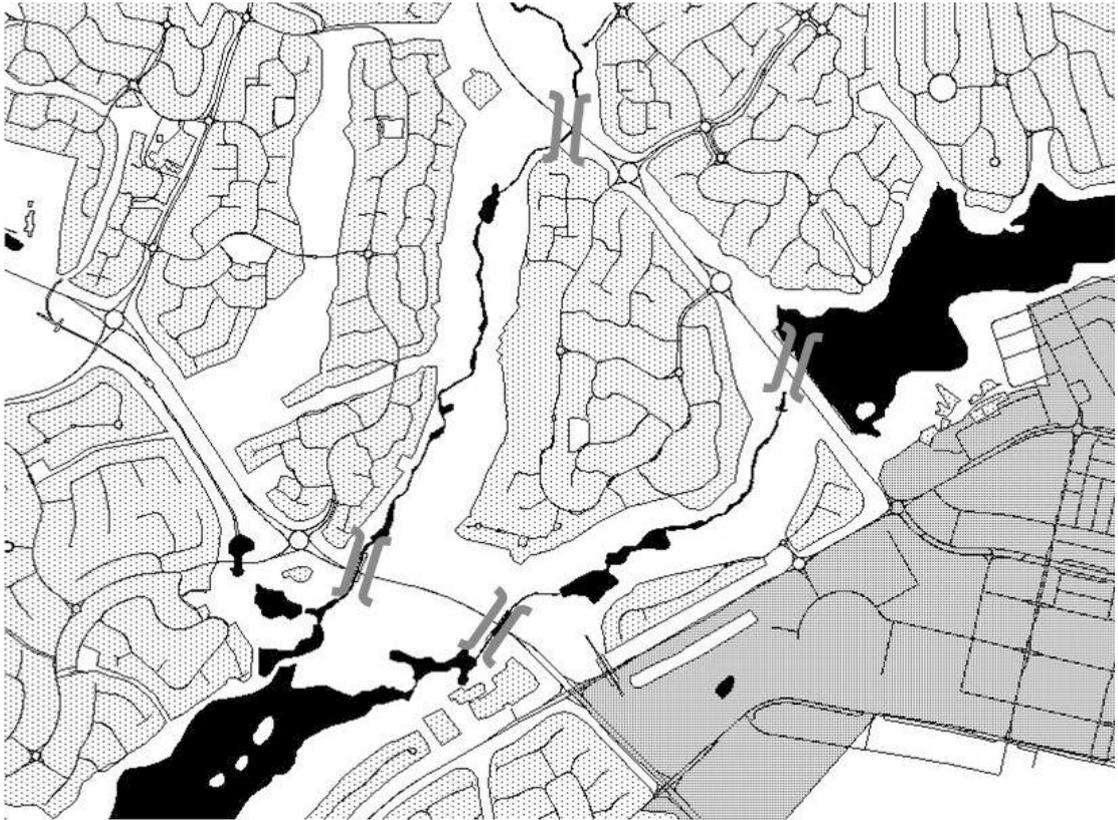


Figure 3. Structure of the urban landscape in Gungahlin, showing large wetlands, stormwater drainage lines and culverts.

and long movements of turtles in Gungahlin may in part be attributable to the large wetlands found on this site (Fig. 1), as wetland size influences movement distance and spatial ecology in this species (Roe and Georges in press). Movements within Gungahlin are also facilitated along the drainage lines, which in many cases allow turtles to travel between wetlands without having to leave aquatic habitat (Fig. 3). Importantly, road culverts along these drainage lines allow turtles to move between wetlands without having to contend with roads and associated traffic (Fig. 3). Radio-locations of turtles moving along these drainage lines suggest that where culverts are present, turtles tend to use them to cross under roads. However, even in areas where culverts are not present, turtles readily crossed roads and maintained associations with several different wetlands. The relative safety and inconspicuousness of turtles travelling along drainage lines and using culverts in the urban landscape most likely accounts for their high survivorship, though it is possible that their activities and movements between wetlands occurred at times of low traffic (i.e., at night or early morning). However, a nearby population was found to be strictly diurnal in their terrestrial movements (Graham et al. 1996), coinciding with the periods of most intense traffic. My observations of several road-killed turtles in the area and the one road killed radio-tagged turtle indicate that culverts are not always used even when available, but they can still substantially reduce the number of turtles that encounter vehicles.

Where structures such as culverts and drainage lines are present, high mobility may confer benefits to *C. longicollis* in urban areas. The ability of species to move between habitat patches is essential to their persistence in fragmented landscapes (Bowne et al. 2006). For example, high dispersal ability in the tree dtella (*Gehyra variagata*) (a habitat generalist) allowed this species to form a metapopulation in which individuals were able to move between woodland patches in an agricultural landscape in Western Australia (Sarre et al. 1995). In the same landscape another species, the reticulated velvet gecko (*Oedura reticulata*) (a habitat specialist) was unable to form a metapopulation because of low dispersal ability, placing it at an increased risk of extinction. In the case of *C. longicollis*, movements between wetlands are an integral aspect of the species ecology not only for metapopulation dynamics, but also to allow individuals to respond to seasonal or more stochastic habitat fluctuations (e.g., water level fluctuations, prey availability (Kennett and Georges 1990, Graham et al. 1996, Roe and Georges 2007).

Differential road mortality of nesting females and the associated altered sex ratios in populations have been demonstrated in freshwater turtles in urban environments (Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Aresco 2005; Gibbs and Steen 2005), but this study found no mortality differences between sexes. Biased mortality between sexes is often associated with different movement patterns employed by males and females (*op cit.*). Male and female *C. longicollis* did not differ in the distance they travelled terrestrially or the frequency of movements between wetland, but males had larger home range sizes, moved between a larger number of wetlands and travelled greater overall distances than did females at both sites. Greater movements of males have been reported for other freshwater turtle species (Morreale et al. 1984; Tuberville et al. 1996; Chelazzi et al. 2007) and are usually attributed to increasing mating opportunities. The need for female turtles to travel during nesting excursions is substantially reduced if suitable habitat is located close to the wetland (Baldwin et al. 2004). Nearby nesting locations may explain why terrestrial movements were not female biased in this study, as most wetlands are surrounded by open grassy areas (such as a golf course in Gungahlin) that can be readily used for nesting. The more extensive movements and area use of males in Gungahlin apparently do not increase their risk of mortality.

2.4.2. Terrestrial Aestivation

Turtles in the urban site differed dramatically in their use of terrestrial habitats for aestivation compared to turtles on the nature reserve. *C. longicollis* exhibits variability in their preference to use terrestrial habitat for aestivation based on several natural habitat variables including hydroperiod variation, wetland isolation and perhaps individual energy reserves (Roe and Georges 2007 in press). Here, I demonstrate that variation in aestivation behaviour can be attributable to urban influences as well. Turtles in Gungahlin remained in aquatic habitats for the study duration, whereas those on the reserve aestivated terrestrially for several months. One likely explanation why turtles did not aestivate in the urban site is the lack of suitable burrowing sites. Turtles that aestivated in the nature reserve chose sites with specific micro-habitat and structural conditions, typically sheltered sites located in woodland habitat with dense canopy and deep litter. Aestivation sites were similar to those chosen by other species of freshwater turtle (Morales-Yerdeja and Vogt 1997; Buhlmann and Gibbons 2001). Woodland habitats were rare in Gungahlin, especially when compared to the extent of this habitat

type in Mulligans Flat. Turtles would therefore find it more difficult to find a suitable terrestrial aestivation site in the urban landscape.

C. longicollis commonly seeks terrestrial refuge, but this behaviour is typically exhibited only when wetlands dry (Roe and Georges 2007 in press). The deep permanent wetlands characteristic of Gungahlin may have provided little reason for turtles to aestivate terrestrially in the urban site. Although only one out of the nine Mulligans Flat wetlands dried completely, only the deepest wetlands were selected for trapping. Most other wetlands on Mulligans Flat dried at least once over the study duration and five out of the nine trapped wetlands reached depths of <1 m. When wetlands become this shallow, a mass exodus of freshwater turtle species often ensues because of the deteriorating conditions, high water temperatures and crowding (Buhlmann and Gibbons 2001, Roe and Georges 2007 in press). In comparison, one of the nine trapped wetlands dried in Gungahlin and only two reached depths of <1 m. The majority of wetlands that were not trapped on this site remained flooded. The increased permanence of wetlands in the urban site is may be attributable to increased stormwater runoff and supplementary watering from golf courses. Turtles are therefore more likely to leave the temporary wetlands of Mulligan's Flat to seek terrestrial refuge than they are in Gungahlin. Regardless of whether aestivation behavioural variation was owing to differences in the suitability of terrestrial habitats or the relative stability (i.e., permanence) of the wetland habitats, the differences in aestivation between sites can be attributed to the influence of urbanisation.

The tendency for Mulligans Flat females to aestivate terrestrially while males moved to other wetlands upon wetland drying is a striking result that has not previously been demonstrated for *C. longicollis* (Roe and Georges 2007 in press). Although two males spent time in terrestrial habitat, the time periods were brief and were either immediately prior to a long distance movement away from a wetland that had dried, or immediately after moving to a wetland that had not yet refilled. The difference in terrestrial habitat use between sexes was not owing to body size, as there was considerable overlap in size between aestivating female turtles and radio-tracked males, nor was turtle size a significant predictor of terrestrial aestivation of *C. longicollis* in a previous study (Roe and Georges 2007 in press). The causes of sex-specific variation in aestivation behaviour in *C. longicollis* and other species of freshwater turtle (Buhlmann and Gibbons 2001) require further investigation.

2.4.3. Management Implications

Many wildlife species have been severely impacted by anthropogenic landscape change as a result of extensive loss and fragmentation of their habitat (Meffe and Carol 1994; Lindenmayer and Fischer 2006). Species persistence within fragmented landscapes is in part determined by their ability to move between patches of suitable habitat (Bowne et al. 2006). However, in urban landscapes characterised by high road densities and other urban hazards, species persistence can be determined by their ability to change behaviour to avoid such dangers. This study demonstrates that some aspects of *C. longicollis* behaviour are influenced by urbanisation. *C. longicollis* may limit terrestrial exposure in unsuitable urban sites by forgoing aestivation, but their motivation to aestivate may also account for differences in terrestrial behaviours. However, *C. longicollis* did not reduce movements and home range size in an effort to minimise risks of these travels.

I suggest that elements of the Gungahlin landscape facilitated turtle movements between wetlands, so that they became more mobile while avoided roads. In urban landscapes where features such as culverts and drainage lines are not present I believe that *C. longicollis* would be heavily impacted by road mortality. In the context of managing freshwater turtles and other semi-aquatic reptiles in urban environments, our study highlights the importance of maintaining structures such as drainage lines and culverts that allow safe passage through otherwise inimical habitat and under roads. Urban design can potentially combine infrastructure requirements, such as storm water drainage, with wildlife conservation. A number of studies have shown freshwater turtles and other wildlife species to readily use culverts to cross roads (Yanes et al. 1995; Ruben et al. 2001; Aresco 2003) and other studies have shown that culverts combined with barrier fences are even more successful at reducing road mortality rates of wildlife (Yanes et al. 1995; Aresco 2003; Dodd et al. 2004). I suggest that culverts could be combined with barrier fences (to flank either side of culvert openings) so that freshwater turtles and other semi-aquatic wetland species are prevented from straying up on to roads. The presence of wildlife crossing signs during active seasons, or reduced speed limits in areas where wildlife are known to cross roads, could also help prevent road mortality of wildlife living in urban environments.

This study suggests that wetland management strategies focussed on managing wetlands as individual units would not protect *C. longicollis* living in urban environments, unless habitat structures are already in place to facilitate its movement between wetlands. Many other studies have demonstrated the need for wetland management strategies to focus on maintaining linkages between wetlands (Joyal et al. 2001; Amezaga et al. 2002; Roe et al. 2003; Roe et al. 2006) and between wetland complexes (Roe and Georges 2007), to allow semi-aquatic wetland species to safely travel between wetlands. To further identify management actions suitable for protecting *C. longicollis* and other species of semi-aquatic reptiles in urban environments, demographic responses of species to road-mortality need to be demonstrated. Although I provide some evidence that mortality rates may be low for a subset of adult *C. longicollis* in Gungahlin, individuals are likely to react differently to the same landscape based on costs and benefits specific to age and sex (Bowne et al. 2006). In addition, even low rates of mortality may affect population structures of turtles, which may have implications for turtle recruitment in urban areas. Further investigations are therefore required to determine population-level responses of *C. longicollis* to urbanisation.

Chapter 3

Demography of Freshwater Turtles in an Urbanised Landscape in South-eastern Australia

3.1. Introduction

Urbanisation is a major cause of local species extinction (Shochat et al. 2006) that is occurring at an accelerating rate (Germain and Wakeling 2001; Adams et al. 2006; Bradley and Altizer 2006). Urbanisation can affect species through loss and fragmentation of habitat (Wilcox and Murphy 1985), increased spread of disease (Bradley and Altizer 2006), pollution (Anderson 1965), predation (Jokimaki and Huhta 2000) and additional threats such as road traffic (Ashley and Robinson 1996; Marchland and Litvaitis 2004; Steen and Gibbs 2004). Although biodiversity is often reduced in urban areas, increased resource availability (Marzluff 2001) and altered trophic interactions (Faeth et al. 2005) can increase the abundance of certain urban dwelling species (Emlen 1974; Bradley and Altizer 2006). How a species responds to urbanisation depends both on their ability to use urban habitat (Adams et al. 2006) and avoid urban dangers (Koenig et al. 2001).

Response of reptile populations to urbanisation is less well understood than for other vertebrate taxa (Germaine and Wakeling 2000) despite the fact that many reptile species are declining as a result of anthropogenic landscape change (Gibbons et al. 2000). Semi-aquatic reptiles, such as freshwater turtles, are at risk of encountering urban dangers when they use terrestrial habitats for nesting, aestivation and foraging (Burke and Gibbons 1995; Ashley and Robinson 1996; Buhlmann and Gibbons 2001; Joyal et al. 2001; Semlitsch and Bodie 2003). Several species are also vulnerable to urban threats when moving overland between wetlands for dispersal, escape from drying wetlands, reproduction or seasonal migrations (Roe and Georges 2007). Such a reliance on both aquatic and terrestrial habitats makes freshwater turtle populations particularly vulnerable to urbanisation as they must contend with impacts both within their local wetland as well as the surrounding landscape (Marchand and Litvaitis 2004).

Road networks and high traffic flows characteristic of urbanised landscapes are one urban hazard that impacts turtle populations. Roads not only further degrade habitat and

impede critical terrestrial movements of turtles, but can also result in additional road mortality (Mitchell and Clemens 2000; Forman et al. 2003). Freshwater turtles are vulnerable to the effects of road traffic because they are relatively slow moving animals that readily use roadside habitat (Ashley and Robinson 1996; Aresco 2005; Roe et al. 2006). Freshwater turtles additionally display a number of life-history characteristics such as long lifespan, slow growth, late maturity and high egg and hatchling mortality that make it difficult for populations to recover from additional sources of mortality (Congdon et al. 1993; 1994).

A number of studies have demonstrated high mortality rates for freshwater turtles living in urban areas (Dodd et al. 1989; Mitchell 1988; Haxton 2000; Hoff and Marlow 2002; Conner et al. 2005; Budischak et al. 2006) and others have documented population level consequences for turtles in urban environments (Gibbs and Shriver 2002; Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Aresco 2005; Gibbs and Steen 2005). Male-biased turtle populations have been reported in wetlands surrounded by high road densities, resulting from differential road mortality of nesting females (Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Aresco 2005; Gibbs and Steen 2005; Steen et al. 2006). Turtle populations may also experience reduced recruitment in urban areas owing to a lower number of females, increased nest predation and road mortality of hatchlings (Marchland 2002; Aresco 2005). A decreased ability to undertake normal movements in urbanised landscapes can also isolate turtle populations and increase their risk of extinction through inbreeding and other stochastic processes (Rubin et al. 2001).

Despite the additional threats in urban landscapes, some species continue to persist and may even benefit from living in urban environments (Mitchell 1988; Conner et al. 2005). Urban open space, such as that provided by residential lawns, can provide additional nesting habitat for turtles (Joyal et al. 2001; Marchland and Litvaitis 2004; Aresco 2005). The presence of artificial urban lakes and residential ponds can also provide additional aquatic habitat for freshwater turtles (Rubin et al. 2001) and some species benefit from increased primary productivity resulting from eutrophication of urban wetlands (Knight and Gibbons 1968; Graham and Doyle 1977). In addition, freshwater turtles that feed opportunistically, particularly those that are omnivorous, may benefit from food sources in urban areas, such as gardens and dumpsters (Dodd 2001). Determining the demographic response of turtle species to urbanisation is critical

for determining appropriate management strategies for turtle species living in urban environments. Although there have been several studies of turtle populations in urban settings, most have been based in heavily urbanised parts of the eastern USA and have focussed on a limited number of species. In order to thoroughly understand the responses of turtles to urbanisation it is important to study a wide range of species inhabiting environments with differing degrees of urbanisation.

This study is the first to examine the demography of a native Australian turtle found in a moderately urbanised environment. Capture-mark-recapture methods were used to examine abundance and population structure of eastern-long necked turtles, *Chelodina longicollis* living in wetlands located in an urban area and compare them to those in an adjacent nature reserve. Sex ratios, size frequency distributions, relative abundance, growth rates, survivorship, movements and injury incidence were assessed in wetlands surrounded by varying land-uses and road densities. Although *C. longicollis* is a common and well studied turtle, very little is known of the effects of urbanisation on this species. Given the high terrestrial mobility of *C. longicollis* for nesting, aestivation and inter-wetland movements (Roe and Georges 2007) I expected that *C. longicollis* populations would be negatively impacted by urbanisation. Specifically I predicted there would be (1) differential mortality of nesting females in the urban site resulting in male-biased populations; (2) decreased recruitment in the urban site resulting in adult-biased populations; and (3) decreased abundance and higher incidence of injuries the urban area.

3.2. Methods

3.2.1. Study Area

Eastern long-necked turtles (*Chelodina longicollis*) were studied between September 2006 and November 2007 in Mulligans Flat and Goorooyaroo Nature Reserves and adjacent suburbs in the Gungahlin area of the Australian Capital Territory (ACT), south-eastern Australia. The Gungahlin area was used for livestock grazing since at least the 1950s (National Capital Development Commission 1988) and was developed as Canberra's fourth satellite town in 1975 (ACT Government 2004). Although the first residents moved into Gungahlin in the early 1990's, development of its outer suburbs was still occurring over the duration of this study. Most of Gungahlin is characterised

by large areas of residential housing, high road densities and areas of open space. A large part of western Gungahlin is agricultural land used as a field site for the Australian Commonwealth Scientific and Research Organisation (CSIRO). There are two large permanent ponds in Gungahlin: Gungahlin pond constructed in 1989 is located south west of the site; Yerrabi Pond constructed in 1994 is located north-east of the site. A number of smaller urban ponds are located within the site, most of which are permanent and constructed as stormwater drainage reservoirs or as golf course ponds. Wetland plant species in Gungahlin are visually dominated by the sedges *Typha orientalis*, *Eleocharis acuta*, *Scirpus validus* and *Phragmites australis* and the waterweeds *Vallisneria gigantea* and *Potamogeton tricarlinatus*.

Mulligans Flat and Gorooyaroo Nature Reserves together form an area of 1600 ha (Australian Government 2004). Grazing by domestic livestock and firewood collection stopped in Mulligans Flat Nature Reserve when it was designated as a protected area in 1995. Gorooyaroo Nature Reserve was designated in 2004 (Manning et al. 2007). Both reserves are almost completely bordered by farmland (primarily sheep and cattle grazing), however Mulligans Flat is bordered in the south-west by expanding urban development. Habitats on both reserves consist of woodlands and grasslands recently used as rangeland and a number of small ponds that were originally used for stock (Lepschi 1993). Some ponds are deep and permanent and others are ephemeral (i.e., occasionally dry). Wetland plant species in Mulligans Flat and Gorooyaroo are visually dominated by the sedges *Typha orientalis* and *Eleocharis acuta* and the waterweeds *Potamogeton tricarlinatus* and *Myriophyllum crispatum*. Because only one wetland was trapped in Gorooyaroo, and given the close proximity of the two reserves, this study site will hereafter be referred to as Mulligans Flat.

3.2.2. Capture Methods

Turtles were captured from Gungahlin and Mulligans Flat Nature Reserve using crab traps baited with cut meat and sardines. Traps were set for a 48 hour period on four separate occasions; September and November 2006 and January and October 2007. A total of 18 wetlands were sampled initially (nine from each site) and an additional wetland was added on the border of Gungahlin and Mulligans Flat on the second trapping session. On the second trapping occasion a wetland from Mulligans Flat was excluded from the sampling period because it had completely dried. A total of five traps

were set in each wetland, three of which were modified with a snorkel that allowed traps to sit on the bottom in up to 2 m of water while giving turtles the opportunity to surface for air and two unmodified traps were set along the wetland edge in shallow water (< 0.5 m). Traps were checked approximately 24 hours and 48 hours after they were set and captured turtles were removed and released after data had been recorded. If turtles were captured in the one trapping session (i.e., in two consecutive 24 hour periods) they were not counted as a recapture and were immediately released.

Once captured, each turtle was marked by filing a notch into a unique combination of marginal scutes with a hand held saw. The soft, unossified carapaces of young juvenile turtles were notched using a small pair of scissors. The mass of each turtle was measured on a top-loading balance (+ 0.5 g) and their straight-line carapace length (CL) and midline plastron length (PL) (+ 0.01 mm) was measured using vernier calipers. Sex was determined for turtles with a CL > 145 mm by examining the plastron curvature (see Kennett and Georges 1990) and all turtles with a CL < 145 mm were classified as juveniles. Minimum size at maturity in males is 145 mm and in females is 165 mm (Kennett and Georges 1990). Males > 145 mm and females > 165 mm were classified as adults, while females between 145.0 – 164.9 mm were classed as subadults. Additionally missing limbs and other injuries were noted for each turtle.

3.2.3. Data Collection

For each wetland the number of captures and recaptures of turtles were recorded. This data was used to estimate the abundance, proportion mature and sex ratios of sample populations in each wetland. Turtle abundance, population structure (proportion adult and proportion female), survival estimates and size frequency distributions (PL, 5 mm size classes) were compared between the two sites using turtles captured from the original 18 wetlands. For all 19 wetlands, the demographic characteristics (abundance, proportion adults and proportion females) of each sample population (those turtles captured in a single wetland) were assessed and analysed with respect to several wetland and landscape habitat variables. Growth was defined as the incremental change in carapace length (CL) between captures and growth rates were calculated from the following equation:

$$\text{Growth rate (mm/year)} = \frac{\Delta \text{CL}}{\text{Active no. of growing days}} \times \frac{365}{2}$$

Days of growth were considered to be only those days falling within the season of maximum activity (Beginning of September to the end of February). Growth rates, movement rates (proportion of recaptured individuals that had moved between at least two wetlands) and incidence of injuries were compared between the sites using individuals captured from all 19 wetlands, plus individuals that were collected opportunistically within the greater study area.

Landscape Variables

Habitat boundaries were delineated using a Geographic Information System (ArcView 3.1, Environmental Systems Research 1992) using digitized aerial photographs (ACT Planning and Land Authority 2006). Boundaries were verified by field examination and corrected where appropriate. Habitat was classified as grassland, woodland, farmland (agricultural and grazed land), infrastructure (man-made structures including buildings and car-parks), green space (managed green areas including golf-courses and recreational ovals), land under development and wetland (dams, ponds, lakes and creeks). Because many wetlands have fluctuating water levels, the wetland edge was defined at the high water level mark. The proportion of each habitat type and road density (ACT Planning and Land Authority 2007) was calculated within a 1km radius of each of the 19 wetlands.

Wetland Variables

Several wetland variables were considered, including surface area, mean maximum depth, pH, conductance, percent emergent vegetation and standard crop of turtle food. Surface area (hectares) of each wetland was calculated from digitized aerial photographs using the X-tools function in ArcView GIS. Maximum depth of each wetland was measured using a LSD digital sounder (Hondex PS-7) and water level was monitored fortnightly. Dissolved oxygen, pH, and conductance for each wetland was measured during each trapping period using a water quality analyser (hydrolab, Surveyor 4A), with measurements taken at five locations within each wetland (north,

south, east, west and centre). The proportion of emergent vegetation was measured in January and February 2007 by setting up equally spaced transects across each wetland. The number of transects varied between two and seven, depending on the size and the vegetative heterogeneity of the wetland.

Standing crop biomass of turtle prey items was estimated by sampling each wetland in mid-march 2007. Samples were taken from the four dominant habitats identified in the littoral zone of each wetland including open water over bare substratum, submerged macrophytes, floating macrophytes, masses of filamentous green algae, strands of the bulrush *Typha orientalis* and the spikerush *Eleocharis acuta* and other emergent plant communities. Samples were collected for a standard period of time (30 sec) using a long-handled net (250 μ m mesh) which was vigorously moved in a zigzag motion from the water surface to the bed ensuring agitation of sediment and from the base of the plants to the water surface. Samples were stained with rose Bengal and preserved in 10 % formaldehyde before being taken back to the laboratory for sorting. In the laboratory samples were washed and placed in a large sorting tray divided into 16 sections. Turtle prey items were picked out with the aid of a magnification lamp and each section was picked until two minutes of searching revealed no further results. Samples were then washed and the process was repeated. Sorted samples were stored in 90 % ethanol until the above process had been completed for all samples and each sample was then dried on absorbent paper for 10 mins and weighed (+ 0.01 g). Standing crop biomass likely provided an accurate measure of food availability because *C. longicollis* obtains food opportunistically and there is little evidence that they are selective in what they eat (Georges et al. 1986).

3.2.4. Statistical Analysis

Statistical analyses were performed with SPSS Version 14.0 (SPSS 2006) and SAS Version 8.2 (SAS Institute 2001). The assumptions of normality and homogeneity of variances were examined where appropriate and when data failed to meet these assumptions the data was transformed to approximate normal distributions and equal variances. If the data still deviated from these assumptions non-parametric tests were used. Statistical significance was accepted at the $\alpha = 0.05$ level unless specified otherwise. Means are reported with their standard errors unless otherwise stated.

I was initially concerned with examining demographic differences between sites. Firstly turtle abundance and proportion adults was compared between sites using one-factor ANOVA's, with total captures and proportion adults as the dependent variables and site as the independent variable. Total captures was \log_{10} transformed and proportion adults was arcsine transformed prior to analysis. Proportion females was also compared between the two sites using a Mann Whitney U test. Specific independent variables (wetland and landscape characteristics) contributing to variation in these same responses were then further explored using multiple linear regression analysis and stepwise comparisons. Before analysis I \log_{10} -transformed total captures and arcsine-transformed proportion adults and proportion females.

Survival probabilities of males and females were estimated from both sites using the Cormack-Jolly-Seber (CJS) open-population capture-recapture models (Cormack 1964, Jolly 1965, Seber 1965). Immature turtles were excluded from this analysis. The Program MARK version 4.2 (White and Burnham 1999) was used to generate maximum-likelihood estimates of survival and recapture probabilities. I started with a fully-saturated model in which apparent survival (ϕ) and recapture probabilities (p) were site and sex dependent $\phi(\text{site.sex})$, $p(\text{site.sex})$ and fitted a series of reduced-parameter models. The Akaike Information Criterion (AIC) was used to rank candidate models; if competing models had QAIC values of ≤ 2.0 , they were considered as having some support for the model (Lebreton et al. 1992). The general models adequacy to describe the data were assessed using a bootstrap goodness-of fit test using 500 simulations and an overdispersion parameter, \hat{c} , was derived by dividing the model deviance by the mean of simulated deviances (Cooch and White 2004). If there was evidence for overdispersion ($\hat{c} > 1$), the model was adjusted with the derived \hat{c} to improve model fit and calculated a quasi-likelihood estimator, QAIC_c (Burnham and Anderson 1998). Parameter estimates were derived as weighted averages based on their AIC values.

A chi-squared test was used to compare size frequency distributions of 5 mm size classes for turtle PL between sites. An ANCOVA was used to compare adult growth rates between sites, with growth rates as the dependent variable, site as the independent variable and initial carapace length as the covariate. Both growth rates and initial carapace length's were \log_{10} -transformed prior to analysis. Males and females were grouped for this analysis because of the low number of recaptures. Juveniles were

separated from adults because of their faster growth rates (Kennett and Georges 1990). The number of recaptured individuals that had moved between at least two wetlands and the incidence of injuries were compared between sites and between sexes using Fisher's Exact Tests.

3.3. Results

A total of 581 captures were made of 514 individual turtles from 18 wetlands distributed across the two sites. Approximately 2.7 times as many turtles were captured in Gungahlin wetlands than in wetlands from Mulligans Flat, but this difference was not significant (ANOVA: $F_{\text{site}} = 2.823$; $df = 1, 16$; $p = 0.112$; Table 7). More adults were captured than juveniles at both sites, with the proportion of adults averaging 0.78 ± 0.09 (range = 0.27 – 1; $n = 375$) in Gungahlin and 0.76 ± 0.07 (range = 0.40 -1; $n = 139$) in Mulligans Flat. This difference was not significant (ANOVA: $F_{\text{site}} = 0.159$; $df = 1, 16$; $p = 0.695$). There were a greater number of females captured than males at both sites, giving a sex ratio (female : male) of 2.2 : 1 in Gungahlin and 1.7 : 1 in Mulligans Flat. Proportion females did not differ significantly between sites (Mann Whitney U-test: $Z = -1.546$, $df = 289$; $p = 0.122$; Table 7), with the proportion of females in Gungahlin being 0.71 ± 0.05 (range = 0.43 – 1; $n = 328$) and the proportion of females in Mulligans Flat being 0.57 ± 0.08 (range = 0 - 0.9; $n = 105$). The number of recaptures at both sites was low (Table 7). Initial CL and PL of captured turtles are presented in Table 7.

3.3.1. Demographic Characteristics and Habitat Variables

A total of 555 individual turtles were captured from 19 wetlands (including the 18 wetlands used for all other analysis) distributed across Gungahlin and Mulligans Flat Nature Reserve. Landscape composition and wetland characteristics varied greatly among wetlands (Fig. 4; Table 8). Because standing crop of turtle food was unable to be sampled in one wetland (owing to wetland drying) the regressions of abundance and population structure against landscape and wetland variables were originally run with only 18 wetlands. Prey biomass was not a significant predictor variable for any of these regressions and was therefore removed as an independent variable ($R^2 < 0.295$, $F = 0.593$, $df = 17$, $p = 0.327$). All further regressions were then run using all 19 wetlands.

Table 7. Captures, recaptures and initial sizes of *C. longicollis* captured in 18 wetlands from Gungahlin and Mulligans Flat Nature Reserve. Values are reported as means \pm 1 SE (minimum value – maximum value)

	Total (n)	Recaptures (n)	Captures per wetland (n)	Carapace length (mm)	Plastron length (mm)
Gungahlin					
F	225	32	25.0 \pm 8.0 (2 - 72)	195.6 \pm 1.7 (146.3 – 279.1)	158.4 \pm 1.3 (112.5 – 209)
M	103	16	11.4 \pm 4.2 (1 - 11)	181.0 \pm 1.1 (156.0 – 214.0)	145.8 \pm 0.8 (125.7 – 171.3)
J	47	1	5.2 \pm 1.9 (0 - 40)	107.3 \pm 3.3 (66.3 – 143.0)	87.7 \pm 2.5 (57.1 – 114.8)
Combined	375	49	41.7 \pm 12.4 (5 - 119)		
Mulligans Flat					
F	66	10	7.3 \pm 1.8 (0 – 16)	187.3 \pm 3.3 (145.9 – 239.4)	152.3 \pm 2.7 (117.2 – 196.8)
M	39	7	4.3 \pm 1.2 (0 – 14)	170.1 \pm 1.9 (146.0 -195.2)	137.8 \pm 1.4 (120.5 – 158)
J	34	3	3.8 \pm 1.4 (0 – 19)	118.5 \pm 4.0 (68.2 – 143.8)	97.0 \pm 3.2 (57.3 – 118.5)
Combined	139	20	15.4 \pm 3.5 (2 – 30)		
Total					
F	291	42	16.2 \pm 4.5 (0 – 72)	193.7 \pm 1.5 (145.9 – 279.1)	157.0 \pm 1.2 (112.5 – 209)
M	142	23	7.8 \pm 2.3 (0 – 40)	178.0 \pm 1.0 (146.0 – 214.0)	143.6 \pm 0.8 (120.5 – 171.3)
J	81	4	4.4 \pm 1.2 (0 – 19)	112.0 \pm 2.6 (66.3 – 143.8)	91.6 \pm 2.0 (57.1 – 118.5)
Combined	514	69	28.5 \pm 7.0 (2-119)		

a)



b)

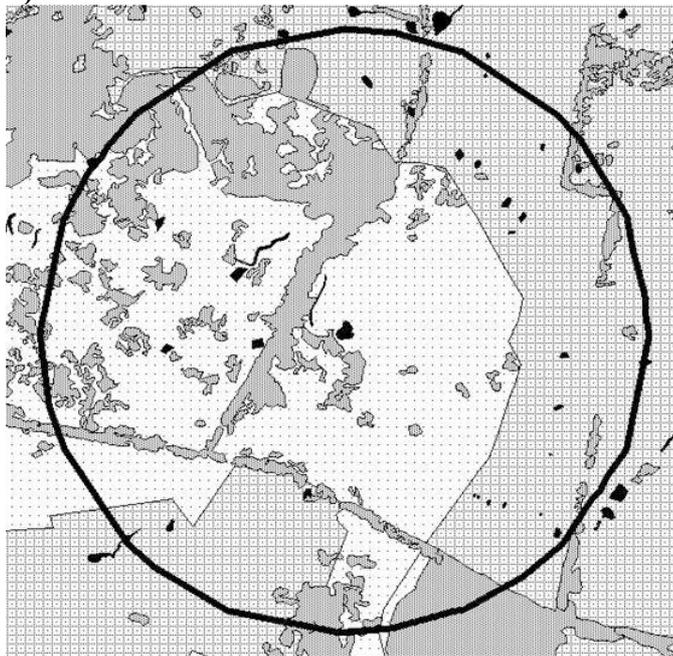


Figure 4. Typical variation in habitat composition and road density surrounding wetlands from Gungahlin and wetlands from Mulligans Flat Nature Reserve. Buffer drawn 1km from the edge of a wetland in a) Gungahlin and b) Mulligans Flat Nature Reserve.

Table 8. Summary of wetland characteristics and landscape variables 1 km from edge of wetlands in Gungahlin and Mulligans Flat Nature Reserve. Values are presented as means \pm 1 SE (minimum value – maximum value)

	Mulligans Flat	Gungahlin
Wetland characteristics		
Surface area (ha)	0.97 \pm 0.77 (0.05 – 7.1)	1.08 \pm 0.43 (0.15 – 3.76)
Depth (cm)	142.44 \pm 20.67 (25 – 225.25)	161.33 \pm 31.8 (35.5 – 337.5)
Emergent vegetation (%)	48.56 \pm 12.93 (0.0 – 100)	51.08 \pm 13.83 (0.0 – 100)
Turtle Food (g)	3.72 \pm 1.04 (0.15 – 9.8)	2.09 \pm 0.54 (0.76 – 5.26)
Oxygen (mg/L)	10.27 \pm 0.75 (8.01 – 15.09)	9.83 \pm 0.57 (6.92 – 12.76)
Conductance (μ S/cm)	77.49 \pm 10.61 (34.48 – 133)	479.24 \pm 83.64 (52.09 – 985.55)
pH	5.83 \pm 0.1 (5.16 – 6.24)	6.55 \pm 0.14 (5.84 – 7.21)
Landscape characteristics		
Road density (km/km ²)	1.29 \pm 0.52 (0 – 4.3)	8.88 \pm 1.34 (1.75 – 13.77)
Grassland	0.36 \pm 0.03 (0.25 – 0.56)	0.25 \pm 0.04 (0.01 – 0.42)
Woodland	0.34 \pm 0.05 (0.18 – 0.58)	0.05 \pm 0.01 (0.01 – 0.11)
Farmland	0.20 \pm 0.05 (0.0 – 0.40)	0.17 \pm 0.11 (0.0 – 0.89)
Infrastructure	0.0 \pm 0.0 (0.0 – 0.0)	0.33 \pm 0.07 (0.0 – 0.59)
Wetland	0.01 \pm 0.0 (0.0 – 0.03)	0.04 \pm 0.01 (0.0 – 0.08)
Green space	0.0 \pm 0.0 (0.0 – 0.0)	0.05 \pm 0.01 (0.0 – 0.12)
Land under development	0.09 \pm 0.04 (0.0 – 0.29)	0.10 \pm 0.08 (0.0 – 0.68)

Note: all landscape characteristics except for road density are proportions

A stepwise regression revealed pH to be the only wetland variable and proportion grassland to be the only landscape variable that explained variation in turtle abundance (pH: $R^2 = 0.459$, $F = 20.678$, $df = 18$, $p < 0.001$; grassland: $R^2 = 0.348$, $F = 9.081$, $df = 18$, $p = 0.008$), with turtles responding positively to increasing pH levels and negatively to increasing proportion of grassland. No wetland or landscape variable explained variation in the proportion of females in wetlands (wetland: $R^2 = 0.181$, $F = 0.441$, $df = 18$, $p = 0.838$; landscape: $R^2 = 0.369$, $F = 0.921$, $df = 18$, $p = 0.526$). Proportion of adults increased with depth ($R^2 = 0.209$, $F = 4.483$, $df = 18$, $p < 0.049$). Landscape variables also explained some of the variation ($R^2 = 0.726$, $F = 4.164$, $df = 18$, $p = 0.018$), with grassland ($p = 0.025$) and green space ($p = 0.016$) negatively affecting the proportion of adults.

3.3.2. Survivorship Analysis

Model Validation

Based on 500 bootstrap simulations, the most saturated model ($\emptyset(\text{site.sex}).p(\text{site.sex})$) that adequately fit the data ($P=0.116$) was determined. The overdispersion parameter \hat{c} was 1.27, which was used to adjust the model. Quasi-likelihood (QAIC_c) was therefore used for survival estimation and model fitting.

Survival Probability

The most parsimonious survival model for *C. longicollis* was $\emptyset .p.$ (Table 9); that is, survival (\emptyset) and recapture (p) rates were constant between site and between sex. Three other models also had some support (ΔAIC_c of < 2). The next most parsimonious model was $\emptyset(\text{sex}).p(\text{sex})$, where survival and recapture were independent of site but not sex. The other two models receiving some support were $\emptyset(\text{site}).p.$ where survivorship depended on site but not sex holding recapture rates constant and $\emptyset(\text{sex}).p.$, where survivorship depended on sex but not site holding recapture rates constant (Table 9). Akaike weights suggest the most parsimonious model $\emptyset .p.$ is over twice as plausible as the next 'best' model $\emptyset(\text{sex}).p(\text{sex})$. All other candidate models lacked any support from the data (Table 9);

Table 9. Survivorship and capture probability model statistics between site and sex for the mark-recapture model set for *C. longicollis* in Gungahlin and Mulligans Flat Nature Reserve. Reported values are Akaike's Information Criterion (QAIC_c), number of parameters (*k*), deviance (Dev), change in QAIC_c compared to the best ranked model (Δ QAIC_c) and model weights (*w*).

Model	QAIC _c	<i>k</i>	Dev	Δ QAIC _c	<i>w</i>
$\emptyset .p.$	339.90	2	33.23	0	0.362
$\emptyset(\text{sex}).p(\text{sex})$	341.53	4	30.79	1.63	0.161
$\emptyset(\text{site}).p.$	341.82	3	33.12	1.91	0.139
$\emptyset(\text{sex}).p.$	341.89	3	33.19	1.98	0.134
$\emptyset(\text{site}).p(\text{sex})$	343.48	4	32.74	3.57	0.061
$\emptyset(\text{site}).p(\text{site})$	343.80	4	33.06	3.90	0.052
$\emptyset(\text{sex}).p(\text{site})$	343.90	4	33.16	4.00	0.049
$\emptyset(\text{site*MFsex}).p(\text{site})$	345.63	5	32.85	5.73	0.021
$\emptyset(\text{site*GPsex}).p(\text{site})$	345.85	5	33.06	5.94	0.019
$\emptyset(\text{site.sex}).p(\text{site.sex})$	349.40	8	30.41	9.50	0.003

Table 10. Model averaged estimates of survivorship and recapture rates of *C. longicollis* from Gungahlin and Mulligans Flat Nature Reserve

Group	Survivorship (%)		Recapture rates (%)
	Mean and 1 SE	Upper and lower 95% confidence interval	
Gungahlin			
F	46.6 ± 9.5	28.2 - 65.9	14.2
M	51.7 ± 11.4	26.1 - 76.4	15.5
Mulligans Flat			
F	48.0 ± 11.2	25.5 - 71.3	15.2
M	53.4 ± 13.3	24.7 - 80.0	18.0

however these results are uncertain in light of the large confidence intervals. Sex and site specific survivorship and capture estimates are presented in Table 10.

3.3.3. Size Frequency Distribution

Gungahlin turtles overall were more frequently in the larger size classes than were turtles from Mulligans Flat (total: $X^2 = 74.18$, $df = 30$, $p < 0.01$; Table 7, Fig. 5). Size frequency distributions of juveniles were mostly similar between the two sites, whereas adults from Mulligans Flat were more frequent in the smaller size classes (115 - 140 mm). Adult turtles from Gungahlin were more frequent in the larger size classes (> 145 mm), except for the 175 - 180 mm size class where there were a higher number of turtles from Mulligans Flat. No turtles from Mulligans Flat appeared in the two largest size classes (205 – 215 mm).

3.3.4. Growth Rates

Growth rates were obtained from 41 recaptured turtles from Gungahlin and 30 recaptured turtles from Mulligans Flat (Table 11). Because of the low number of recaptures, growth rates for males and females were combined for each site. The percentage of recaptured adult turtles that had grown since their initial capture was 75% in Gungahlin and 31.3% in Mulligans Flat. Growth rates were compared between sites using ANCOVA with initial CL as the covariate, but as CL was not significant (ANCOVA: $F_{CL} = 0.006$; $df = 1, 36$; $p = 0.452$) the analysis was repeated as an ANOVA. Adult growth rates were significantly larger in Gungahlin than in Mulligans Flat (ANOVA: $F = 10.758$; $df = 1, 37$; $p < 0.01$; Table 11). The one juvenile recaptured from Gungahlin had a growth rate that was slightly below the average growth rate for juveniles captured in Mulligans Flat (Table 11).

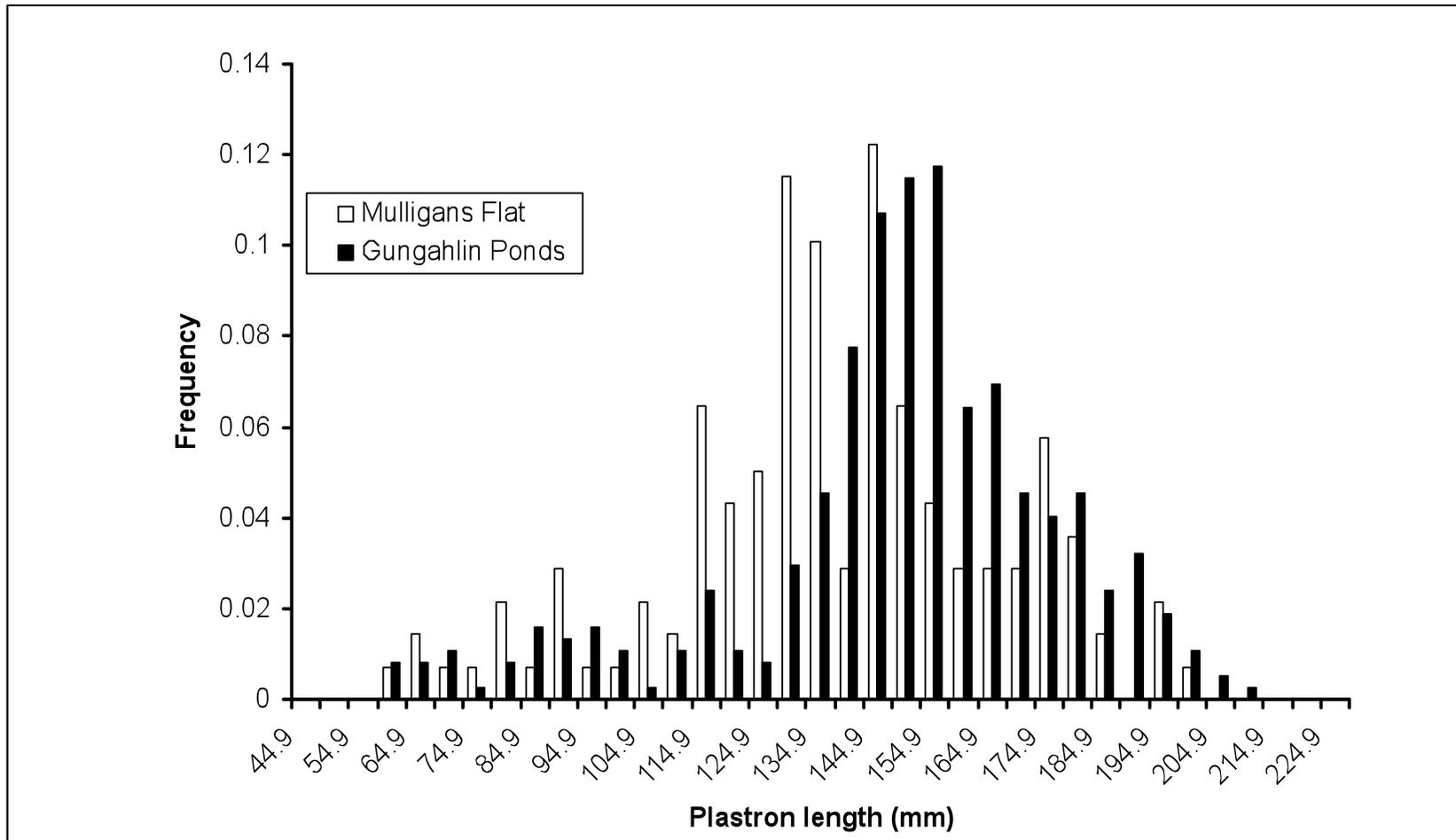


Figure 5. Size frequency distribution for *C. longicollis* captured in Gungahlin and Mulligans Flat Nature Reserve. Plastron size at maturity is 118.4 mm for males and 134.5 mm for females (John Roe, unpublished data).

Table 11. Growth rates for *C. longicollis* captured in Gungahlin and Mulligans Flat Nature Reserve. Growth rates are presented as means \pm 1 SE (minimum value – maximum value)

Group	n	Growth rates (mm/year)
Gungahlin		
M	12	1.4 \pm 0.4 (0 – 4.3)
F	28	1.3 \pm 0.3 (0 – 4.6)
J	1	5.2
Mulligans Flat		
M	10	0.2 \pm 0.2 (0 – 1.7)
F	12	0.2 \pm 0.1 (0 – 1.0)
J	8	6.8 \pm 2.5 (0 – 18.0)

3.3.5. Movement Rates

Fifty-three individual turtles from Gungahlin and 36 from Mulligans Flat were recaptured one or more times over the study duration. Of the recaptured individuals, 14 from Gungahlin and 12 from Mulligans Flat had moved between at least two wetlands. This difference was not significant (Fisher's Exact Test: $df = 1$; $p = 0.64$). Out of 30 males and 51 females that were recaptured at least once over the study duration, 12 males and 14 females had moved between two or more wetlands. Again this difference was not significant (Fisher's Exact Test: $df = 1$; $p = 0.32$).

3.3.6. Injuries

A total of 461 turtles from Gungahlin and 397 from Mulligans Flat were assessed for injuries. The incidence of injuries was similar between sites (Fisher's Exact Test: $df = 1$; $p = 0.15$) and between sexes (Fisher's Exact Test: $df = 1$; $p = 0.17$), with injuries occurring in 10% of turtles from Gungahlin compared with 6.5% of turtles from Mulligans Flat and 10.8% of females compared with 7% of males.

3.4. Discussion

I failed to detect any negative impacts of urbanisation on *C. longicollis* populations, but instead found that *C. longicollis* may respond somewhat positively to an urban environment. If there is a negative effect of urbanisation on this species, the level of sampling intensity and endpoints measured in this study were not suitable to detect it. Relative abundance, population structure, survivorship, movement rates and injury incidence of *C. longicollis* in the urban site were not different from an adjacent nature reserve. However, adult turtles in the urban site were larger and grew faster than those in the nature reserve. In addition, this study demonstrates that abundance and population structure of *C. longicollis* within wetlands is not impacted by surrounding road density. These results were unexpected given the high terrestrial mobility of *C. longicollis* (Roe and Georges 2007) and the associated likelihood of frequent encounters with roads and other hazards in the urban landscape. These findings lead

to the rejection of all three of my hypotheses, which were that urban populations would be 1) male-biased owing to differential mortality of nesting females, 2) adult biased owing to reduced recruitment and 3) that abundance and survivorship would be lower and injury incidence higher at the urban site. This study provides an example of a freshwater turtle that is apparently resistant to at least a moderate level of urbanisation. Given that most previous studies have found urbanisation to be detrimental to freshwater turtles (Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Budischak et al. 2006; Aresco 2005; Gibbs and Steen 2005), this study highlights the need to examine species-specific responses of wildlife in landscapes with differing degrees of urbanisation.

Even though abundance was not significantly greater in the urban landscape compared to the reserve, relative abundances of turtles were almost three times higher in the urban landscape. This, together with the significantly faster growth rates and higher frequency of larger adults in the urban site (Table 11; Fig. 5) suggests that a moderate level of urbanisation may even be beneficial for *C. longicollis*. One explanation for this positive impact could be that urban wetlands may be more productive as a result of eutrophication from urban runoff (Lee et al. 2006). Although there is some evidence that growth rates may be lower for turtles living in urban areas (Mitchell 1988; Budischak et al. 2006), eutrophication has previously been shown to benefit freshwater turtle species (Knight and Gibbons 1968; Graham and Doyle 1977). Eutrophication results in wetland macro-invertebrate communities that have low diversity but very high abundance (Brainwood and Burgin 2006), a condition that is likely to benefit generalist carnivores that obtain food opportunistically, such as *C. longicollis*. However, there were no differences in prey biomass detected in urban and nature reserve wetlands, though sampling was limited to a single event that did not account for seasonality. In addition, standing crop does not account for production which may be higher in the urban site but subject to greater predation levels. Higher abundance of turtles in the urban site may also be attributable to Gungahlin having almost five times as much aquatic habitat as Mulligans Flat. However, abundances were not population sizes, but instead relative capture rates with standardized and similar trapping efforts at each site. Perhaps the most likely explanation for benefits to turtle populations in the urban site is that wetlands in the urban site afforded turtles a longer growing season. Gungahlin wetlands are deeper and most remained

permanently flooded even during drought, whereas Mulligans Flat wetlands tended to dry and become unproductive for much of the year. Many turtles responded to wetland drying in Mulligan's Flat by aestivating terrestrially as early as mid-January and for maximum durations of 280 days, whereas all urban turtles remained in wetlands where they could continue foraging (Chapter 1). Thus, urbanisation not only increased the amount of habitat for this species, it also increased some measure of habitat quality (e.g., stability).

Few characteristics of the focal wetland and surrounding landscape features were found to influence the abundance and population structure of *C. longicollis*. The only wetland characteristic that explained some of the variation in *C. longicollis* abundance was pH, with more turtles occurring in wetlands that were less acidic. pH levels have not previously been found to influence freshwater turtle abundance, though pH does influence chemical solubility (including phosphates, heavy metals and oxygen) and macro-invertebrate communities (Brainwood and Burgin 2006) which may indirectly affect turtle abundance. The only landscape variable to explain variation in turtle abundance was grassland, with a higher number of turtles found in wetlands surrounded by lower proportions of grassland. This result is surprising given that grassland provides suitable nesting habitat for freshwater turtles and contrasts with another study that found abundance of painted turtles (*Chrysemys picta*) to be positively associated with the proportion of nesting habitat that surrounds wetlands (Marchland and Litvaitis 2004). *C. longicollis* abundance was not impacted by road density or any other landscape modification surrounding wetlands. Although this result was not expected, similar results were obtained for another common wetland reptile species, the northern water snake *Nerodia sipedon sipedon* (Attum et al. 2007).

Proportion grassland and green space (as found on the Gunghalin golf course) was negatively associated with the proportion of adult *C. longicollis* found in wetlands, suggesting that higher numbers of juveniles are found in wetlands surrounded by suitable nesting locations. Surprisingly, the abundance of females within wetlands was not influenced by nesting habitat or any other wetland or landscape variable. Marchland and Litvaitis (2004) found the proportion of adults to be positively associated with forest cover and the proportion of females to be negatively associated with forest cover, suggesting that higher numbers of juveniles and females were found

in wetlands surrounded by grassy areas suitable for nesting. The only wetland variable that explained some of the variation in proportion adults was depth, with a higher proportion of adults found in deeper wetlands. One possibility is that juveniles are largely confined to shallower vegetated wetlands where they are less conspicuous to predators, whereas the larger body size of adults and associated immunity from many predators allows them to occupy deeper wetlands (Pappas and Brecke 1992; Bury and Germano 2003).

The failure to detect any wetland or landscape variables that influenced the proportion of females also contrasts with a number of studies that have demonstrated male-biased turtle population in wetlands surrounded by high road densities (Marchland and Litvaitis 2004; Steen and Gibbs 2004; Aresco 2005). Male-biased turtle populations in urban environments are typically attributed to differential road mortality of nesting females. The similarity of turtle sex ratios over a wide gradient of road densities was unexpected given that *C. longicollis* females are known to regularly make terrestrial nesting excursions (Roe and Georges 2007) and are therefore likely to expose themselves to roads and other urban hazards. Open grassy areas suitable for nesting habitat (such as a golf course) surround most of the urban wetlands, perhaps reducing the need for females to travel long distances to nest (Baldwin et al. 2004). This reduces the likelihood that both females and emerging hatchlings would encounter roads and may also partly explain why the proportion of adults did not differ between the two sites. Roads have previously been shown to influence age structure of turtles, with higher proportions of adult painted turtles (*C. picta*) found further from roads (Fowle 1990). There was no evidence that this occurs for *C. longicollis*.

In spite of the hazards associated with urban habitats and the continued and extensive movements of turtles within it (Chapter 1, this study), survivorship of urban turtles was not different than in an adjacent nature reserve (Table 10). Although the validity of the survival estimates are compromised by large confidence intervals, this result is supported by radio-telemetry observations, where survivorship was found to be similar between the two sites for 36 turtles (Chapter 1). Incidence of injuries was also found to not differ between the urban area and the nature reserve. Injuries have previously been linked with higher incidence of mortality via predation (Rand 1954; Schoener 1979), and therefore the similarity of injury rates between sites provides

further evidence that turtles are not at increased risk of mortality in the urban landscape. This result was surprising given that mobile species, such as *C. longicollis*, are presumably at increased risk of mortality in urban areas owing to encounters with traffic and other urban dangers (Carr and Fahrig 2001; Roe et al. 2006). One explanation for low mortality rates may be the flooded and vegetated drainage lines in Gungahlin that facilitate turtle movements between wetlands and combined with culverts allow turtles to travel around much of this site without exposing themselves to roads and other urban hazards (Chapter 1). Several studies have found high mortality and injury rates for turtles living in urban environments (Mitchell 1988; Marchland and Litvaitis 2004; Conner et al. 2005; Budischak et al. 2006), though most of these studies have been based in urbanised landscapes in the eastern United States.

Survivorship estimates obtained in this study are much lower than those reported from the radio-telemetry study (>80%; Chapter 1) and those previously reported for *C. longicollis* (>90% Shine and Iverson 1995; >85% Roe 2008) and many other freshwater turtles (Frazer et al. 1991; Congden et al. 1993; Shine and Iverson 1995; Spencer and Thompson 2005). Low survival estimates are most likely owing to the low number of recaptures resulting from high emigration rates out of the 'study site' wetlands (i.e., in which trapping took place). To estimate emigration rates, I used observations of radio-tracked turtles for which movements and fates were known (Chapter 1). Emigration rate, calculated as the number of radio-tracked turtles originally captured in the 'study site' wetlands that had emigrated by the end of the study, was indeed high at 58 %. However, the model used to estimate survival was unable to distinguish emigration from death. Two other survivorship models that could incorporate emigration rates were investigated, a robust design model (Kendall 1999) and the joint live-dead model (Sandercock 2006), but my data would not fit the model assumptions.

Most studies of freshwater turtles have indicated that abundance, population structure and survivorship are negatively affected by urbanisation. One exception to this is a study that found abundance and population structure of painted turtles (*Chrysemys picta*) and stinkpot turtles (*Sternotherus odoratus*) to not be affected in an urban lake in central Virginia, USA (Mitchell 1988). Another study found that a turtle

assemblage in an artificial canal in Indiana, USA, was in good health despite impervious surfaces and a high density of roads surrounding the canal (Conner et al. 2005). This study provides evidence that some species of freshwater turtle may not be detrimentally affected by a moderate level of urbanisation and that turtle populations may even thrive in such environments. Similarly, abundance and species richness of lizards were found to peak in moderately urbanised areas of Arizona, USA, presumably because of increased primary productivity in these urban areas (Germaine and Wakeling 2001). However, beyond a moderate level of urbanisation, species richness and abundance of lizards declined rapidly (Germaine and Wakeling 2001).

The degree of urbanisation, as well as other properties of a landscape, will interact with specific life-history characteristics to influence the affect that urbanisation has on a species. I predicted that *C. longicollis* would be negatively affected by urbanisation owing to its high terrestrial mobility (Roe and Georges 2007). However, I have found that elements of the urban landscape allowed *C. longicollis* to remain mobile while avoiding roads and other urban dangers. *C. longicollis* is also a habitat generalist and can readily utilize urban habitat and exploit urban food sources. Many studies have shown generalist species, including reptiles, survive better in fragmented landscapes than specialists (VanDruff and Rowse 1986; Sarre et al. 1995; Smith et al. 1996; MacNally and Brown 2001; Marvier et al. 2004; Wilson et al. 2007). The amount of available habitat may even increase for some generalists living in modified landscapes as alternate resources become readily available (Saunders and Ingram 1995), such as may occur with increased aquatic habitat and food availability for *C. longicollis* in urban areas. Urban wetlands may even provide aquatic refuge for *C. longicollis* in periods of environmental instability, such as during drought. For instance, the urban wetlands in Gungahlin remained permanent and stable year-round whereas the reserve wetlands were highly variable and the only movements between sites were from the nature reserve into the urban area (Chapter 1). Freshwater turtles in urban environments may fare well in permanent wetlands, as wetland drying would force them to leave aquatic habitat and seek refuge in a highly hazardous urban landscape.

While other studies have suggested that some reptile species are able to cope with landscape modification (Dickman 1987; Mitchell 1988; Burkey 1995; Conner et al. 2005) and may even be benefited in such environments (Germaine and Wakeling

2001; Koenig et al. 2001), most recent research has demonstrated that reptiles can be particularly sensitive to habitat fragmentation (Sarre 1995; Smith et al. 1996; Boudjemadi et al. 1999; Driscoll 2004). This study demonstrates the importance of considering the interaction between species-specific responses of wildlife and properties of the landscape when determining the impact that landscape modification will have on a species. This information is critical for determining appropriate management strategies aimed at mitigating wildlife species decline in urban environments and in other types of modified landscapes.

Chapter 4

Synopsis

This study is the first to investigate an Australian freshwater turtle species in a moderately urbanised landscape and an adjacent nature reserve. I investigated the behaviour of *C. longicollis* in response to urbanisation, and compared population demography and survivorship of this species in wetlands surrounded by various land uses and road densities. This approach allowed me to determine the impacts of a moderate level of urbanisation on this species, and shed light on appropriate conservation strategies for managing *C. longicollis* in an urban setting. This study also provides ecological information that may assist wildlife managers in developing wetland management approaches for conserving semi-aquatic reptiles in urban areas, and increases understanding of anthropogenic habitat fragmentation and wildlife conservation.

4.1. Impacts of Urbanisation on *Chelodina longicollis*

I expected that *C. longicollis* would be detrimentally impacted by urbanisation because of their high terrestrial mobility (Roe and Georges 2007) and the associated risk of encountering roads and other urban dangers. However, this study demonstrates that *C. longicollis* is not negatively affected by a moderate level of urbanisation despite remaining highly mobile in the urban area. In fact, this study even provides some evidence that a moderate level of urbanisation benefits this species, at least in particular contexts. Given that most previous studies have demonstrated urbanisation to negatively affect abundance, survivorship and population structure of freshwater turtle species (Baldwin et al. 2004; Marchland and Litvaitis 2004; Steen and Gibbs 2004; Budischak et al. 2006; Aresco 2005; Gibbs and Steen 2005), this study highlights the need to consider interactions between species-specific responses and properties of the landscape when determining the impact that landscape modification will have on a species.

In Gungahlin, *C. longicollis* interacts with their landscape in a manner that allows them to persist and possibly thrive in this urban environment. Turtles are even more

mobile in Gungahlin than in the Mulligan's Flat nature reserve, with movement patterns and some estimates of home range sizes being larger for urban turtles (Table 3 and 4; Fig. 2). This suggests that the system of heavily trafficked roads and other urban hazards that bisect the landscape do not inhibit their natural movements. Such mobility within an urban landscape could be expected to lead to high rates of road mortality, but I could not demonstrate a difference in mortality rates of *C. longicollis* between Gungahlin and the adjacent nature reserve. Nor did the incidence of injuries differ between these two sites. Thus, it is apparent that the structure of the Gungahlin landscape enables turtles to remain mobile while avoiding roads and other urban dangers, a presumption which was corroborated by radio-locations of turtles.

Radio-telemetry revealed that turtles typically moved along drainage lines and used culverts, thereby avoiding roads along their travels between urban wetlands. High mobility is unlikely to harm *C. longicollis* in urban landscapes if those hazards can be avoided, as is apparently the case in Gungahlin. This was demonstrated by high abundance and similar population structures of turtles in Gungahlin and Mulligans Flat Nature Reserve (Table 7), and the fact that these population attributes were not sensitive to urban landscape modifications such as road density surrounding wetlands. Since *C. longicollis* is a habitat generalist that can readily utilise urban habitat, the large amount of aquatic habitat, high food availability, and the relative stability of flooding in the Gungahlin landscape is beneficial to the turtles. This was evidenced in the higher growth rates (Table 11) and larger sizes of adult turtles in Gungahlin (Table 7; Fig 5) relative to those in the nearby nature reserve. Thus, three of the primary detrimental fragmentation processes that typically alter urban environments (habitat loss, habitat isolation, and habitat degradation) are not threatening *C. longicollis* in this modified landscape.

Terrestrial aestivation behaviour for some individuals was substantially modified in the urban area, though it is unlikely that the motivation for this behaviour was to avoid roads. Mulligans Flat turtles spent on average 108 days in terrestrial habitat, whereas there was no indication that Gungahlin turtles spent any time in terrestrial habitats other than during movement between wetlands. Terrestrial aestivation behaviour also varied between sexes, with a greater number of females spending longer time periods in terrestrial habitat. *C. longicollis* are known to exhibit variability

in their preference to use terrestrial habitat for aestivation based on several natural habitat variables including hydroperiod variation, wetland isolation, and perhaps individual energy reserves (Roe and Georges 2007 in press). This study demonstrates that variation in aestivation behaviour can also be attributable to urban influences. Turtles in Mulligans Flat chose very specific microhabitats in which to aestivate, using woodland sites that 1) had denser canopy covers, 2) had deeper leaf litter, and 3) were closer to protective structures such as trees, logs and stumps (Table 6). I propose that turtles forego terrestrial aestivation in Gungahlin because they cannot find the specific micro-habitat conditions suitable for aestivation in this urban landscape, as the amount of woodland habitat is much lower in Gungahlin than in Mulligans Flat. Alternatively, Gungahlin wetlands remain relatively permanent, removing the need for *C. longicollis* to seek terrestrial refuge as a result of wetland drying. Gungahlin wetlands are also likely to be more productive resulting in high resource levels all year-round. This could increase the activity season of turtles as the need to aestivate as a response to seasonal variation is avoided. This contention is supported by the fact that turtles from Gungahlin are larger and grow faster than turtles from Mulligans Flat.

Although Gungahlin was not designed with the intention of conserving freshwater turtles, it provides an excellent example of how urban design could potentially be combined with wildlife conservation. Natural and artificial drainage lines and culverts, used for stormwater drainage purposes, were effective in allowing *C. longicollis* to maintain associations with several wetlands while avoiding urban hazards such as roads. In urban landscapes that do not have structures to facilitate movements so that roads can be avoided, it is likely that *C. longicollis* would be threatened by high road mortality rates. This is supported by other studies that have demonstrated high road mortality rates for other species of freshwater turtle (Dodd et al. 1989; Mitchell 1988; Haxton 2000; Hoff and Marlow 2002; Conner et al. 2005; Budischak et al. 2006), and highlights the need to maintain linkages between wetlands when managing urban turtle populations. The presence of two golf courses may also have substantially increased *C. longicollis*'s ability to persist in this urban area. The use of golf courses for wildlife habitat or corridors in urban areas is often overlooked (Terman 1994). In the case of freshwater turtles, golf courses can provide additional

nesting habitat and buffer wetlands from adjacent residential development and infrastructure.

4.2. Management Implications

Management of wildlife at the urban-reserve interface is challenging for many reasons, and wildlife managers are often faced with a critical decision. Do they fence in the nature reserve to protect resident wildlife from urban threats such as roads and domestic pets? Or do they allow the reserve to remain open to wildlife movements, thus threatening the integrity of the reserve for conservation purposes? Reserve managers are currently considering fencing Mulligans Flat Nature Reserve as a strategy to protect it from impacts of the encroaching Gungahlin suburbs. Our study suggests that managing wetlands as individual units would not protect *C. longicollis* from urban threats, and highlights the need to maintain habitat linkages between wetlands, but how important is maintaining linkages at a larger spatial scale, such as between the urban landscape and the adjacent nature reserve?

At least four turtles were observed to move from Mulligans Flat into Gungahlin but there were no movements in the opposite direction. *C. longicollis* often go in search of more permanent water when wetlands dry, and usually return to exploit the abundant resources characteristic of wetlands when they fill (Kennett and Georges 1990). But in urban areas where permanent wetlands are also highly productive due to urban runoff, the need for turtles to return to ephemeral water sources is reduced. This may have been why turtles were not observed to move from the urban site into the nature reserve. As Gungahlin is downstream of Mulligan's flat along the same drainage, it may be that the natural downstream movements of turtles during drought predated the urbanisation of the area (approximately 15 years ago). In the case of a fence, turtles would be prevented from leaving Mulligans Flat in search of the more permanent and downstream wetlands located in Gungahlin. In times of environmental instability, such as during drought, this could be potentially devastating to the reserve population as permanent wetlands in Gungahlin potentially provide refuge for turtles when Mulligans Flat wetlands dry. Thus, this study provides an example of a semi-aquatic reptile species that may be adversely affected by a management decision that does not

account for the importance of its movements at a landscape scale. Surprisingly, it is turtles from the nature reserve and not the urban area that that would be affected.

While the local population of *C. longicollis* would likely be affected by such a management decision, it is unlikely to be affected at a regional scale. The importance of studying such a common species is to determine the reasons that it is common and able to persist in a modified landscape. Lessons learnt from this study may assist with developing appropriate management strategies for other species of semi-aquatic reptiles in other urban areas, including species that are of conservation concern. The importance of maintaining inter-wetland connectivity is relevant to all semi-aquatic species that regularly or occasionally move between wetlands, and the provision of drainage lines and culverts, such as occur in Gungahlin, could be incorporated into urban design to facilitate species movements through developing landscapes. This may be especially important for semi-aquatic reptiles that live at the interface of a reserve and an urban area, where individuals may be attracted to urban wetlands but have a greater risk of mortality from roads, potentially resulting in a source-sink situation (Pulliam 1988; Meffe and Carol 1994).

4.3. Recommendations for Further Study

This study was limited both on a temporal and spatial scale due to resource limitations and a number of recommendations for further investigations have arisen. First, research should be conducted to delineate the true effect that fencing Mulligans Flat will have on turtle persistence in the Gungahlin area. This should include a long-term monitoring study that covers periods of environmental instability (e.g. drought) and that determines whether mitigating actions need to be implemented. Such actions may include turtle crossing structures or daily patrols of the fence perimeter during the season of optimal turtle activity. Second, an extensive and long-term survey of the abundance and population structure of turtles in the Gungahlin area should be undertaken to further substantiate the results of this study. Because Gungahlin was only recently developed, the full impact of urbanisation on turtles may not yet be evident, or at least may not be detectable with our level of sampling intensity. Further investigations are therefore required to confirm that turtles are not negatively affected

by urbanisation in this landscape. Studies are also required to examine the impacts of urbanisation on turtle recruitment.

While this study has management implications for other semi-aquatic reptiles, urbanisation will affect species differently depending on species-specific and landscape characteristics. Further investigations are therefore also required to determine appropriate management actions for such species. For example, in situations where urban wetlands are variable in their flood-dry cycles, freshwater turtles are likely to be heavily affected because they may leave the wetland to seek refuge in a highly hazardous landscape. Wildlife management in this situation may need to focus on maintaining suitable terrestrial aestivation sites located close to wetlands, perhaps by preventing firewood collection and maintaining litter cover and woodland habitat. Wildlife species that are habitat specialists are less likely to utilise urban habitat, and some species may not cope with increased pollution levels in urban areas. Management for these species will need to focus on maintaining or rehabilitating patches of habitat that are of high quality and that are buffered from urban processes. Determining suitable management strategies for urban wildlife is therefore likely to require case by case examination. However, gaining a general understanding of the types of processes likely to influence species persistence in urban areas is an important first step to effectively mitigate wildlife decline in urban landscapes.

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