AN INVESTIGATION OF THE DISTRIBUTION
PATTERNS OF AQUATIC VERTEBRATES
ACROSS FOUR SITES IN THE
UPPER PARRAMATTA RIVER CATCHMENT

by

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Jason Ross

September, 2000
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ABSTRACT

Freshwater vertebrates were sampled in different waters of the Upper Parramatta River catchment to determine their patterns of distribution with respect to water quality. A total of 730 animals (831 captures) were caught between November 1997 and December 1998. Data were collected on seven vertebrate species. These included three freshwater turtle taxa: *Chelodina longicollis* (Shaw, 1794), *Emydura macquarii dharruk* (Cann, 1998) and *Trachemys scripta elegans* (Wied, 1839). The four fish species studied were *Anguilla australis* (Richardson, 1841), *Anguilla reinhardtii* (Steindachner, 1867), *Tandanus tandanus* (Mitchell, 1838) and *Cyprinus carpio* (Linnaeus, 1758). Two of these species, *T. s. elegans* and *C. carpio*, are exotic species.

With the exception of *T. s. elegans*, that was represented by a single capture from Toongabbie Creek, turtles and fish were collected from all sites, however, their abundance varied among sites. Viable populations of the recently described *Emydura macquarii dharruk* (Cann, 1998) were found at all sites. They were collected across all size cohorts and juvenile recruitment rates were similar to the endemic *C. longicollis*. Capture rate for the two turtle species, *T. tandanus* and *C. carpio* were highest from Toongabbie Creek, while eels (*A. australis* and *A. reinhardtii*) were in fewer numbers at this site. In contrast, the largest number of both eels and lowest numbers of other vertebrates were caught from Darling Mills Creek.

The abundance of aquatic vertebrates was correlated with physiochemical water quality parameters. The large aquatic vertebrates studied did not correlate with the commonly used
parameters for aquatic health. The assumption that ANZEEC water quality parameters are the definitive determinant of aquatic health for vertebrates is, therefore, erroneous.

Endemic turtles were correlated with different parameters to the endemic fish: *C. longicollis* abundance correlated with water temperature and chlorophyll-a, *E. m. dharruk* was a strong indicator of turbidity and suspended solids and, to a lesser extent, salinity. Both eel species *A. australis* and *A. reinhardtii* were correlated with dissolved oxygen levels and *T. tandanus* with salinity, while the feral *C. carpio* was found to be ubiquitous against all parameters, making management of this pest species through specific physiochemical manipulation unlikely.
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CHAPTER I: INTRODUCTION

1.1 Introduction

Freshwater is a limited resource, particularly in Australia which is the driest inhabited continent. Tempered by drought, Australia’s total effluent is $345 \times 10^6$ megalitres annually. By comparison, rivers on other continents may discharge almost as much water. For example, the Danube discharges $281 \times 10^6$ megalitres annually (Dougherty, 1996).

With the exception of the Murray - Darling Basin system, most of Australian freshwater habitats are restricted to coastal drainage systems at the peripheries of the continent (Legler et al., 1993). Australia, therefore, possesses relatively limited resources for aquatic vertebrates.

In New South Wales, there has been conflict over water use. Many riverine environments are degraded, aquatic natural resources are in decline and biodiversity threatened (Harris, 1996). Efforts to manage and restore such rivers require effective tools to measure their ‘health’. One difficulty has been developing quantitative indicators that are applicable in different ecological regions, from free-flowing montane to regulated and lowland river reaches. Together with physio-chemical water quality parameters, aquatic fauna and flora have been used as bioindicators to establish a system’s health (Heath, 1987; Arthington, 1986).

1.2 Aquatic vertebrates as bioindicators

Organisms are subject to a variety of environmental (both man-induced and natural) stresses and multiple measures of health are required to identify them accurately. Bioindicators are typically used to assess the effects of such stressors on aquatic organisms. They can range
from the assessment of biomolecular responses to overall population and community level responses (Arthington, 1986). Aquatic vertebrates, used as bioindicators, may provide multiple measures of community health, including biological organisation and time scale response (Heath, 1987).

A range of aquatic species have been used by researchers as bioindicators of environmental condition and ecosystem status as diverse as *Cyprinus carpio* (Linnaeus, 1758) (Bales, 1992) and nematodes (Arthington, 1986; Niles and Freckman, 1996; de Ruiter and Moore, 1993). Genetic markers have also been assessed, including the effects of contaminants on DNA integrity and genetic diversity of fish (Adams, 1989). In addition, the health of fish populations (Adams, 1990), their environmental tolerance, resistance and adaptation to pollution have also been investigated as potential bioindicators (Baine, 1985). Physiological and biochemical measures have also been investigated (Heath, 1987).

A study by Heath (1987) illustrated that bioindicators may provide an early warning of environmental problems, identify cause and effect between stressors and biological responses, assess the integrated responses of organisms to environmental stress and evaluate the effectiveness of remedial actions on system health. By measuring health responses in terms of response time and level of biological organisation in such ways, researchers may establish causal relationships between stressors and biological effects.

Harris (1998) surveyed freshwater fish populations for their potential as bioindicators, with the aim to develop better catchment management strategies. Fish populations were intensively studied over two years, at 80 sites, at varying altitudes. He found that the State’s
degraded river ecosystems were rapidly losing biodiversity and that fish species diversity was in decline. The threatened status of 11 New South Wales species was confirmed, and one species, *Tandanus tandanus* (Mitchell, 1838), previously abundant, was recommended for review with the potential of being classified as a threatened species. The main identified cause of the fish decline was the prevalence of alien pest species, particularly *C. carpio* (carp), the dominant fish across the Murray-Darling system. Modification of waterways, especially flow regulation, have resulted in high numbers of *C. carpio*. For example, Harris (1998) found one carp per square metre of river surface area in rivers such as the Bogan.

However, fish are not the only aquatic vertebrate species affected by river degradation. Cann (1993) suggested that the major environmental threat to Australia’s freshwater turtles was continued degradation of waterways, particularly in eastern Australia. This degradation may include potentially toxic organisms such as cyanobacteria (Cann, 1993; Wong and Burgin, 1997). Such responses to pollution indicate that these species may also be of benefit in environmental assessment of waterways.

When utilising aquatic vertebrates the relationship between behavioural factors and environmental variables (eg. water quality data), need to be considered (Adams, 1989). Such variables may include habitat and food availability, competition for resources, levels of contaminants and varying physiochemical factors, for example water temperature and dissolved oxygen levels (Heath, 1987).

These phenomena may result in an integrated response impact on growth, fecundity and population response. Alterations at community and population level can range from gender
ratio imbalance and food web alterations, to a complete breakdown of the immune system and, ultimately, DNA damage. The effective use of vertebrate bioindicators therefore involves determining the integrated physiological and bioenergetic effects of environmental stressors, for example, on fish populations and evaluating the ecological significance of these effects on the biotic integrity of stream, river or reservoir systems (Adams, 1989).

While effective indicators and monitors of ecosystem health are necessary, they are frequently poorly developed. Brooks and Mulvihill (1995) assessed different bioindicators to develop an index of ecological integrity for forested headwater ecosystems in Louisiana, U.S.A. They observed that a suite of monitoring tools were a matter of priority in ascertaining baseline data for assessing human impact and establishing effects on management measures such as rehabilitation.

1.2.1 Water quality and aquatic vertebrates

Chemical factors such as dissolved oxygen levels, pH and dissolved nutrients and turbidity, can impact on distribution and abundance of aquatic vertebrates (Ferraro and Burgin, 1993). Physiochemical water quality monitoring has therefore been widely used as a measure of environmental health. Water quality parameters commonly utilised are briefly summarised.

1.2.1.1 Water fluctuations

Annual water level fluctuations are frequently important cues for fishes and other migratory species. In the drier areas, with the advent of the wet season, intermittent streams flow, stagnant pools and backwaters on the flood plains are flushed, lake levels rise and river flow increases. As a consequence of such flooding, terrestrial nutrients are flushed into the rivers
and provide habitat and food for aquatic vertebrates. Reproduction and growth are, therefore, often correlated with water level (Lowe-McConnell, 1975).

Such water fluctuations, may trigger migration in both turtle and fish species. Some fishes may move onto flooded plains to spawn and fish growth may be maximised during such periods (Moyle and Cech, 1982). Seasonal water level fluctuations may also result in specific species interactions. For example, food may be abundant, resulting in reduced competition among species. In contrast, receding water levels may confine fish to pools and competition for resources may intensify (Zaret and Rand, 1971). Competition may be greatest during periods when quantity and diversity of food has become limited (Lowe-McConnell, 1975).

1.2.1.2 Water temperature

Since temperature governs the speed of chemical reactions, it impacts on all aspects of aquatic biology. Water temperatures in creeks and lakes may undergo annual cycles, related to seasonal changes in air temperature. The smaller the volume of the water body, the larger the amplitude of seasonal water temperature, while diurnal temperature changes are superimposed on the seasonal temperature trends (Laxton, 1997).

In contrast, to smaller, shallow water bodies, fish from relatively large lakes and rivers live in environments with comparatively smaller temperature fluctuation. As a consequence, they may exhibit rapid growth and shorter life cycles. However, temperature may be less important than other parameters, such as water fluctuations, on movement and reproduction. Cooler water temperatures are likely to impact on reproductive cycles, growth and movement (Moyle and Cech, 1982).
Water temperature has been observed to play an important role in feeding and brumation/aestivation activities. For example, freshwater turtle and eels may brumate at lower water temperatures, when they live off body fat and minimise body movement (Chessman, 1988a; Jellyman, 1991).

Spencer (1995) compared the digestive performance of *Enydra macquarii* (Gray, 1831) on diets of fish or plants at two temperatures, and related how both diet and temperature affect its food selection in nature. Digestive efficiency of *E. macquarii* was affected little by body temperature, in contrast to consumption rates and rates of passage, which were strongly influenced by both temperature and diet. In combination, these responses resulted in a slower rate of digestion at 20°C than at 30°C.

A study of *T. tandanus*, (freshwater catfish) revealed that it can withstand temperatures as high as 35°C, but temperatures below 4°C are lethal (Lake, 1967). Growth rates have been observed to be highest in water temperatures between 20°C and 25°C (Lee, 1973). *Anguilla australis* (short-finned eel, Richardson 1841) was shown to tolerate higher temperatures (39.7°C) than other taxa, based on a test of temperature tolerance of over 200 freshwater fish species (Richardson *et al.*, 1994).

Lethal water temperature for both *Anguilla australis* (Richardson, 1841) and *Anguilla reinhardtii* (long finned eel, Steindachner 1867) was observed to increase with eel size (Jellyman, 1974). In a subsequent longer-term trial on *A. reinhardtii*, water temperature was observed to be the only parameter that significantly affected catch per unit effort. When water
temperatures dropped below 12°C, catch rate declined and eels effectively ceased movement (Jellyman, 1991).

A correlation was also observed between mean movement per day and average water temperature. Eels showed minimal movement below 10°C. During periods of low temperature, eels congregated at the centre of the lake in waters slightly deeper than 2m, or on the soft clay bottom (Jellyman, 1996).

Similarly, C. carpio survive a wide temperate range (5°C to 32°C) and can cope with low oxygen levels (Hume, 1982). Templeton (1995) found that they possessed a wider temperature tolerance than other fishes studied. When forced by local environmental conditions to migrate, they actively swim and have been observed jumping obstacles 1.0m in height (Hume, 1982).

1.2.1.3 Nutrient levels

In aquatic systems, nutrients are predominantly carried by water and are essential for the growth of flora and fauna. Although a range of nutrients are required for growth, the most common nutrients measured in water quality studies are nitrogen and phosphorous (Laxton, 1997).

Nitrogen occurs in natural waters as dissolved gas and may occur as simple ions or complex organic molecules. Some primitive aquatic organisms such as cyanobacteria, can obtain their nitrogen requirements from dissolved nitrogen in the water column. However, all
multicellular animals obtain their nitrogen by consuming complex organic molecules (Laxton, 1997).

Ammonia is the major end product of protein catabolism in fish and is excreted primarily as un-ionised ammonia from the gills. It is also produced during decomposition of organic material and is contained in fish faeces (Tucker and Boyd, 1985). In aquaculture systems, management of ammonia levels has proven difficult, but reduction in dietary components leads to lower ammonia concentrations (Tave, 1993).

In water quality studies nitrogen, in the form of ammonia, organic and oxidised nitrogen, may be measured. Oxidisation in the environment forms nitrite ions and ultimately nitrate ions. The sum of nitrate and nitrite concentrations is known as oxidised nitrogen, whereas organic nitrogen incorporates the nitrogen bound in organic material. This sum of ammonia, organic nitrogen and oxidised nitrogen is referred to as total nitrogen and includes all of the non-gaseous nitrogen in the aquatic environment (Laxton, 1997).

In a study by Tave (1993), 400 channel catfish, *Icterus punctatus*, were randomly allocated to six ponds. The impact of lowered ammonia levels on catfish was assessed, together with the impact on other water quality parameters such as pH, dissolved oxygen and water temperature. Reduced stocking rates were observed to reduce ammonia levels and water temperature was observed to be significantly higher in low ammonia ponds, as was dissolved oxygen and pH. The increases in pH and dissolved oxygen resulted in increased phytoplankton populations, shifts in the composition of phytoplankton populations and decreased rates of decomposition and respiration amongst catfish.
1.2.1.4 Turbidity

Turbidity is a measurement of the clarity of water or the amount of material carried in suspension. It is expressed in nephelometric turbidity units (NTU) and is measured as scattered light, caused by suspended particles in a sample of water (Laxton, 1997).

While lakes and most smaller streams are usually relatively clear, large rivers are turbid with suspended and/or solid materials. However, there are seasonal variations in turbidity, with the clearest water flowing during dry periods (Roberts, 1972).

One impact of high turbidity, has been the radiation of fishes in rivers that do not rely primarily on vision for prey capture, for example *T. tandanus*. In clear water, bright colours may be an advantage in communication among fishes and as a result, most brightly coloured ‘tropical’ fishes come from lakes and clear streams (Moyle and Cech, 1982). *Cyprinus carpio* can create turbidity in its localised environment, due to a combination of highly active spawning behaviour, sometimes benthic diet and strong, sometimes thrashing, swimming motion (Lake, 1978). When introduced to a waterbody it frequently changes habitat characteristics to a more turbid environment.

Manipulations of biomass of *C. carpio* revealed that the species had significantly increased turbidity and algal blooms (King, 1997). This was confirmed by a study of two billabongs on the floodplains of the Murrumbidgee River, New South Wales. These billabongs were partitioned and densities of *C. carpio* were manipulated to establish high and low density treatments. Throughout the experimental period turbidity and phytoplankton mass were
significantly higher in the high density treatment and the total concentration of phosphorous was greater.

Studies of the effects of *C. carpio* on water quality have also included manipulating stock density and food availability. Under high impact conditions, it was observed that turbidity increased ten fold within four days, with complete loss of two of the five plant species assessed. There was also triple the level of plant uprooting than was observed under low impact conditions. In contrast, there was no evidence of increased nutrients or associated algal blooms in ponds stocked with *C. carpio* but this was thought to be due to low phosphorous levels in the surrounding sediment (Roberts, 1995).

1.2.1.5 Salinity

Water salinity may increase with elevated dissolved solids. It is normally measured as parts per thousand (ppt.) against dissolved solids. In freshwater systems salinity is generally low (< 2 ppt.), compared to ocean salinity of approximately 35 ppt. However, in most estuaries pronounced salinity gradients occur where freshwater and seawater mingle under the influence of tides (Moyle and Cech, 1982). Salinity may rise during drought and fall with high rainfall (Laxton, 1997).

Extremes of salinity has been observed to affect fecundity in some species. For example, the reproductive capacity of the fecund *C. carpio* is affected by low salinity and pH levels, at least when combined with lead toxicity (Laxton, 1994).
1.2.1.6 pH

pH is defined as minus the logarithm of the Hydrogen ion concentration. It is expressed on a scale of 0 to 14, the lower the number the stronger the acidity. Seven is the neutral point where the number of hydrogen ions equals the number of hydroxyl ions. Values above neutral denote alkalinity (Laxton, 1997).

pH is a biological indicator that provides a clue to the origin of the water, the geology of the catchment, the type of biological activity occurring within the water body and the likelihood that it is contaminated by industrial or domestic waste (Laxton, 1997).

There is often correlation between fish distribution patterns and abundance, and water chemistry patterns of parameters such as pH, nutrient concentration and dissolved inorganic and organic salts. Although most lake fishes are capable of living in a pH range of 5-8.5, few freshwater species cope with pH outside of this range. (Johnson, 1976). The desirable pH ranges for *T. tandanus* have been recorded as 6.3-7.5 (Lee, 1973; Tucker and Boyd, 1985).

*Cyprinus carpio* has recorded optimum growth levels with a pH range between 7.5-8.2. This pH range promotes the best conditions for growth of dietary food organisms, such as zooplankters, for example the water flea, *Daphnia* spp. (Michaels, 1988). More recent studies by Templeton (1995) have found that *C. carpio* possesses a wider tolerance range for pH than other fish species investigated.
1.2.1.7 Dissolved Oxygen

Most life forms require dissolved oxygen for life and its concentration is probably the most important single parameter measured in water quality studies. Some aquatic animals use atmospheric oxygen but many, such as fish, extract oxygen directly from the water (Laxton, 1997) while others, such as freshwater turtles, extract oxygen both from atmospheric oxygen and from aquatic sources (Dalem, 1998).

If the dissolved oxygen level is below saturation, oxygen will diffuse from the air into water, until saturation is attained (Harris, 1998). Oxygen enters the water through the surface film, but as the dissolved oxygen levels rise at the atmospheric/water interface, oxygen entering the film must be quickly mixed throughout the water column. This only readily occurs where the water body is turbulent. If this does not occur, the rate of diffusion will fall to a slow rate (Tucker and Boyd, 1985). The rate of oxygen diffusion is dependant upon an oxygen saturation deficit (concentration at saturation minus actual concentration), the ratio between water surface, water volume and the degree of turbulence. Natural aeration is performed by wave action and wind velocity across the surface may mix surface water with the underlying waters.

Highly saline waters do not hold as much oxygen as freshwater and warmer water temperatures result in lower dissolved oxygen levels. For example, Lee (1973) observed that a freshwater body at 1°C held 12.8ppm of dissolved oxygen and possessed 7.6ppm when water temperatures were raised to 25°C.
The amount of oxygen (or any other gas) at normal atmospheric pressure required to saturate water varies with water temperature and dissolved solids concentration. Species, such as *C. carpio* (Hume, 1982) and freshwater turtles (Chessman, 1988a), are able to survive low oxygen environments.

Most fish species die when dissolved oxygen levels are reduced to 1.0 ppm., with levels of 4 to 5 ppm. required for ideal growth (Lee, 1973). The body functions of fish require oxygen. As fish feed they become more active and hence use more oxygen. The process of food digestion also requires oxygen (Tucker and Boyd, 1985). However, *C. carpio* has been shown to thrive at very low dissolved oxygen but during winter months when this species undergoes a form of hibernation, levels can be further reduced by comparison to other fish species, such as trout, which require four times the level of dissolved oxygen under the same conditions (Michaels, 1988).

Decay of organic matter may also affect dissolved oxygen levels, as oxygen bound in organic matter decomposes (Lee, 1973). For example, in the presence of a die-off of phytoplankton, exotic fish (e.g., *C. carpio*) or plants consume oxygen and native species, such as *T. tandanus*, need to compete for it (Tucker and Boyd, 1985).

Weather conditions can also impact on dissolved oxygen levels, with consecutive cloudy days combined with low wind velocity, causing supplies of oxygen to decline (Laxton, 1997). With limited light, algae die and decay and this, in turn, may deplete oxygen. In contrast, sunlight causes phytoplankton to produce oxygen through photosynthesis (Lee, 1973).
Catfish oxygen consumption rates vary with dissolved oxygen concentration, feeding status, weight of fish and water temperature. For example, it was observed that oxygen consumption in ponds with *T. tandanus* lowered as dissolved oxygen concentrations decreased and there was a high correlation between capture numbers and average dissolved oxygen levels at dawn (Tucker and Boyd, 1985).

1.2.1.8 Chlorophyll-a

Phytoplankton biomass may be estimated by a measure of chlorophyll-a in a known volume of water. Concentrations vary greatly in natural waters, with values as high as 500 micrograms/L in nutrient rich environments with high light intensities (Laxton, 1997).

Chlorophyll-a concentration has been shown to gradually increase during the growth season of species such as *T. tandanus* and this was observed to coincide with increased feeding rates (Tucker and Boyd, 1985). Its abundance also increases in response to the increased nutrients available for plant growth during warmer summer months. As the principal nitrogenous waste product is excreted relative to food intake, concentrations of total ammonia increase, together with ammonia nitrification, when food intake is elevated. This may result in high nitrate concentrations in aquatic environments with substantial animals populations (Lee, 1973).

1.3 Purpose of study

Physiochemical water quality parameters therefore influence aquatic vertebrate population dynamics. As indicated above, water quality parameters impact on aquatic organisms such as fishes and turtles. However, there is generally a paucity of data on such linkages. The
current study was undertaken in the Upper Parramatta River Catchment to further investigate these relationships. It was designed to investigate the correlation between physiochemical water quality parameters was routinely taken in the area to monitor the health of the aquatic ecosystem and the most abundant resident vertebrates. The major focus of the study is to assess the effectiveness of water quality parameters in determining the health of such populations.

In this chapter I have introduced the concept of bioindicators, in the next chapter (Chapter 2) I will provide a context for the study of the fish of the area, and in Chapter Three, the same will be done for the freshwater turtles. In Chapter Four there will be a brief description of the Upper Parramatta River Catchment and a description of the study sites. Chapter Five contains the methodology, in Chapter Six the results are presented and finally in Chapter Seven the significance of the results will be discussed in the context of the broader knowledge base.
CHAPTER II: INTRODUCTION TO FISH OF THE
UPPER PARRAMATTA RIVER CATCHMENT

2.1 Introduction

Fish are the most numerous and diverse of the major vertebrate groups and dominate the
waters of the world (Moyle and Cech, 1982). Their diversity is reflected in the large number
of extant species, estimated at 18,000 (Nelson, 1990) while 60% of all vertebrate taxa are
fish (Bone et al., 1995): 41% of these are freshwater inhabitants and 1% (eg. eels) are able to
move between marine and freshwater environments (Cohen, 1970).

Within Australia the freshwater fish fauna is unique. Although the fishes are diverse, the
number of species is relatively low, compared to other continents. Approximately 190
indigenous, non-estuarine species from 39 families, have been described (Merrick and
Schmida, 1984). While worldwide the majority (over 93%) of primarily freshwater fishes are
catfish, carp and characins, the only two primary freshwater fishes that are endemic to
Australia are lungfish *Neoceratodus* and osteoglossid, *Scleropages* (Bone et al., 1995).

It is widely accepted that the reasons for the paucity of Australian freshwater species is due to
continental isolation for over 40 million years, together with the unique geological and
climatic conditions that prevailed during this period. A few species, such as the lungfish,
have fossilised remains dating back 400 million years (Lake, 1971).

In his study of New South Wales river systems, Harris (1998) found that of the 22,580 fish
catched across 80 sites during the two year survey, 18.4% were exotic species such as *C.*
*carpio*. These ferals differed significantly in abundance across regions and river types, for example, *C. carpio* were not found in any of the 20 montane sites, but were present at altitudes <500m. In coastal sites, they occurred in six regulated lowland sites with an altitude of between 0-60m while all inland rivers had higher *C. carpio* biomass than the coastal rivers.

Although spatial differences among regional types have been identified as the major source of variation in introduced populations, such as *C. carpio*, they have also contributed to the change in the proportional abundance of native species. Flow regulation has also reduced the resilience of many N.S.W. rivers and their associated native fish populations and has sometimes favoured alien fish species. The status of some endemic fish species is being reviewed. For example, scientists from the Cooperative Research Centre for Freshwater Ecology have reviewed the status of *T. tandanus* and *A. australis* and recommended that these once abundant species need careful monitoring due to their declining numbers (Harris, 1996). Both these species are endemic to the Upper Parramatta River Catchment, together with the endemic eel *A. reinhardtii* and the feral species *C. carpio* (Laxton, 1994, 1997).

### 2.1.1 Eels

There are >600 species of eels, belonging to 22 families. These are found in a variety of habitats, ranging from freshwater lakes and streams, to coral reefs and the deep sea. Within the family *Anguillidae*, there are 15 species. These animals live in freshwater and spawn in the ocean (Moyle and Cech, 1982). Within Australia this family is represented by four diadromous species of the genus *Anguilla* (Lake, 1971). They spend most of their life cycle
in freshwaters and migrate downstream to spawn at sea when sexually mature. After spawning it is assumed that the adults die (Lake, 1971).

Schmidt (1928) and Ege (1939) provided comprehensive reviews of the taxonomy and distribution of Australian eels. Two of these species, *A. australis* and *A. reinhardtii*, occur in the Upper Parramatta River Catchment (Laxton, 1994, 1997).

*Anguilla australis* possesses a large mouth that extends from below the small eye-teeth on the palate. It has narrow vertical gill openings and a dorsal fin that is extended just forward of the anal pore. The belly is grey to silver and the fin coloration is dark. This species is restricted to the Tasmanian, south-east and southernmost north-east coastal areas, however, it also occurs in New Zealand and as far north as Lord Howe Island (Merrick and Schmida, 1984).

*Anguilla reinhardtii* can be recognised by a broad head with fleshy lips and a large mouth that extends back to behind the eye. Teeth on the palate form a long narrow band and a dorsal fin extends well forward of the anal pore. Coloration varies from olive-green to brown with distinct purplish mottling. When seaward spawning migration commences the adults lose their spots and become bright silver (Schmidt, 1928). This loss of pigmentation can result in misidentification. As a consequence migrant *A. reinhardtii* may be referred to as shortfinned eels (Sloane, 1984).
2.1.1.1 Distribution and abundance

*Anguilla* species possess a broad range in coastal areas from the Cape York Peninsula to Melbourne on mainland Australia and are also found in northern and eastern Tasmania (Lake, 1971).

Marquet (1996) recorded that *A. reinhardtii* were established everywhere on the mainland of Australia in running waters across the 5-580m altitudinal ranges. In Tasmania, the species is restricted to lower, freshwater reaches of waterways and the estuaries of the north-eastern and eastern rivers (Sloane, 1984). This species also occurs in New Caledonia and Lord Howe Island. Although it is known to occur in a variety of habitats, *A. reinhardtii* most often occurs in rivers, rather than still waters (McDowell and Beumer, 1980).

Since 1888, 29 dams of >7m height and an additional 52 smaller water supply structures have been built across urban Sydney, Wollongong and the Blue Mountains (N.S.W.). Gehrke *et al.* (1999) assessed the changes in fish assemblages due to modified catchment flows. They found that the impact was significant and that there were differences between regulated and unregulated rivers. *Anguilla reinhardtii* were more abundant in regulated reaches, in contrast to the feral *C. carpio* that were more common in lowland reaches across systems such as the Hawkesbury-Nepean.

Overall population estimates for *A. australis* are difficult to accurately assess, due to paucity of data. However, the most extensive study of the distribution of freshwater fishes in Victoria, undertaken between 1967-1991 (Raadik, 1991) incorporated over 153 sites. It was revealed that *A. australis* was the most widespread of all species, found at 78 sites, while *A.*
*reinhardtii* was the fourth most widespread species. *Anguilla reinhardtii* was recorded at 63 of the 153 sites, but was predominantly limited to low or intermediate elevations, whereas *A. australis* was present at higher elevations and into sub-alpine areas across sites (Raadik, 1991).

Recapture rates for adult *A. australis* and *A. reinhardtii* may be high within a 1km of initial capture. During a three-year period, Chisnall (1993) reported a 60% recapture rate for larger *A. reinhardtii*. Over the same period, *A. australis* recapture rate was 40%. In a more extensive study (5 years) of movement of short-finned eels in Lake Ellesmere (New Zealand), Jellyman (1996) recaptured approximately 20% (1982) of his catch (n = 9956). Some individuals were recaptured up to seven times. Most non-migratory eels were recaptured at, or adjacent to, their original capture point, although some eels moved considerable distances within three weeks of tagging. Recapture rates from other studies of *A. australis* include 15% (Helfman, 1984), 11% (Vladykov, 1957) and 18.5% (Beumer, 1979).

In most eel population estimates small juveniles (<400mm) are under-represented in such catches. This is assumed to be a result of biased sampling methods or due to factors such as migration (Berg, 1990).

**2.1.1.2 Migration**

Little is known of the route or duration of the oceanic migration for either eel species, however, they are thought to spawn in the same general area, near New Caledonia. Details of the return migration of the leptocephali and glass eel stages are also sketchy. However, it is assumed that the drift back from New Caledonian spawning grounds, to eastern Australian
rivers, takes approximately one year (Mann, 1979). *Anguilla reinhardtii* elvers arrive in New South Wales waters in early summer, whereas glass eels and elvers enter Victorian estuaries between January and late May (Beumer and Harrington, 1980).

The same life cycle and behavioral pattern of sea migration, spawning, development and return of glass eels and elvers to freshwater has been noted in both *A. australis* and *A. reinhardtii*. However, they are thought to use separate spawning grounds in the tropical pacific. The larvae, or leptocephalus, are the dispersal phase and the East Australian current acts as the transport mechanism (Sloane, 1984). After migration to the upper reaches of coastal rivers, the young eels spend between 15 and 30 years in freshwater, where they grow and mature, before descending the river again, to migrate to their oceanic spawning grounds (Lake, 1971).

Although adult European freshwater eels are reported to migrate downstream at night, *A. reinhardtii* has been observed to move downstream and over dam walls, including Warragamba Dam (N.S.W.) during daylight (Mann, 1979).

Glass eels of *A. reinhardtii* have been observed to exhibit a more restricted movement into freshwater, during late summer when water temperatures decline from their summer high. In contrast, the high degree of variability in the timing of movement of *A. australis* larval results in a prolonged influx of glass eels into waters at varied times of the year. *Anguilla reinhardtii* glass eels were recorded to move into freshwater between February and July (February and April in the Parramatta River: Ege, 1939), whereas *A. australis* glass eels were only captured between March and December (Sloane, 1984).
Beumer and Harrington (1980) reported that in Victoria *A. australis* migrate from the sea into estuaries from May onwards, and *A. reinhardtii* eels from January to late May. This migration may occur at temperatures as low as 4.5°C, which is lower than studies for the European eel (*A. anguilla*) that have been shown to prefer temperatures around 6-8°C (Tesch, 1977).

Both species move upstream in response to factors such as falling salinity and rising water temperatures. Large numbers of *A. reinhardtii* elvers have been observed climbing a weir into freshwater at Parramatta on the Parramatta River in February (Ege, 1939). In contrast, groups of elvers (75 to 100mm long) have been known to migrate upstream in the Hawkesbury - Nepean system in August and September (Powell, 1930; Mann, 1979; Beumer and Harrington, 1980).

Mann (1979) observed that each year when elvers migrate upstream in the Hawkesbury-Nepean system, they congregate at barriers, such as Warragamba Dam spillway. Migration peaks every three years (in terms of number of individuals) and this has coincided with the year of peak adult migration.

The highest catch of *A. australis* juvenile per unit effort in these circumstances is under complex cover. Smaller eels (<400mm) are closely associated with marginal habitat (Chisnall, 1996). Sampling has revealed that eels <350mm occurred closer inshore, whereas larger eels were more commonly found offshore. Size gradation was observed, whereby smallest larvae were closely associated with the edge but beyond 50m from the river’s
margin, the size of the eels did not continue to increase. While most studies have shown eels to move large distances, Jellyman (1996) observed that smaller adult eels did not move more than a few kilometres, whereas larger eels moved greater distances (Sloane, 1984).

Both eel species have different life-cycle movement patterns (Sloane, 1984). The development time of larvae has been observed to be a critical factor in their distribution and size at time of freshwater invasion (Tesch, 1977).

Despite variations in sampling methodology, figures available on the average age of captured migrating eels in freshwater systems are similar. Calculated using the burnt otolith method, the average age of migrating A. australis from the Clyde River (Australia) was 22.1 years (Sloane, 1984). This compared with Todd’s (1980) study of A. schmiditii. He calculated that the average age of migrating eels in three rivers was between 19 and 24 years. Details for A. reinhardtii have not been published but this species is thought to migrate at a size similar to its New Zealand relative, A. dieffenbachii (Scott, 1934) and individuals caught while migrating have been calculated to be 35-45 years old (Sloane, 1984).

2.1.1.3 Habitat preferences

Chisnall (1996) found that juvenile A. australis are more abundant in lake and lotic waterways, than lentic or open water systems. Within these systems they show a preference for underwater woody debris and dense log jams, rather than open water. Such cover aids in the ambush of their prey (Koehn et al., 1994).
By comparison, *A. reinhardtii* prefer gravel or sandy bottom, rather than a muddy one. This habitat, in combination with a decline in abundance of several catadromous fish species, as well as an avoidance of areas with a low winter water temperature, are all considered to play a part in *A. reinhardtii* habitat selection (Scott, 1934).

2.1.1.4 Morphometrics, growth and maturation

*Anguilla australis* and *A. reinhardtii* do not possess external physical characteristics for sex determination (Todd, 1980). The only sexual dimorphism detected in *A. australis* is size. Males are much shorter than females and they comprise most of the catch in freshwater systems, they are also relatively larger than *A. reinhardtii* at all stages of the life cycle and may grow to 1.1m in length and weigh 7kg (Lake, 1971).

In contrast, *A. reinhardtii* may attain a length of 2.0m, 500mm in girth and up to 16.3kg in weight. More commonly, however, they are observed to be in the range of 1.2-1.4m (Whitley, 1960), although the longest documented *A. reinhardtii* was 2.0m and reliable reports suggest that individuals may grow to 3.0m in deep isolated lakes where downstream migrations may be prevented for long periods by factors such as drought (Merrick and Schmida, 1984).

Glass-eels of *A. australis* have been observed to be larger than *A. reinhardtii*. A collection of *A. reinhardtii* indicated that they enter Tasmanian waters at essentially the same size as they do in New South Wales and New Caledonia (Sloane, 1984).
A number of studies have shown growth of temperate *Anguilla* species at temperatures as high as 28°C (Jellyman, 1977). However, Sloane (1984) demonstrated that eel growth rates were sub-optimum during late spring/early summer and attributed this to low summer water levels and associated reduced available feeding areas. Growth is slower in *A. australis* than for *A. reinhardtii* (Sloane, 1984). Burnet (1969) also observed differential growth rate between the two species, across different river systems. However, there is variability in individual growth rate and this is also reported for other *Anguilla* spp. (Frost, 1945; Deelder, 1957).

Chisnall (1993) reported that growth rates for *A. reinhardtii* averaged 65mm +/-15mm annually, whereas *A. australis* grew an average of 29mm +/-18mm. Individuals monitored were estimated to be between 13 and 23 years, although most were 11 to 16 years. New Zealand studies have demonstrated that *A. australis* commence maturation at a length of >480mm and weight of 225g (McDowell and Beumer, 1980).

No valid estimates of fecundity are available for *A. australis* or *A. reinhardtii*, but a female *A. australis*, presumed to be on its spawning migration, was reported to have ‘millions of eggs’ in its ovaries (Whitley, 1956).

2.1.1.5 Eel Diet and feeding

The timing of feeding activity for *A. australis* and *A. reinhardtii* is closely correlated with water temperature (Vollestad, 1986; Tesch, 1977; Jellyman, 1991). Increased *A. reinhardtii* activity occurred with rising water temperatures and changes in atmospheric pressure. In contrast, *A. australis* responded significantly to water level, temperature and pressure change.
Maximum captures of *A. australis* are, therefore, expected during summer, particularly during high flow periods. This would coincide with feeding at the margins of water bodies (Jellyman, 1991).

Although principally carnivorous, *A. australis* has been described as an opportunistic omnivore. Adults eat other fish species, worms, amphipods, shrimps, mayflies, mollusks and water plants. Victorian studies have shown that feeding follows a seasonal pattern and is most intense at night, near the shoreline, during spring and summer. *Anguilla reinhardtii* is also principally carnivorous, although some plant material may be ingested. Their feeding pattern is also influenced by seasonal changes, with feeding most intense at night during spring and summer (Stephenson, 1953; Lake, 1971).

Sloane (1984) suggested that previous studies of eel diet tended to have been in standing waters with few published data from running water habitat. He observed that the three most important food items in such an environment were *Diptera, Ephemeroptera* and *Trichoptera*. The same dietary components were important to both species, although relative abundance of prey taken by *A. reinhardtii* was reversed (namely, *Trichoptera < Ephemeroptera < Diptera*) compared to *A. australis* (namely, *Trichoptera > Ephemeroptera > Diptera*). Most prey consumed were *Simuliidae (Diptera), Trichoptera (Rhyacophilidae)* and *Ephemeroptera (Baetidae)* (Sloane, 1984). However, in another study it was observed that small fish comprised up to 80% of the overall eel diet (Hayes, 1991). Mysids (zooplankter) contributed both directly and indirectly to the diet of eels; they are prey for eels and small fish. Turbid waters reduce the feeding efficiency of such fish and hence there is frequently a high biomass of mysids in turbid waters, where eels tend to feed.
Fish are often only prey of larger *A. reinhardtii*. Fish bulk accounted for 9% of stomach content in *A. reinhardtii* of <20cm, 18% in individuals 20-40cm, and 30% in animals >40cm (Sloane, 1984).

### 2.1.1.6 Disease and abnormalities

Eels are attacked by a large number of parasites (Gosper, 1991) and bacterial infections, such as Aeromoniasis, which attacks *Anguilla* spp. in all phases of their life cycle. However, there is differing opinion regarding whether this disease attacks predominantly larger eels (Egusa, 1979) or is more widespread amongst pigmented elvers (Rickards, 1978). Other common diseases include Myxobacterial (gill disease), red spot disease (caused by excessive salt levels), abdominal dropsy (an infection that results in fluid accumulating in the abdominal cavity) and Branchionephritis (gill nephritis) (Egusa, 1979).

In a comparison of European populations of *A. anguilla* and *A. reinhardtii*, it was observed that regional parasite species richness was higher in *A. reinhardtii* than *A. anguilla* and that helminth populations were more diverse than in any other fish species studied: 27 parasite species were collected from 82 eels. This parasite diversity is comparable with populations found in some birds. Climatic conditions, age and endemism of the host were suggested as possible explanations for this richness (Kennedy, 1995).

There is a paucity of data on *A. reinhardtii* disease in Australia but studies of the Brisbane River population have shown that some individuals were infected with a specialized fish blood fluke (Gosper, 1991). The population also had gills infected with a protozoan of the
Myxidium group. Other flatworms and nematodes have also been reported (Martin, 1974). For example, the nematode Anguillicola australiensis was found in A. reinhardtii (Johnson and Mawson, 1940). It was collected from nine of ten locations sampled. The population biology of this nematode in its host is apparently similar to such parasites in other Pacific eels (for example, Anguilla japonica) and the lack of pathogenicity observed may reflect a long period of host-parasite co-evolution and/or lower infection rates (Kennedy, 1994).

Anguillicola australiensis is apparently restricted to A. reinhardtii, whereas A. australis has also been recorded with Anguillicola novaezelandiae. Only 7.8% of larvae were uninfected. Both A. australiensis and A. novaezelandiae passed through the swimbladder wall into the lumen and moulted into adults. Alternatively they were destroyed, if passage past the wall was delayed (Johnson and Mawson, 1940).

2.1.2 Tandanus tandanus

In addition to the two species of eels in the upper Parramatta River, an endemic member of the catfish, T. tandanus, occurs. Family characteristics include an elongate eel-like body with the tail tapering to a point. In addition, members possess no scales, a flattened or rounded head and four pairs of mouth barbels. Dorsal and pectoral fins possess serrated spines and no adipose fin (Munro, 1955). There are over 1,250 species of catfish (Tucker and Boyd, 1985) and they are usually distinguished by the arrangement of colour pigment and other external features, most prominently fin features (Tucker and Boyd, 1985).
There are 11 described species of eel-tailed catfishes (Family Plotosidae), representing four genera, however, it is generally accepted that there are additional undescribed species (Lake, 1971).

The *Tandanus* originated in the Western-Pacific region tens of millions of years ago (Merrick and Schmida, 1984) and fossils assigned to *T. tandanus* have been reported from the Oligocene period, some 38 million years ago. It is the largest Australian endemic freshwater photosid. They grow to 900mm in length and may weigh 6.8kg, although individuals are rarely collected above 2.0kg (Lake, 1967). They are recognised by a large head, flattened below with moderate sized eyes, a ventral mouth with thick fleshy lips, anterior tubular nostrils on front of the upper lip and a high anterior dorsal supported by serrated spines and six rays (Merrick and Schmida, 1984). Colouration may vary with size. Up to 150mm, individuals are grey or brown and often mottled with brown to black blotches. Larger fish usually have less mottling than juveniles (Llewellyn and Pollard, 1980). It is one of few Australian native fish species to have been studied in any depth.

Evidence of speciation has been investigated using morphological and electrophoretic characteristics amongst five populations east of the Great Diving Range (N.S.W.) and six populations west of the range. It was concluded that there was an absence of morphological differentiation, however, electrophoretic evidence demonstrated three discrete gene pools, including an undescribed species from the Bellinger River and one from the Nymboida River. The remaining populations were not differentiated (Musyl, 1996).
2.1.2.1 Distribution, abundance and movement

Endemic to Australia, *T. tandanus* possesses a wide range, including most of the South Australian Gulf and Murray-Darling drainage. It is also present in eastern coastal streams from south of Sydney to north of Cairns (Lake, 1967) and is most abundant in lakes and backwaters. It inhabits and spawns in flowing streams (Llewellyn and Pollard, 1980), however, Keenan (1997) observed that abundance and age to size ratio varied significantly between adjacent waterways.

*Tandanus tandanus* tend to be solitary and do not generally migrate. Courting behaviour occurs at water temperatures above 20°C in spring and summer and water temperature is considered the primary stimulus for spawning. Flooding apparently is not necessary (Davis, 1977).

2.1.2.2 Reproduction

Female *T. tandanus* generally spawn annually after sexual maturity. In contrast, males may spawn several times during the mating season (Lee, 1973). The presence of reproductive repressive factors, such as *C. carpio*, has been shown to affect fecundity (Tucker and Boyd, 1985). The paired ovaries can contain between 2000 and 21000 oocytes, but numbers are dependant upon the size of the female. Larger fish have higher numbers of eggs, with an egg diameter range of between 2.3 and 3.2mm (Lake, 1978).

As water temperatures rise, the male prepares a nest ranging in size from 0.6 to 2.0m in diameter, utilising materials such as sand and gravel. If water levels fluctuate, perhaps
exposing a nest above water level, several nests may be built. This limits spawning success (Lake, 1967).

2.1.2.3 Pollution and *Tandanus tandanus*

The influence of chemical factors on *T. tandanus* population structure has been investigated in New South Wales. It was observed that populations are endangered by localised pollution. Endosulfan, an organochlorine pesticide, was observed to affect populations across the Gwydir River (cotton growing areas), compared to control samples at Horton River and Pindari Dam (Novak, 1989). It has also been observed that herbicides, such as Simazine, used to control phytoplankton blooms, significantly reduced their reproductive capacity (Tucker and Boyd, 1985). However, the introduction of other chemicals, such as copper, to *T. tandanus* habitat was shown not to have a significant impact on growth rate, feeding capacity or dietary intake (Gatlin and Wilson, 1985).

2.1.2.4 Growth and maturity

*Tandanus tandanus* growth rate is variable. Length may range from 120-280mm at 18 months, while 170-360mm may be achieved in 30 months and 250-480mm in 42 months (Davis, 1977). Both female and male *T. tandanus* mature at a similar size (400-500mm in length), however, they may not mature until they are five years old.

2.1.2.5 Diet and feeding

In contrast to both eel species, adult *T. tandanus* are essentially carnivorous benthic feeders, capable of exploiting a wide range of food sources, however, larger fish prefer larger prey (Davis, 1977). Crustaceans are the most important components of the diet, followed by
insects, snails and small fishes, such as gudgeons. Young *T. tandanus* prey on small fish more than adults. Shrimps, such as *Macrobrachium australiense*, are the most important food in summer, whereas insects (especially midge larvae) are the most important food in winter months. Unlike phytoplankton, zooplankton often forms a natural food for catfish (Tucker and Boyd, 1985).

They rely on temperature as a key for feeding and movement within a water body and may be less active during winter months when food supplies and water temperatures decrease, compared to other seasons (Davis, 1977).

### 2.1.2.6 Disease and abnormalities

*Tandanus tandanus* commonly suffer from bacterial diseases such as *Columnaris*, *Aeromonas* and *Pseudomonas* (Lee, 1973). *Columnaris* disease is visible as lesions on external areas of the body or in the mucous membranes, such as those in the mouth. *Aeromonas* is commonly characterised by grayish patches on the skin or bloody areas that erode, including eye and body cavities. *Pseudomonas* symptoms are similar to *Aeromonas* and require laboratory analysis for identification (Tucker and Boyd, 1985). Other diseases that can affect catfish include viral, fungal and algal infections, nutritional and environmental diseases, such as ‘gas bubble’ (result of abnormally high oxygen levels) (Lee, 1973).

During an extensive study undertaken within New South Wales (Harris, 1998), high levels of abnormality were observed in many species, including *T. tandanus*, with up to 25% showing evidence of parasites or disease. Continued exposure to low dissolved oxygen levels has been shown to increase bacterial infection (Tucker and Boyd, 1985). They may also act as host to
a range of protozoa, flatworms, tapeworms and nematodes and are susceptible to ‘ich’ and Lernae infection (Beumer, 1982).

### 2.1.3 Cyprinus carpio

In addition to endemic eels and catfish, the upper Parramatta also has populations of a feral species, *C. carpio*, belonging to the family Cyprinidae. This family has one of the largest number of species (approximately 1450) of all fishes and comprises many freshwater species that have an extensive range throughout Asia, Africa, North Europe and North America (Lee, 1975). They are endemic to most areas of the world, except polar regions, South America and Australia (Michaels, 1988).

*Cyprinus carpio* (Eurpoean or Common carp) can be recognised by a thick compressed body, a small mouth with a pair of barbels at each corner, large body scales, a long dorsal fin with three or four spines and 15 to 24 rays. It also possesses a deep caudal and forked fin and is golden or olive-green in colour with dark fins, sometimes with reddish borders (McDowell and Shearer, 1980). They can grow up to 1.2m and weigh 60kg. In Australia, individuals may reach 16kg, but are more commonly collected at less than 3kg (Lake, 1978).

#### 2.1.3.1 Distribution, abundance and movement

Several species of Cyprinids were introduced to Australia over a century ago and *C. carpio*, introduced into Victoria in 1960, underwent a population explosion in the Murray-Darling system (Lee, 1975). At least three distinct stocks have been introduced. The ‘Prospect’ group was one of the first known introductions of this species and has been confined to Prospect Reservoir (western Sydney region) since 1908. A golden strain is thought to be confined to
the Riverina irrigation system and can be distinguished by its reddish coloration (Llewellyn and Pollard, 1980). The third group, the Boolara group, was the strain introduced into Victoria in 1960. In addition, hybrids between C. carpio and Carassius auratus (goldfish) are common (Shearer and Mulley, 1978).

*Cyprinus carpio* has been described as the worst pest of Australian waterways (Roberts, 1997). It uproots vegetation which changes the structure of the habitat and increases turbidity. This, in turn, affects the physical characteristics of the aquatic environment and may impact on other species (Ebner et al., 1996). For example, it may form over 80% of the total fish biomass in areas across the Murray-Darling Basin (Roberts, 1997).

An indication of its impact can be gleaned from a report submitted to the N.S.W. Fisheries Research and Development Corporation (FRDC). It was revealed in that report that major species once exploited by the inland fishery had altered since 1980. Historically the dominant catch included *T. tandanus* and *Perca fluviatilis* (Redfin) another problem introduced species, whereas the major catch more recently has been *C. carpio*, (Reid, 1996). Catch numbers have now stabilised at about 150 tonnes, after rising rapidly in the 1970s to a peak of 548 tonnes.

### 2.1.3.2 Population structure

In a recent study, Gawne (1996) investigated the age structure of *C. carpio* in freshwater systems to evaluate the role of hydrological factors across the Murray – Darling Basin on the recruitment of *C. carpio*. The study was designed to facilitate management strategies to reduce *C. carpio* numbers through minimising their reproductive success.
2.1.3.3 Reproduction, growth and maturity

Male *C. carpio* usually mature at two to four years and females at between three and five years. They are capable of rapid growth, compared to other species such as *T. tandanus* (Koehn et al., 1994).

Fecundity in *C. carpio* is high. A 900g female can produce approximately $10^4$ oocytes, while females that weigh 4-5kg can produce $10^5$ oocytes (Lake, 1971). Large females may breed at intervals of four weeks, when conditions are favourable. However, spawning can vary greatly between sites, locations and in successive years (Bowerman, 1975; Cadwallader and Backhouse, 1979).

This species does not migrate to spawn and they have a small home range and generally only make short, random movements (Lake 1967; Reynolds 1983). Spawning and recruitment are most prevalent in spring and summer (water temperatures between 17°C and 25°C; Lake, 1967), but the success of spawning was observed to be influenced by specific hydrological 'triggers', such as water temperature. Despite this, *C. carpio* has been shown to have enhanced fecundity in areas impacted by agricultural development (Harris, 1996). River regulation is a major cause of such degradation and may impact detrimentally on endemic fish populations and favour reproduction in *C. carpio*.

2.1.3.4 Diet and feeding

Compared with the essentially carnivorous *T. tandanus*, *C. carpio* are omnivorous with feeding behaviour adapted for sucking and straining the benthos or insects and plants at the
water's surface. Juveniles feed on microscopic algae, rotifers and crustaceans (Cadwallar, 1979).

Hepher (1968) found that protein-rich diets were necessary for C. carpio to maximise growth efficiency. In aquaculture, it has been observed that a lack of Vitamin A results in a significant decrease in body weight, resulting in dulled skin colour and hemorrhage of the fins (Aoe, 1968).

2.1.3.5 Disease

Dove (1998) found four species of exotic monogeneans in C. carpio from the Australian Captial Territory (A.C.T.) including Dactylogyrus extensus and Dactylogyrus anchoratus, the first published record of these species in Australia. They have also been recorded with the parasitic copepod, Lernaea cyprinacea (Robinson, 1981).
CHAPTER III: INTRODUCTION TO TURTLES OF
THE UPPER PARRAMATTA RIVER

3.1 Introduction

There are currently seven genera and 30 described forms of Australian freshwater turtle: 25 species and five sub-species. However, this may be an underestimation of their diversity and many currently recognised, variable and widespread species may ultimately consist of several taxa (Cann, 1998).

Most Australian species are members of the suborder Pleurodira (family Chelidae) which possess distinct ankle joints and webbed feet, each with four or five claws. Cryptodira (hidden neck turtles) are represented by the monotypic family, Carettochelyidae (Legler et al., 1993).

Generally, Australian chelids are small to moderate in size with adults ranging from a carapace length of 100mm for Emydura kreffti males and 120mm for Pseudemydura umbrina females (Burbidge, 1981), to a maximum of 485mm for Chelodina expansa (Legler et al., 1993). Size dimorphism is also variable. Males are larger in Pseudemydura umbrina, the sexes are the same size in Rheodytes leukops, and males are smaller than females in all other species (Legler et al., 1993).

Juvenile chelids are generally more brightly coloured than adults. For example, hatchlings of C. longicollis have a bright orange and black plastron. In contrast most adults are drab in
colour and largely unpatterned, although bright yellow, orange and pink colours have been observed as markings on the head and neck of adults in some populations (Harness and Morlock, 1989).

The carapace and plastron are joined at the side by bridges. The head and forelimbs may be extended from, and withdrawn within, the front opening and the hind limbs and tail likewise at the rear end. This affords protection from predators and in winter slows the loss of body heat (Beck, 1991).

In south-eastern Australia, three species of freshwater turtle, *Chelodina expansa* (Gray), *Chelodina longicollis* (Shaw, 1794) and *Emydura macquarii* (Gray, 1831), occur sympatrically (Chessman, 1988a). *Trachemys scripta elegans* (red-eared slider: Wied, 1839), have been introduced into the waters of the Sydney region, probably via the pet trade from North America (Burgin and Emerton, 2000).

**3.1.1 Description, distribution and abundance for *C. longicollis* and *E. m. dharruk***

The adult carapace of *C. longicollis* is broadest behind the centre, oval to elliptical in shape, dark-brown to black in colour, flattened and usually with a smooth marginal rim (Ernst and Barbour, 1989). Each forelimb possesses four claws and the gular shields of the plastron meet in front of the inter-gular (Cann, 1998). It can be distinguished from other species of the genus by its plastron, which is greatly expanded anteriorly and is usually 1.5 times longer than broad. It is widest at about the level of the humerals. The intergular is at least twice as long as the suture between the pectorals and the expanded anterior lobe is broad, extending
laterally beyond the inner edges of the overlying marginal plates of the carapace (Cogger, 2000).

_Chelodina longicollis_ possesses a smooth posterior plastron rim, depressed vertebrales and a pronounced medial groove on the second to fourth vertebrales. The plastron is large compared to _E. m. dharruk_, almost covering the entire carapacial opening and it has a deep posterior notch compared to _E. macquarii_ (Ernst and Barbour, 1989). In previous studies of _C. longicollis_, Parmenter (1976) and Dalem (1998) took measurements of a range of shell characteristics, including carapace length and breadth, plastron length and breadth and shell depth. All of these measurements have been demonstrated to have a straight line correlation that approaches unity.

The endemic genus, _Chelodina_, includes the semi-aquatic, long-necked chelid turtles and _C. longicollis_, the most abundant and widespread turtle in Australia (Parmenter, 1985). It possesses a range widespread across eastern and south-eastern Australia where it occupies permanent waters, backwaters, swamps, farms dams and other ephemeral water bodies (Goode, 1967; Cann, 1978; Stott, 1988; Dalem, 1998). It occurs farther south than any other Australian chelid with a range that extends from Cape York Peninsula to eastern Victoria, including inland areas throughout the Murray-Darling Basin to Lake Alexandrina (Legler _et al._, 1993). Smaller disjunct populations have been reported from areas throughout Victoria and South Australia (Beck, 1991).

It undertakes overland migration, particularly following rain (Goode, 1967; Dalem, 1998) and is one of the most physiologically suited for such activity, due to its high desiccation
tolerance. For example, it possesses a much lower rate of water evaporation, under desiccating conditions, than does *E. macquarii* (Chessman, 1984).

The short-necked, sidenecked turtles of the genus *Emydura*, are semi-aquatic and restricted to the Australian mainland and New Guinea. Absence of turtles from Tasmania has been linked to low temperatures (Beck, 1991). One factor affecting current turtle distribution is the deliberate relocation of individuals to new habitat by human intervention. For example, *C. longicollis* were translocated from the Swan River to effluent dams of South Australia in the 1920's (Beck, 1991). Although only small number of individuals may have been transferred, numbers have expanded to colonies of in excess of five hundred individuals (Stott, 1988).

Within the ‘*macquarii*’ group are the Murray Darling River turtle (*E. m. macquarii*), Brisbane River turtle (*Emydura macquarii signata*), Clarence River turtle (*Emydura macquarii binjing*), Macleay Hastings River turtle (*Emydura macquarii dharra*), Hunter River turtle (*Emydura macquarii gunabarra*) and *Emydura macquarii dharruk* (Sydney basin turtle: Cann 1998). *Emydura macquarii dharruk* was described by Cann (1998) as possessing a head and back with a light band running back from the angle of the mouth, extending along the neck. This species is known from several locations in the Sydney basin, including a range that encompasses the Upper Parramatta River Catchment, the Nepean-Hawkesbury, and Lane Cove rivers. It is sufficiently taxonomically different to indicate a distinctive race, making it the southern most of this group (Cann, 1998).

*Emydura macquarii* were thought to prefer habitation of larger rivers, lagoons and waterholes than *C. longicollis* (Parmenter, 1976). However, both *C. longicollis* and *E. macquarii* have
been found in ephemeral aquatic habitats. This is probably due to their high productivity, particularly after complete drying and refilling of the water body (Crome, 1986). Species of turtles exploiting the greater production of such ephemeral waters, must also overcome the problem of periodic, and often unpredictable, habitat loss (Burbidge, 1981).

Chessman (1984), found no *E. macquarii* where water depth was less than two metres. He concluded that, despite sometimes being caught in the same net, *C. longicollis* and *E. macquarii* had different microhabitat preferences. He further suggested that the two species partition habitats vertically within water bodies. Since *E. m. dharruk* is a sub-species of *E. macquarii*, in the absence of data, it is hypothesised that *C. longicollis* and *E. m. dharruk* have similar niche separation.

### 3.1.2 Population structure and gender differentiation

The proportion of juveniles within a population of a particular species, for example *C. longicollis*, may vary from one area to another or from one habitat to another (Chessman, 1978). For example, the proportion of juveniles at Laureldale Research Station (Armidale, N.S.W.) was observed to approximate 25% of the population (Parmenter, 1976). In Gippsland farm dams, they represented 52% of the overall population, while in nearby lagoons only 14% of the catch was juveniles (Chessman, 1978). The reason for this variability may include sampling techniques and capture methods that bias towards particular cohorts (Ream and Ream, 1966; Dalem, 1998). However, since Chessman (1978) utilised the same techniques in both farm dams and lagoons, the differences are assumed to reflect differences in population dynamics.
The sex ratio of *C. longicollis* has been shown to vary among different water bodies. It was observed to approximate 1:1 males : females in Ryan’s swamp (Jervis Bay, Kennett and Georges, 1990). In contrast, the sex ratio in water bodies in the Armidale region (Parmenter, 1976) and in Jervis Bay (Kennett and Georges, 1990) were observed to be skewed towards females, while Dalem (1998) found differences between dams in the Richmond (N.S.W.) area.

### 3.1.3 Growth and longevity

Growth rates for *C. longicollis* and *E. macquarii* can be related to food availability and temperature (Kennett and Georges, 1990). Most growth therefore occurs during spring and summer (Kennett and Georges, 1995). Both species are long-lived, taking as much as ten years to reach maturity in the wild. *Chelodina longicollis* is known to live up to 36 years in captivity (Goode, 1967) and may live as long as 150 years in the wild (Parmenter, 1985).

It has been shown that turtle growth is variable, even within a specific waterbody (Dalem, 1998; Parmenter, 1976). Both mean growth rate and percentage of turtles showing appreciable growth, have been linked to habitat conditions (Kennett and Georges, 1990). For example, turtles show greater growth during the wet season than they do in dry periods (Kennett and Georges, 1995). A comparison among systems has shown that *C. longicollis* growth and body condition is related to food availability (Kennett and Georges, 1990).

Any delay in growth as a result of being restricted to a drought-refuge has consequences for an individual’s reproductive capacity. Under these circumstances, occupation of ephemeral
waters affords much greater selective advantage than predicted from a comparison of production in ephemeral and permanent waters alone (Kennett and Georges, 1990).

3.1.4 Habitat Utilisation and Migration

Long range terrestrial movements are an important component of the population dynamics of many species of turtle (Gibbons, 1970; 1986). Turtles move overland seeking nesting sites or sexual partners, in response to seasonal cues or as habitat becomes unfavourable (Gibbons, 1986). This may occur as waters recede during droughts (Gibbons, 1983) or when there is a reduction in available foods (Parker, 1984).

Mature females of virtually all species must leave water to nest, thereby necessitating overland movements of a metre to more than 1km (Graham et al., 1996). Such studies have been undertaken for a number of terrestrial species in Africa (Stickel, 1950), North America (Bertram, 1979; Strang, 1983) and Australia (Stott, 1987; Dalem, 1998).

In trials conducted by Graham et al. (1996), the use of both olfactory and ocular cues for navigation were investigated in open field vs. arena (no view of horizon). He suggested the probable use of terrestrial landmarks for navigation, in trials conducted in a Y-tank, in which turtles showed highly significant directional preference when swamp and debris were used in alternative tank arms, confirming the use of olfactory cues. This was further supported by observations of gular-pumping during terrestrial movement. Since this species bears a pungent odour, individuals may use this to assist in navigation (Graham et al., 1996). Dalem (1998)
found that individuals tended to move in a straight line when moving out of water and many individuals returned to the dam of origin, indicating that they were navigating.

The freshwater turtle, *C. longicollis*, has frequently been observed undertaking terrestrial movement. Three types of movement have been identified: pond-to-pond, nesting, and movement involving the use of refuges. In pond-to-pond migration there was no evidence of a search pattern, indicating a navigational capacity (Stott, 1987). While this may involve familiarity with local landmarks (Bertram, 1979), individuals have been observed to navigate distances of over three hundred metres over two nights, through a stubble paddock with no visible landmarks (Stott, 1987). After nesting was completed animals tended to move in a straight line to the nearest pond, a pattern consistent with movement patterns observed for this species by Dalem (1998) and for *Chrysemys picta* (Gibbons, 1968).

Non-nesting movement of immature and adult turtles has also been observed. These movements have been attributed to factors such as habitat loss and prolonged drought (Gibbons, 1983), food shortage (Goode, 1967) and rainfall (Parmenter, 1976). Dalem (1998) observed that over time larger turtles moved from a dam that he was monitoring and smaller animals moved into the dam, such that after one year although the population size remained the same the size distribution of turtles had changed significantly. He hypothesized that an increase in emergent vegetation over time had changed the turtle habitat to favour smaller sized turtles.

Stott (1987) observed that the emergence of *C. longicollis* from man-made ponds was directional and was polarised in the direction of other nearby ponds. Such studies of diurnal
movements for *C. longicollis* contradict reports finding no such movement (Chessman, 1978; Dalem, 1998). For example, while Dalem (1998) observed that turtles navigated, they did not necessarily move towards the closest waterbody. It has also been observed that under sunny conditions in both open field and arena trials, *C. longicollis* showed significant non-random movement. Despite this, mean compass bearings of two experimental groups of *C. longicollis* differed by 42°. It was proposed that this difference was due to orientation by landmarks in the open field, not available for navigation in the arena (Graham *et al.*, 1996).

To orient using the sun as a compass, an animal needs a view of the sun and an internal biological clock in phase with local time to compensate for the movement of the sun across the sky (Ferguson, 1967). After acclimation of turtles to a six hour, phase-advanced light regime, it was demonstrated that there was a clockwise shift in their angle of orientation which was not significantly different from the expected 90°. This implicated a biological clock component in *C. longicollis* sun-compass orientation (Graham *et al.*, 1996).

Stott (1987) assessed the ability of hatchling *C. longicollis* from natural nests to find water. Control hatchlings (nasal irrigated with distilled water and released downwind at night) all moved in the direction of a nearby pond, whereas experimental animals (nasal irrigation with the local anesthetic amethocaine) did not orientate correctly. This indicated that there was an olfactory component to their navigation.

It has also been noted that *C. longicollis* disperse to ephemeral swamps with rainfall where growth is rapid, animals improve condition and reproductive output is enhanced, compared with individuals that inhabit permanent waters (Kennett and Georges, 1990). The propensity
of this species for overland migration following rain may be explained by the advantage of colonising highly productive waters. This may be reinforced by the severe disadvantage of remaining in permanent waters that have the potential to become overcrowded drought refuges (Kennett and Georges, 1990).

Any delay in growth as a result of being restricted to a drought-refuge has consequences for an individual’s reproductive capacity. Under these circumstances, occupation of ephemeral waters affords much greater selective advantage than predicted from a comparison of production in ephemeral and permanent waters alone (Kennett and Georges, 1990).

### 3.1.5 Diet

Diet of many turtle species is varied and dependent upon habitat (Chessman, 1984). While species such as *C. longicollis* are predominantly carnivorous (Legler, 1976; Burbidge, 1981; Cann, 1998), *Elseya dentata* and *Emydura* spp. are omnivores (Legler, 1976; Chessman, 1978; Georges, 1982).

The limitation of stomach content analysis in feeding analysis is that it does not address differential rates of digestion and passage of food through the gut (Spencer, 1995). This can result in a biased estimate of the relative abundance of food types consumed. As a result, the importance of foods with a relatively slow rate of passage will be overestimated (Georges et al., 1986). For example, turtles collected in winter (June-August), have been shown to possess almost no material in their stomachs (Chessman, 1984). Chessman (1984) observed the average stomach content volume varied between seasons, with summer (~1.7+/−4.5ml), spring (~2.1+/−3.1ml), winter (0.004+/−0.002ml) and autumn 1.1+/−2.7ml.
Spencer (1995) studied *E. macquarii* diet and found filamentous algae comprised 61% of stomach content. The turtles rarely feed on motile prey, but rather selected carrion from the lagoon bottom and terrestrial insects (*Diptera, Hymenoptera* and *Coleoptera*) trapped on the surface of the water. Digestive efficiency of *E. macquarii* (49%) on a herbivorous diet at 30°C was about half that of turtles on a carnivorous diet (91%), but food had longer transit time through the gut. The study also demonstrated lower consumption rates and longer mean retention times in turtles fed plants compared to those fed fish. This was related to slower digestive processing of plant material. Rapid processing and higher consumption rates of fish by *E. macquarii* resulted in higher energy gains compared to turtles consuming plants (almost 100 times more energy at 30°C). The laboratory results suggest that fish carrion and aquatic and terrestrial invertebrates are probably essential dietary items of *E. macquarii* in the wild, because its metabolic requirements cannot be met from aquatic macrophytes alone.

*Chelodina longicollis* is an opportunistic feeder (Legler, 1976) and is able to exist in low nutrient, low biotically diverse waterbodies, such as dune lakes (Georges *et al.*, 1986). In contrast to *C. expansa*, a specialised motile prey feeder, *C. longicollis* is catholic in dietary approach. Examination of stomach contents has revealed more numerous, but smaller prey, than equivalent-sized *C. expansa* from similar localities. In addition to feeding on both small and medium-sized prey, including zooplankton, crustaceans and fishes, it also eats carrion (Chessman, 1984; 1988a). Such material includes the remains of *C. carpio* (Chessman, 1984).
From a survey of 105 *C. longicollis* individuals, Chessman (1984) observed that the most important food items were carrion and decapod crustaceans in rivers, while carrion and littoral-benthic invertebrates were most commonly taken in lakes and ponds containing fish. Planktonic crustaceans, for example *Cladocera sp.*, were most commonly consumed by turtles living in ponds and pools without fish.

In terms of foraging technique, *Emydura macquarii* secure their food through slow prowling of benthic levels (Legler, 1976). Food is ingested through inertial feeding movements, combined with the gape and suck feeding method. Sensory cues in locating edible items appears to be tactile, visual and olfactory. By contrast, *C. longicollis* combines foraging with a sit-and-wait ambush feeding strategy. All *Chelodina* species executed a strike at prey targets, which may be moving, and consume such food items via a gape and suck approach. Both strategies preclude chelid turtles from feeding out of water (Legler *et al.*, 1993).

Parmenter (1976) found that in the New England area of north-eastern N.S.W., the diet of *C. longicollis* consisted primarily of aquatic insects, especially of the orders Diptera, Ephemeroptera, Hemiptera, Odonata and Trichoptera. This contrasted with Chessman’s (1984) analysis of diet. He observed that <11% of the stomach contents of Murray Valley *C. longicollis* consisted of aquatic insects belonging to these orders. He also observed that *C. longicollis* consumed terrestrial invertebrates that accidentally fall into the water. Nektonic insects, and fish were also eaten. Reasons for these differences were considered to include the scarcity of aquatic insects and the variety of littoral benthic invertebrates in the Murray River and its backwaters, probably because of the lack of aquatic vegetation. In small ponds
and rain pools, colonising turtles are able to exploit the dense zooplankton populations that develop in the absence of fish (Chessman, 1984).

Georges et al. (1986) found that there were quantitative differences between the littoral components of turtle diet and the composition of the local littoral fauna, this could be explained by difference in accessibility and ‘noticeability’ among prey species. In other words, there was no evidence that _C. longicollis_ were selective, within the confines of carnivory.

Juvenile _C. longicollis_ eat relatively less carrion, and more littoral and benthic invertebrates, than adults (Kennett and Georges, 1990). Differences between waterbodies in the growth rates and body condition of turtles have also been shown to coincide with differences in food availability. In particular Chessman (1984) observed that carrion formed a large proportion of the diet of mature/adult _C. longicollis_, but was rarely consumed by juveniles. In addition, littoral benthic vertebrates (aquatic invertebrates other than crustaceans and nektonic insects) were eaten more widely by juveniles than adults. Adult males and females of equivalent size had broadly similar diets, although whole fish were only found in male diets.

### 3.1.6 Reproduction and seasonal behaviour

Two broad reproduction patterns can be identified among Australian chelidae. Species of the temperate zone commonly nest in spring and hatch in summer, while patterns of tropical zone chelids can be more variable. Higher temperature provides a greater freedom for tropical chelidae. This pattern is exhibited by _E. krefii_ (Georges, 1983), _E. macquarii_ (Chessman, 1978), _R. leukops_ (Legler, 1976) and _C. longicollis_ (Parmenter, 1976).
Male *C. longicollis* reach sexual maturity at around seven to eight years, and females at ten to eleven (Parmenter, 1976). However, sexual maturity of turtles is more closely correlated with size than age (Gibbons, 1982). Males produce sperm at sizes between 145-147mm, whereas the presence of oviductal eggs in females has been noted to occur at a size of 162-165mm (Kennett *et al.*, 1990).

Upon sexual maturity, overall growth rate slows, weight increases and the shell deepens. Despite individual variation, females tend to grow faster and ultimately attain a larger size than males (Georges *et al.*, 1993). Mating usually takes place in spring, soon after emergence from brumation. Sex determination is independent of incubation temperature (Georges, 1988; Thompson, 1988).

The nesting period may begin as early as August in warmer areas and as late as November in colder areas (Georges, 1982). Although in some areas reproduction may be restricted to a single clutch per season (Parmenter, 1976), multiple clutching does occur (Chessman, 1987). Some females, kept in captivity, have retained sperm for a period of four years after copulation. The percentage of fertile eggs, however, decreased with each clutch (Goode, 1967).

### 3.1.7 Brumation, Hibernation and Aestivation

When winter conditions are not severe, *C. longicollis* and *E. macquarii* will achieve a state of brumation. If not disturbed, they will remain motionless for months, however, if stimulated they may react and move, albeit at a reduced activity level (Beck, 1991).
*Chelodina longicollis* has been noted to brumate at temperatures as high as 8°C, withdrawing the head and limbs, and resting on the swamp bed for up to three months, until water temperatures rise (Parmenter, 1976). During summer, when shallow swamps are nearly dry and the temperature of the water may reach 40°C, turtles may increase water intake, leave the water and bury themselves in shaded leaf litter (Beck, 1991). Individual turtles remained motionless for weeks (Beck, 1991). Dalem (1998) also observed that turtles left the water and could remain immobile for long periods of time, but there was no indication that this was a result of low water levels or unacceptable temperatures.

In the Murray River Valley, *C. longicollis* activity has been observed to be dependent on water temperature and they were trapped in all months except June and July. Healthy individuals were observed surfacing in winter at water temperatures as low as 8.3°C, although no animals were captured at that time (Chessman, 1988b).

### 3.1.8 Predation

A number of species predate on turtles. The most common are large birds which may target animals during nesting when females are most vulnerable (Beck, 1991). Foxes (*Vulpes vulpes*) are also a common threat to both *C. longicollis* and *E. macquarii* (Thompson, 1983; Green, 1995). Worrell (1996) reported that hatchlings may be eaten by herons, cormorants, kookaburras, crocodiles, goannas and snakes. In addition, he identified predators of adult turtles as pelicans, jabirus, water rats, dingoes, foxes, eagles and hawks while Green (1995) suspected that water rats had preyed on hibernating turtles.
Loss of individuals at the egg and hatchling stage of the lifecycle, due to predation, is likely to be greatest (Chessman, 1978; Thompson, 1993). *Vulpes vulpes* has been noted within the upper Parramatta catchment as a potential predator to hatchlings (Laxton, 1997).

Adults tend to be more susceptible to predation during terrestrial movements, although despite their propensity for overland movement, mortality rates among adult *C. longicollis* have been estimated to be less than 2% a year (Parmenter, 1976).

Ehmann (1992) observed that *C. longicollis* hatchlings turned on their back in the presence of eels and some oriented their bright orange and black plastron towards fish that had approached them at the surface of the water. However, Dalem (1998) observed that both eels and turtles were generally captured in the same nets and avoidance of eels by larger turtles was not observed.

### 3.1.9 Parasites, disease and abnormalities

*Chelodina longicollis* can carry ectoparasites such as turbelarian worms and glossiphonid leeches in their auxillary, ingual and tail areas (Betts, 1995). *Sigmapera cincta* (trematode) and *Austrampilina elongata* (cestodarian) have also been reported as endoparasites. No apparent correlation has been observed between body condition and leech infestation (Parmenter, 1976; Betts, 1995) and blood parasites were not shown to influence health (Rosser, 1997). *Sychnocotyle khoło* (Aspidogastrea: *Aspidogastridae*) and the plagiorchiidan species, *Choanocotyle elegans* have also been described from the small intestine of *Emydura macquarii* (Rosser, 1997).
The most common injuries for freshwater turtles are missing or damaged eyes, partial mutilation or amputation of limbs, lacerations of the head or neck and fractures or cuts on the carapace. Most of this injury is a result of direct trauma, predation or disease. However, freshwater turtles are resilient and most damage observed in studies had already healed. Only a few individuals were weak and probably would have died in Parmenter’s (1976) study. Rosser (1997) reported on the impact of environmental sickness on individuals, while Dalem’s (1998) study found animals were healthy and few (5%) were even assessed as being thin.

3.2 Aim of the study

Despite some detailed studies of turtles (Parmenter, 1976; Chessman, 1988; Dalem, 1998) and fish (Ege, 1939; Raadik, 1991; Jellyman, 1996), there is a paucity of data on most non-commercial aquatic species. In the Sydney region, data are particularly limited. For example, there are few records available on the endemic aquatic fauna of the Upper Parramatta River Catchment area and the most recent study of eel populations (elvers) in the area was undertaken by Ege (1939). There are apparently no published data on the other large aquatic vertebrates of the area. In contrast, physiochemical data have been regularly collected for over a decade (Laxton, 1994; Laxton, 1997).

Although such data are useful to determine the water quality for human usage (Heath, 1987; Arthington, 1986), it does not provide for an indication of the ‘health’ of the waterways for resident aquatic vertebrates. Most previous studies that have incorporated vertebrate bioindicators have been restricted to macroinvertebrates (Arthington, 1986) and feral species.
(Bales, 1992), with little attempt to quantify what water health means for even the most common of non-commercial vertebrates.

The aim of the current study is to:

a) collect baseline data on the abundance and population structure of the large aquatic vertebrates contained within the Upper Parramatta River catchment, and

b) based on water quality data for these sites, investigate underlying patterns of vertebrate distribution.
CHAPTER IV: STUDY CATCHMENT AND SAMPLING SITES

4.1 Upper Parramatta River Catchment geography and demography

The Upper Parramatta Catchment is predominantly urbanised, encompasses some 80,000 properties and has a population in excess of 240,000. Historically it was one of the first rural areas settled in Australia, providing grazing and farmlands. However, the importance of agriculture has declined as the urban population has expanded (Anon., 1999).

Approximately one-third of the area of Parramatta, the second largest city in the Sydney Metropolitan Area, was flood prone: 2,316 homes were subject to flooding in a once in 100-year storm in 1989 when flood mitigation was restricted to one flood basin. Since that time, major flood mitigation works have been undertaken that have resulted in modification of the waterways, although Toongabbie Creek (Plate 4.1) and Darling Mills Creek (Plate 4.2) are still subject to rapid stormwater run-off (Anon., 1999). All sites used in this study have been subject to some modification (two were dammed for city water supplies, the other two have been flood mitigated: Figure 4.1).

The catchment area comprises 108km$^2$ of which the most substantial waterways are Darling Mills and Toongabbie creeks. The majority of land use zoning is urban (94%), comprised of 72% residential, 10% industrial and 12% infrastructure.

The highest point across the catchment is 190m, located at Castle Hill, with declining grade over the 8.5km land distance to Charles Street Weir which is at sea level. The underlying
geology of the catchment is Bringelly Shale of the Wianamatta group (Middle Triassic). Toongabbie Creek possesses soils that are generally fine-grained silts and clays, however, Darling Mills Creek sub-catchment contains soils that are generally coarser grained sands and sandy loams (Morrison, 1999). This affects areas of the catchment differently as the Darling Mills Creek system is typical of deeply incised sandstone terrain of the Hornsby Plateau, while Toongabbie Creek drains from and through flat shale-based terrain of the Cumberland Plain. The result is that there are billabongs and back-swamps scattered across the sub-catchment (Morrison, 1999).

4.2 Climatic data

The area has a temperate climate. The maximum temperature was from a high of 28.1°C in January 1998 to a low of 4.5°C in June 1998 (Figure 4.2).

Average rainfall for the catchment is 989.1mm (1352.5 mm of pan evaporation), with the highest being recorded in January through March, and a lesser peak in November and December (Figure 4.2 : Morrison, 1999).

During the study trends were similar to the longer term recordings, although periods of peak rainfall differed. During the study, total annual rainfall for the Catchment was 921.3mm. Highest monthly rainfall (556.2mm), occurred in June 1998 (Source : National Climate Centre, Bureau of Meteorology).
Figure 4.1 Sample stations of the Upper Parramatta River Catchment

(inset map of Australia, highlighting Parramatta)
Figure 4.2 Monthly maximum and minimum mean temperatures and rainfall between November 1997 and December 1998 in the Upper Parramatta River Catchment (Source: National Climate Centre, Bureau of Meteorology)
4.3 Environmental health and pollution in the catchment

An assessment of environmental health, using macroinvertebrates as bioindicators, was undertaken by Lee Lau and Associates (1997). The advantage of using such organisms to monitor water quality over physiochemical methods is that the organism diversity and relative abundance may provide a more effective account of ecosystem health (Morrison, 1999).

Toongabbie Creek was categorised as possessing poor macroinvertebrate diversity and pollutant sources were highly diffuse in the sub-catchment, compared to Darling Mills Creek, where macroinvertebrate diversity decreased downstream indicating a tendency towards point source impacts (Lee Lau et al., 1997).

A number of water quality studies have been undertaken and currently, Laxton (1990-present) samples monthly at night. His sampling sites include Toongabbie Creek (at Redbank Rd, Plate 4.1), Darling Mills Creek (behind Parramatta gaol, Plate 4.2), Lake Parramatta (at the overflow to Hunts Creek, Plate 4.3) and Marsden Weir (where it joins Parramatta River, Plate 4.4). Sampling sites used in the present study (1-4) coincided with sampling sites (5-8) of Laxton, 1997-present.

The data for water quality parameters sampled by site, for the four sites listed above, and corresponding with the sampling period for the study (November 1997 – December 1998) is found in Tables 4.1, 4.2, 4.4 and 4.6.
4.4 Study sites

4.4.1 Toongabbie Creek

Toongabbie Creek (Site 1, Plate 4.1 : 150°59′42″E, 33°48′16″S) is located 4.5km north-west of Parramatta on the Cumberland Plain. The creek’s catchment has been substantially modified over the last two hundred years, predominantly due to urban development (Clements, 1996). Average annual rainfall from the nearest meteorological station (Redbank Road) is recorded as 900mm per annum, with a mean maximum January temperature of 29°C and a July mean minimum of 4°C.

Toongabbie Creek is the most lotic system of the four sampling locations and drains the highly urbanised southwest area of the catchment. The creek flows through Triassic sedimentary rocks, dominated by quartz sandstone (Anon., 1999). Soils are predominantly from the Luddenham group. Weeds and elevated nutrient levels now pose a threat to the integrity of local plant species. This is partly due to fill that has been deposited along the banks of the creek (Clements, 1996). This material is usually composed of sandstone, clay and rubbish. Some of the resulting slopes are bare sandy rubble with exposed rubbish while others are weed infested (Mortensen, 1996).

A major management issue is flash flooding that can cause high levels of gross pollution and low levels of dissolved oxygen (Laxton, 1994). In addition, gross pollution impacts on the local environment, particularly after flash floods when there is a rapid reduction in water movement and billabongs filled with waste form (Anon., 1999).
Table 4.1 – Water quality data for Toongabbie Creek between November 1997 and December 1998 (source Laxton., 1999)

<table>
<thead>
<tr>
<th>Month</th>
<th>WT</th>
<th>Sal</th>
<th>pH</th>
<th>O²</th>
<th>Turb</th>
<th>SS</th>
<th>Nutr</th>
<th>FC</th>
<th>Chl-A</th>
</tr>
</thead>
<tbody>
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</tr>
<tr>
<td>Dec</td>
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<td>0.16</td>
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<td>2.2</td>
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</tr>
<tr>
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<td>900</td>
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</tr>
<tr>
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<td>0.44</td>
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<td>22.3</td>
<td>7.81</td>
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<tr>
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<td>150</td>
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<tr>
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<td>7.61</td>
<td>3.4</td>
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<td>105</td>
<td>9.64</td>
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<td>7.35</td>
<td>6.3</td>
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<td>47.9</td>
<td>5.61</td>
<td>900</td>
<td>5.61</td>
</tr>
<tr>
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<td>0.41</td>
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<td>38.5</td>
<td>12.8</td>
<td>19.62</td>
<td>500</td>
<td>19.62</td>
</tr>
</tbody>
</table>

WT = water temperature; O² = dissolved oxygen; SS = suspended solids; Nutr = nutrient levels; Sal = salinity; pH = pH; Turb = turbidity; Chl-A = Chlorophyll-A; FC = faecal coliform.
Antcliff Ecological Surveys (1993) recorded 32 fauna species from Toongabbie Creek’s environs including birds, mammals, reptiles and amphibians. Most of these species were aves (26). Mammals recorded were the pale-headed flying fox (*Pteroptus poliocephalus*) and common ringtail possum (*Pseudocheirus peregrinus*). A single species of reptile (*Lampropholis guichenoti*) and two amphibian species, Eastern common froglet (*Rainella signifera*) and Whistling Tree Frog (*Litoria verreauxi*) were recorded.

A more recent survey (Rawling, 1996) listed 12 reptile species, including the Woody Gecko (*Diplodactylus vittatus*), Bearded Dragon (*Pogona barbata*) and Red-bellied black snake (*Pseudechis porphyriacus*), while finding 9 frog species, including the Common Eastern Froglet (*Crinia signifera*), Brown-striped Frog (*Limnodynastes peronii*) and Green Tree Frog (*Litoria caerulea*).

Stannard (1998) recorded *C. longicollis* (Eastern long-necked turtle), *A. australis* (Short-finned eel), and *Tiliqua scincoides* (Eastern blue tongued lizard) as well as 36 endemic plant species and small, but significant, patches of remnant riparian vegetation, in addition to nine species of exotic trees and 18 species of grasses.

### 4.4.2 Darling Mills Creek

Darling Mills Creek (Site 2, Plate 2 : 150°59'51"E, 33°47'44"S) extends south from Castle Hill to North Parramatta, where it meets Toongabbie Creek to form the Parramatta River (Figure 4.1). It flows through Triassic sedimentary rocks, dominated by quartz sandstone. There are considerable floodplain deposits of sand on the bends of the creek, but little in the bed of the stream (Anon., 1996; Plate 4.2).
Plate 4.1 Sampling Site 1: Toongabbie Creek

Plate 4.2 Sampling Site 2: Darling Mills Creek
The Darling Mills creek catchment is one of the ten largest bushland areas within an urbanised catchment in the Greater Sydney Basin (Anon., 1996). In contrast to the relatively low numbers of other animal groups, Laxton (1994) recorded 42 species of birds at the site. He suggested that the limited diversity of aquatic species may be due to the constant flooding and high turbidity, or overall lotic nature of the waterway. Such flooding has led to extensive erosion of banks and modified aquatic habitat (Manidis Roberts Consultants, 1994).

Vegetation has been observed to be more mesic along the west and south facing slopes, with the east and north facing slopes supporting endemic forest and privet (L. sinense and L. lucidum) that were relatively more xeric. Endemic vegetation is dominant upstream of the basin wall of Darling Mills Creek, while exotic species dominate downstream (Manidis Roberts Consultants, 1994). Twenty-three fern species have also been recorded to be associated with the aquatic environment (Laxton, 1994).

While weed frequency was observed to be inversely related to the distance from the established suburbs of Parramatta City, native vegetation populations have also been impacted upon by the sewer line, located on the valley floor. Additional nutrient enrichment was observed to occur due to sewage overflow in times of high rainfall (Manidis Roberts Consultants, 1994).

In addition to the two privet species weeds, also recorded from the area are Ageratina adenophora (Crofton Weed), Cotaderia selloana (Pampas Grass) and Cestrum parqui (Green Cestrum : Anon., 1996).
Table 4.2 – Water quality data for Darling Mills Creek between November 1997 and December 1998 (source Laxton., 1999)

<table>
<thead>
<tr>
<th>Month</th>
<th>WT</th>
<th>Sal</th>
<th>pH</th>
<th>O²</th>
<th>Turb</th>
<th>SS</th>
<th>Nutr</th>
<th>FC</th>
<th>Chl-A</th>
</tr>
</thead>
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<td>8</td>
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<td>Dec</td>
<td>21.04</td>
<td>0.16</td>
<td>7.28</td>
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<td>5</td>
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<td>45.9</td>
<td>14.1</td>
<td>13.63</td>
<td>900</td>
<td>13.63</td>
</tr>
<tr>
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<td>26.02</td>
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<td>7.44</td>
<td>4.1</td>
<td>50.1</td>
<td>5.3</td>
<td>6.49</td>
<td>600</td>
<td>6.49</td>
</tr>
<tr>
<td>Mar</td>
<td>22.37</td>
<td>0.12</td>
<td>7.17</td>
<td>7.2</td>
<td>83.2</td>
<td>6.4</td>
<td>8.25</td>
<td>70</td>
<td>8.25</td>
</tr>
<tr>
<td>Apr</td>
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<td>500</td>
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</tr>
<tr>
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<td>430</td>
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</tr>
<tr>
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<td>0.25</td>
<td>8.21</td>
<td>8.8</td>
<td>85.8</td>
<td>14</td>
<td>5.20</td>
<td>200</td>
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</tr>
<tr>
<td>July</td>
<td>10.14</td>
<td>0.28</td>
<td>7.75</td>
<td>9.8</td>
<td>86.8</td>
<td>8.5</td>
<td>3.97</td>
<td>350</td>
<td>3.97</td>
</tr>
<tr>
<td>Aug</td>
<td>12.81</td>
<td>0.14</td>
<td>8.06</td>
<td>10.2</td>
<td>96.6</td>
<td>30.6</td>
<td>11.28</td>
<td>1000</td>
<td>11.28</td>
</tr>
<tr>
<td>Sept</td>
<td>14.89</td>
<td>0.20</td>
<td>7.40</td>
<td>8.9</td>
<td>87.8</td>
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<td>6.22</td>
<td>320</td>
<td>6.22</td>
</tr>
<tr>
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<td>7.53</td>
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<td>4.6</td>
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</tr>
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<td>7.36</td>
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<td>6.46</td>
<td>600</td>
<td>6.46</td>
</tr>
<tr>
<td>Dec</td>
<td>23.08</td>
<td>0.23</td>
<td>7.28</td>
<td>5.3</td>
<td>61.6</td>
<td>6.4</td>
<td>11.38</td>
<td>350</td>
<td>11.38</td>
</tr>
</tbody>
</table>

WT = water temperature; O² = dissolved oxygen; SS = suspended solids; Nutr = nutrient levels; Sal = salinity; pH = pH; Turb = turbidity; Chl-A = Chlorophyll-A; FC = faecal coliform.
Table 4.3 Species recorded from Darling Mills catchment (Laxton, 1994; Anon., 1996a)

<table>
<thead>
<tr>
<th>Fish</th>
<th>Insects (common names)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Anguilla reinhardtii (Long finned eel)</td>
<td>greater water boatman</td>
</tr>
<tr>
<td>Gambusia affinis (Mosquito fish – Exotic)</td>
<td>lesser water boatman</td>
</tr>
<tr>
<td>Gobiomorphus australis (Striped gudgeon)</td>
<td>large back swimmer</td>
</tr>
<tr>
<td>Frog</td>
<td>small back swimmer</td>
</tr>
<tr>
<td>Uperoleia tyleri (Tylers toadlet)</td>
<td>mayfly larva</td>
</tr>
<tr>
<td>Aves</td>
<td>dragonfly larva</td>
</tr>
<tr>
<td>Streptopelia chinesis (Spotted turtle dove)</td>
<td>mole cricket</td>
</tr>
<tr>
<td>Picnonotus jocosus (Red whiskered bulbul)</td>
<td>fishing spider</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Mammals</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pseudocheirus peregrinus</td>
</tr>
<tr>
<td>(Ring tailed possum)</td>
</tr>
<tr>
<td>Trichosurus vulpecula (Brush tailed possum)</td>
</tr>
<tr>
<td>Petauroides volans (Greater glider)</td>
</tr>
<tr>
<td>Hydromys chrysogaster (Water rat)</td>
</tr>
<tr>
<td>Tachyglossus aculaetos (Short beaked</td>
</tr>
<tr>
<td>Echidna)</td>
</tr>
<tr>
<td>Perameles nasuta (Long nosed bandicoot)</td>
</tr>
<tr>
<td>Rattus fuscipes (Bush rat)</td>
</tr>
<tr>
<td>Canis familiaris (Domestic dog)</td>
</tr>
<tr>
<td>Felis catus (Cat)</td>
</tr>
<tr>
<td>Canis vulpes (Fox)</td>
</tr>
<tr>
<td>Vulpes vulpes (Red fox)</td>
</tr>
</tbody>
</table>
4.4.3 Lake Parramatta

Lake Parramatta (Site 3, Plate 4.3: 151°1'10"E, 33°47'55"S) is located 2.2km from the city's business district. It was Sydney's first wildlife refuge and, in the late 1700s, one of the first sources of dammed freshwater for the colony of New South Wales (Anon., 1994). The lake and its environs are now a reserve. The catchment encompasses 93.2ha with 10.5ha being water surface. Situated on the edge of the Hornsby Plateau, abutting the Cumberland basin, its topography is one of deeply dissected valleys and rocky outcrops (Anon., 1999).

The lake has flooded the main Hunts Creek Valley and left steep side-slopes. Some parts of the eastern ridge merge with the Wianamatta sandstones and shales. These soils are strongly acidic with low water retention, low soil fertility and now are impacted by urban runoff. The prevalent soil type is Gymea, with some occurrences of Lucas Heights and Hawkesbury Origin found in the northern and western regions (Anon., 1999).

The lakes' waters are now frequently considered unhealthy for aquatic life. This is due to high nutrient concentrations in the sediments stirred by stormwater influx. High bacterial levels have also been recorded and these were assumed to be caused by sewage pollution and possibly waterfowl (Laxton, 1994). The site is still used by the local community for recreation. However although a popular swimming spot in the early 1930's, the lake is no longer suitable for such recreational activities but secondary recreation, such as kayaking and canoeing, are still permitted (Plate 4.3). Such activities are not possible at all other sampling sites including Darling Mills Creek (Plate 4.4).
Table 4.4 – Water quality data for Lake Parramatta between November 1997 and December 1998 (source Laxton., 1999)

<table>
<thead>
<tr>
<th>Month</th>
<th>WT</th>
<th>Sal</th>
<th>pH</th>
<th>O²</th>
<th>Turb</th>
<th>SS</th>
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<td>7.23</td>
<td>6.9</td>
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</tr>
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<td>11.46</td>
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</tr>
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<td>3.53</td>
<td>700</td>
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<td>18.08</td>
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<td>93.8</td>
<td>4.2</td>
<td>5.69</td>
<td>400</td>
<td>5.69</td>
</tr>
<tr>
<td>Nov</td>
<td>21.20</td>
<td>0.24</td>
<td>7.54</td>
<td>7.5</td>
<td>84.8</td>
<td>3.2</td>
<td>5.13</td>
<td>900</td>
<td>5.13</td>
</tr>
<tr>
<td>Dec</td>
<td>25.22</td>
<td>0.23</td>
<td>7.76</td>
<td>8.0</td>
<td>96.9</td>
<td>1</td>
<td>8.56</td>
<td>600</td>
<td>8.56</td>
</tr>
</tbody>
</table>

WT = water temperature; O² = dissolved oxygen; SS = suspended solids; Nutr = nutrient levels; Sal = salinity; pH = pH; Turb = turbidity; Chl-A = Chlorophyll-A; FC = faecal coliform.
Mean annual rainfall is 935mm, with wet summers and drier winters. Mean daily temperatures range between 6.1 to 28°C (Anon., 1999).

Five plant populations, four endemic and one cultural, have been identified in the Reserve.

These are:

- *Open forest*, categorised by *Eucalyptus* spp., and in disturbed areas exotics such as *Ligustrum* spp. (small and large-leaf privets);
- *Riparian forest*, categorised by high levels of organic matter that contribute to higher levels of soil moisture and available nutrients;
- *Woodland* areas;
- *Cleared* and urbanized areas; and
- *Wetland* sections of Lake Parramatta.

Aquatic weeds include *Salvinia molesta*, that may cover much of the water’s surface (Laxton, 1997). Under appropriate conditions, particularly during hot dry spells, *S. molesta* can double the surface area it covers in two days. The high level of nutrients entering from the surrounding catchment and the large area of still water in the lake, provide appropriate habitat for this exotic species (Anon., 1999).

There has been no complete study of frogs or reptiles from Lake Parramatta Reserve. While frogs are present, they are likely to have been affected by herbicide application to aquatic
Plate 4.3 Sampling Site 3: Lake Parramatta

Plate 4.4 Sampling Site 4: Marsden Weir/ Parramatta Park
weeds, pollution from stormwater runoff and ingestion of poisons. However, there are numerous bird species and at least four reptile species.

Table 4.5 Species identified from previous Lake Parramatta studies (Source: Anon., 1999b; Laxton 1997)

<table>
<thead>
<tr>
<th>Chelodina longicollis</th>
<th>Gambusia spp.</th>
</tr>
</thead>
<tbody>
<tr>
<td>(Eastern snake necked turtle)</td>
<td>(Mosquito fish)</td>
</tr>
<tr>
<td><em>Tiliqua scincoides</em></td>
<td><em>A. australis</em></td>
</tr>
<tr>
<td>(Eastern blue-tongued lizard)</td>
<td>(Short-finned eel)</td>
</tr>
<tr>
<td><em>Lampropholis guichenoti</em></td>
<td><em>C. carpio</em></td>
</tr>
<tr>
<td>(Garden skink)</td>
<td>(Common carp)</td>
</tr>
<tr>
<td><em>Ctenotus robustus</em></td>
<td><em>Vulpes vulpes</em></td>
</tr>
<tr>
<td>(Striped skink)</td>
<td>(Red fox)</td>
</tr>
</tbody>
</table>

**4.4.4 Marsden Weir**

The geology of the Marsden Weir (Site 4, Plate 4.4: 151°0′24″E, 33°48′50″S) area consists of Triassic sedimentary rocks, dominated by quartz sandstone. The soils are predominantly of the Luddenham group, with limited flora in open forest areas (Laxton, 1994).

The weir was established as the colony’s first freshwater dam in 1818, supplying water to the newly created Governor’s quarters and parliament house, located in now what is known as Parramatta Park. No longer suitable for freshwater storage, the dam ceased use and a new dam was created in 1853 at Hunt’s Creek/Lake Parramatta (Plate 4.4). The site is at the interface between the freshwater and saline reaches of the Parramatta River. It is a looping waterbody that encircles Parramatta Park.
Table 4.6 – Water quality data for Marsden Weir between November 1997 and December 1998 (source Laxton., 1999)

<table>
<thead>
<tr>
<th>Month</th>
<th>WT</th>
<th>Sal</th>
<th>pH</th>
<th>O²</th>
<th>Turb</th>
<th>SS</th>
<th>Nutr</th>
<th>FC</th>
<th>Chl-A</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nov</td>
<td>20.86</td>
<td>0.34</td>
<td>7.60</td>
<td>5.1</td>
<td>10.5</td>
<td>6.2</td>
<td>9.33</td>
<td>0</td>
<td>14.10</td>
</tr>
<tr>
<td>Dec</td>
<td>22.73</td>
<td>0.26</td>
<td>7.54</td>
<td>4.2</td>
<td>5.5</td>
<td>5.4</td>
<td>11.77</td>
<td>330</td>
<td>8.67</td>
</tr>
<tr>
<td>Jan</td>
<td>23.07</td>
<td>0.11</td>
<td>7.55</td>
<td>4.4</td>
<td>51.1</td>
<td>13.9</td>
<td>9.17</td>
<td>500</td>
<td>9.17</td>
</tr>
<tr>
<td>Feb</td>
<td>26.89</td>
<td>0.13</td>
<td>7.89</td>
<td>7.4</td>
<td>93.2</td>
<td>11.5</td>
<td>5.77</td>
<td>300</td>
<td>5.77</td>
</tr>
<tr>
<td>Mar</td>
<td>23.41</td>
<td>0.21</td>
<td>7.43</td>
<td>7.6</td>
<td>89.4</td>
<td>16.1</td>
<td>12.80</td>
<td>16</td>
<td>12.80</td>
</tr>
<tr>
<td>Apr</td>
<td>18.70</td>
<td>0.26</td>
<td>8.01</td>
<td>6.7</td>
<td>71.5</td>
<td>19.1</td>
<td>13.53</td>
<td>300</td>
<td>13.53</td>
</tr>
<tr>
<td>May</td>
<td>16.76</td>
<td>0.29</td>
<td>7.90</td>
<td>8.2</td>
<td>84.8</td>
<td>25.8</td>
<td>9.00</td>
<td>10000</td>
<td>9.00</td>
</tr>
<tr>
<td>June</td>
<td>14.45</td>
<td>0.29</td>
<td>8.27</td>
<td>6.9</td>
<td>67.4</td>
<td>20.2</td>
<td>6.76</td>
<td>245</td>
<td>6.76</td>
</tr>
<tr>
<td>July</td>
<td>10.55</td>
<td>0.38</td>
<td>7.80</td>
<td>8.9</td>
<td>80</td>
<td>26.9</td>
<td>7.42</td>
<td>250</td>
<td>7.42</td>
</tr>
<tr>
<td>Aug</td>
<td>12.81</td>
<td>0.14</td>
<td>8.08</td>
<td>10.0</td>
<td>94.5</td>
<td>50.3</td>
<td>91.46</td>
<td>5000</td>
<td>91.46</td>
</tr>
<tr>
<td>Sept</td>
<td>16.17</td>
<td>0.20</td>
<td>7.69</td>
<td>9.7</td>
<td>98.3</td>
<td>17.5</td>
<td>4.66</td>
<td>200</td>
<td>4.66</td>
</tr>
<tr>
<td>Oct</td>
<td>17.93</td>
<td>0.55</td>
<td>7.95</td>
<td>7.5</td>
<td>79.4</td>
<td>11.6</td>
<td>9.50</td>
<td>110</td>
<td>9.50</td>
</tr>
<tr>
<td>Nov</td>
<td>20.48</td>
<td>0.24</td>
<td>7.39</td>
<td>5.9</td>
<td>65.4</td>
<td>28.2</td>
<td>10.47</td>
<td>500</td>
<td>10.74</td>
</tr>
<tr>
<td>Dec</td>
<td>24.81</td>
<td>0.33</td>
<td>7.64</td>
<td>7.6</td>
<td>91.4</td>
<td>5.6</td>
<td>8.08</td>
<td>400</td>
<td>8.08</td>
</tr>
</tbody>
</table>

WT = water temperature; O² = dissolved oxygen; SS = suspended solids; Nutr = nutrient levels; Sal = salinity; pH = pH; Turb = turbidity; Chl-A = Chlorophyll-A; FC = faecal coliform.
The waterway is currently unsuitable for primary recreation, however, it is commonly used by the community for non-primary contact activities, such as picnicking, rowing and kayaking (Laxton, 1994).

Few flora or fauna data have been officially recorded from the area, but the dominant open forest includes eucalypt and ironbark, while along the sandy stretches of Marsden Weir’s banks, the dominant tree species are eucalypt and black wattle (*Acacia decurrens*).

Ege (1939) was the first to record *A. australis* in the area and introduced species, such as *Gambusia* spp. and *C. carpio*, have also been recorded (Laxton, 1994).
CHAPTER V. METHODOLOGY

5.1 Sampling strategy
As previously indicated, the study was undertaken at four sites within the Upper Parramatta River Catchment between November 1997 and December 1998. The four sites correspond with the four freshwater sampling sites (Sites 5-8) currently monitored by Laxton (1994, pers. comm.; Figure 4.1) The water quality monitoring data collected by (Laxton, 1997-1998, pers. comm.) were used in the analysis of data collected on large freshwater vertebrates of the area.

At each location, ten potential sampling sites were identified, five along each 500m section of bank, at 100m intervals. Specific sampling sites were then randomly chosen from among these points. At each of the four randomly chosen sites, four fyke nets (Net Sales, Pyrmont, Australia) were set for seven days each month and cleared daily. Sampling continued for 14 months, between November 1997 and December 1998.

5.2 Species sampled
Preliminary studies indicated that there were six species of large aquatic vertebrates (see Table 5.1) and these became the focus of the study. Animals were taken from the net each morning of sampling and turtles were placed temporarily in a hessian bag until processing, while eels were placed individually in a deep mail sack to await processing.
Table 5.1: Aquatic vertebrates sampled at four sites in the Upper Parramatta River Catchment between November 1997 and December 1998.

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>A. australis</strong> - shortfinned eel</td>
<td><strong>E. m. dharruk</strong> – Sydney region turtle</td>
</tr>
<tr>
<td><strong>A. reinhardtii</strong> - longfinned eel</td>
<td><strong>C. carpio</strong> - European carp</td>
</tr>
<tr>
<td><strong>C. longicollis</strong> - Eastern snake-necked turtle</td>
<td></td>
</tr>
<tr>
<td><strong>T. tandanus</strong> - Common Catfish</td>
<td></td>
</tr>
</tbody>
</table>

5.3 Turtle sampling

Upon capture, turtles were sexed: *C. longicollis* based on concavity or convexity of the posterior portion of the plastron (Chessman, 1978) and *E. m. dharruk* were distinguished by genitalia and tail length. Because straight line carapace length has been demonstrated to be a good indicative measure of turtles (Dalem, 1998) and has been previously used as the measure of freshwater turtle size (eg. Parmenter, 1976; Chessman, 1988; Dalem, 1998), it was used as the basis for measurement in this study. Maximum carapace length was measured (cf. Parmenter, 1976; Dalem, 1998). Measurements were obtained using vernier callipers (+/- 0.1mm). Annual growth rate for *C. longicollis* was calculated from capture-recapture data.

Body weight was determined with a portable balance (Sartorius PT1200, Selby Scientific Instruments: Sydney) or one of two Salter Super Samson balances (10g-2kg) and (2kg-10kg), for individuals that weighed more than the 1200g limit on the portable balance. Turtles were uniquely marked (cf. Dalem, 1998; Cagle, 1939) with a series of filed notches in the marginal scutes. Initial cuts were usually less than 4mm deep and 3mm across.
Age of maturity for turtle species have been previously determined for *E. macquarii* (Georges, 1982) and in the absence of more accurate data, it was assumed that since *E. m. dharruk* is a sub-species of *E. macquarii* and attained approximately the same size (Cann, 1998) that its age at maturity was equivalent. Male *E. m. dharruk* and *C. longicollis* were considered mature at >145mm carapace length and females at >165mm carapace length. Individuals <110mm carapace length are difficult to sex (Georges, 1982; Parmenter, 1976) and were therefore considered juveniles. Individuals between 110mm and 145/165mm (dependent upon sex) were categorised as sub-adult.

### 5.4 Fish sampling

Total length, snout length, dorsal and anal fin length (where applicable) were measured using vernier calipers (+/- 0.1mm) or a measuring tape, as appropriate. Weight was determined with spring balances, with 10g-2kg and 2kg-10kg scales used when necessary.

Individuals were then weighed, measured, marked appropriately to their species (Sections : 5.3 and 5.4) and visually assessed for abnormalities, prior to being returned to the water at point of capture, generally within fifteen minutes of the nets being initially checked. Catfish and carp were removed from the net and immediately measured and weighed, using calipers, hanging scales and portable balance and also returned to their point of capture.

As recommended by the New South Wales Fisheries, fish were uniquely marked with (Hallprint, Australia) numbered tags. The tags were applied by inserting the gun needle
(Hallprint, Australia) through the skin surface, forward of the dorsal fin, but posterior to the head and snout region (Jellyman, 1996).

5.5 Data Handling

A major aim of the study was to investigate the patterns of distribution with physiochemical water quality parameters and the six most common large aquatic vertebrates in the system. Multivariate analysis using the program S-Plus (for Windows), principal component biplots (Gabriel, 1971) and constrained correspondence analysis (ter Braak, 1986; 1987) were constructed to visualise the multivariate observations.

This particular usage demonstrates patterns of correlation and covariation between environmental factors, using graphed eigenvectors (ter Braak, 1986). For each environmental factor, the varying length of eigenvectors demonstrated the variability observed among, and between, factors. The diagrams also allow for the relationship among sample units and water quality, together with the relationship between individual sample units and variation/covariation among variables. For all other analysis, data were analysed and represented using graphs constructed in Statistica, SASS and Excel programs.

The format of the bi-plot incorporates eigenvectors, corresponding to water quality factors that are being compared to the species data. The length of the eigenvector for each value represents the degree of variability for that factor. The correlation of factors to one another is represented in the distance between eigenvectors: the closer they are, the more strongly the factors are correlated. The central point where all the eigenvectors join is the mean across all factors, so when species data are inserted, individual species can be charted according to their
relative distribution, above or below the mean, in relation to corresponding water quality data for that period.

In addition to correlation between water quality and capture numbers, a comparison of weight against length for all species was conducted to demonstrate the relationship between weight and length for all species in the study (Section 6: Figure 6.3-6.8).
CHAPTER VI - RESULTS

6.1 Species abundance

A total of 730 fish and turtles (831 captures) were caught between November 1997 and December 1998 (Figure 6.1). Total capture figures result from a sampling effort of 16 nets (4x4 sites) over 5 sampling days (6 netting days) per month, for a total of 14 months. This equated to 831 captures over 1120 sampling units of effort, yielding a rate of 0.74 catches per net/unit effort.

Data were collected on seven vertebrate species, including three freshwater turtle taxa (C. longicollis, E. m. dharruk and T. s. elegans) and four fish species (A. australis. A. reinhardtii, T. tandanus and C. carpio). Two of these species, T. scripta elegans and C. carpio, are exotic species. The most frequently captured endemic species was C. longicollis (284 individuals from 332 captures), while C. carpio was the most frequently encountered exotic species (113 individuals from 122 captures: Table 6.1).

Of the turtle species in the study, data were collected on E. m. dharruk (61 individuals from 70 captures) and one capture of T. s. elegans (Red eared slider turtle) was also recorded during the sampling period. Of the two eel species, A. australis was most frequently caught with 138 individuals (156 captured) compared to A. reinhardtii with 70 (80 captures) and 63 (70 captures) of the local T. tandanus.

Size and weight data, together with range and standard deviation for all species, across all sites are presented in Tables 6.1 and Table 6.2.
Figure 6.1 Total monthly captures between November 1997 to December 1998 in the Upper Parramatta River Catchment

- C. longicollis
- E. m. dharruk
- A. australis
- A. reinhardtii
- T. tandanus
- C. carpio
Table 6.1 Number of individuals (captures) caught at four sites between November 1997 and December 1998 in the Upper Parramatta River Catchment

<table>
<thead>
<tr>
<th>SITE</th>
<th>C. longicollis</th>
<th>E. m. dharruk</th>
<th>T. s. elegans</th>
<th>A. australis</th>
<th>A. reinhardtii</th>
<th>T. tandanus</th>
<th>C. carpio</th>
</tr>
</thead>
<tbody>
<tr>
<td>Toongabbie Creek</td>
<td>98 (114)</td>
<td>26 (28)</td>
<td>1 (1)</td>
<td>28 (30)</td>
<td>9 (11)</td>
<td>23 (26)</td>
<td>33 (35)</td>
</tr>
<tr>
<td>Lake Parramatta</td>
<td>97 (114)</td>
<td>11 (12)</td>
<td>0 (0)</td>
<td>33 (37)</td>
<td>10 (11)</td>
<td>13 (14)</td>
<td>23 (25)</td>
</tr>
<tr>
<td>Darling Mills Creek</td>
<td>25 (30)</td>
<td>9 (10)</td>
<td>0 (0)</td>
<td>46 (53)</td>
<td>34 (39)</td>
<td>9 (9)</td>
<td>23 (25)</td>
</tr>
<tr>
<td>Marsden Weir</td>
<td>64 (74)</td>
<td>15 (20)</td>
<td>0 (0)</td>
<td>31 (36)</td>
<td>17 (19)</td>
<td>18 (21)</td>
<td>34 (37)</td>
</tr>
<tr>
<td>Totals</td>
<td>284 (332)</td>
<td>61 (70)</td>
<td>1 (1)</td>
<td>138 (156)</td>
<td>70 (80)</td>
<td>63 (70)</td>
<td>113 (122)</td>
</tr>
</tbody>
</table>
Table 6.2 Population characteristics of large vertebrates caught between November 1997 and December 1998 in the Upper Parramatta River Catchment

<table>
<thead>
<tr>
<th>Species / Sites</th>
<th>A. australis</th>
<th>A. reinhardtii</th>
<th>T. tandanus</th>
<th>C. carpio</th>
<th>C. longicollis</th>
<th>E. m. dharruk</th>
<th>T. s. elegans</th>
</tr>
</thead>
<tbody>
<tr>
<td>number</td>
<td>156</td>
<td>80</td>
<td>70</td>
<td>122</td>
<td>284</td>
<td>61</td>
<td>1</td>
</tr>
<tr>
<td>mean weight (g)</td>
<td>1101</td>
<td>1500</td>
<td>727.5</td>
<td>1074.5</td>
<td>472.5</td>
<td>527.5</td>
<td>1080</td>
</tr>
<tr>
<td>standard deviation</td>
<td>205.8</td>
<td>110.3</td>
<td>174.5</td>
<td>212.5</td>
<td>240.8</td>
<td>195.5</td>
<td>n/a</td>
</tr>
<tr>
<td>range (g)</td>
<td>300-2400</td>
<td>1000-2400</td>
<td>350-1100</td>
<td>600-1850</td>
<td>60-1215</td>
<td>60-1050</td>
<td>n/a</td>
</tr>
<tr>
<td>mean length (mm)</td>
<td>958</td>
<td>1250</td>
<td>230.4</td>
<td>340.1</td>
<td>158.2</td>
<td>164.1</td>
<td>215</td>
</tr>
<tr>
<td>standard deviation</td>
<td>118.6</td>
<td>68.3</td>
<td>18.6</td>
<td>21.2</td>
<td>38.6</td>
<td>24.6</td>
<td>n/a</td>
</tr>
<tr>
<td>range (mm)</td>
<td>280-1750</td>
<td>1000-2100</td>
<td>140-280</td>
<td>240-480</td>
<td>78-255</td>
<td>40-228</td>
<td>n/a</td>
</tr>
</tbody>
</table>
With the exception of *T. s. elegans*, that was represented by a single capture from Toongabbie Creek, turtles and fish sampled were collected from all sites, however, their abundance varied among sites. Capture of turtles, *T. tandanus* and *C. carpio* were highest from Toongabbie Creek, however, fewest eels of both species, *A. australis* and *A. reinhardtii*, were caught at this site. In contrast, Darling Mills Creek possessed the greatest number of both eels and the lowest numbers of other vertebrates in the study (Table 6.1).

Significantly different numbers of individuals were caught at differing sites in the study. *Chelodina longicollis* and *E. m. dharruk* numbers were significantly different among sites ($\chi^2_{3, 0.001} = 50.28; \chi^2_{3, 0.02} = 11.33$) with Darling Mills Creek possessing the greatest variation of any site.

Of the two eel species, *A. australis* numbers were found to be not significant among sites ($\chi^2_{3, 0.05} = 5.48$), while *A. reinhardtii* was found in significantly different numbers among sites in the catchment ($\chi^2_{3, 0.001} = 22.91$). For *A. reinhardtii*, this was most notable at Darling Mills Creek, however, this site also possessed the highest numbers for both eel species.

Of the remaining fish species, the endemic *T. tandanus* was found in significantly different numbers between sites ($\chi^2_{3, 0.05} = 8.01$) while the feral *C. carpio* was not caught in significantly differing numbers among sites across the catchment ($\chi^2_{3, 0.05} = 4.38$).
6.2 Injuries/Deformities

Visible turtle injuries and/or deformities included shell defects and missing limbs, while fin and body damage were apparent on some fish. Injuries/deformities were observed on 8.5% (5.1 - 14.1%; n = 284) *C. longicollis*. Injuries were recorded at all sites, however the differences were not significant among sites ($\chi^2_{3, 0.05} = 7.63$; Table 6.3). A similar overall number of injuries/deformities (14.5%, n=61) were carried by *E. m. dharruk*, with the results of injuries observed at three of the four sites and these varied between 11% (Darling Mills Creek) and 33.3% (Marsden Weir; Table 6.3). Overall deformity levels were low and variation among sites was not significant ($\chi^2_{3, 0.05} = 6.55$).

Few eels carried signs of injuries/abnormalities (Table 6.3). Only 4.3% (n = 70) of *A. reinhardtii* were observed to have defects and none were recorded at two of the four sites. Differences in injury rates for *A. reinhardtii* were not significantly different among sites ($\chi^2_{3, 0.05} = 3.66$), while *A. australis* collected from three sites had injuries/abnormalities (range 7.2-9.1%, overall 6.5%, n=138), but these differences were not significant ($\chi^2_{3, 0.05} = 3.88$).

At two sites no *C. carpio* were observed to have injuries/abnormalities, while 5.4% of animals captured at Marsden Weir and 17.4% from Toongabbie Creek were damaged. These differences were significant ($\chi^2_{3, 0.05} = 8.60$).

Abnormalities were not significantly different ($\chi^2_{3, 0.05} = 2.15$) among the four populations of *T. tandanus* sampled and varied between 8.3 - 10% (Table 6.3).
<table>
<thead>
<tr>
<th>Sites</th>
<th>C. longicollis</th>
<th>E. m. dharwark</th>
<th>A. australis</th>
<th>A. reinhardtii</th>
</tr>
</thead>
<tbody>
<tr>
<td>Toongabbie Creek</td>
<td>5/98 (5.1%)</td>
<td>3/26 (11.5%)</td>
<td>0/11</td>
<td>8/97 (8.2%)</td>
</tr>
<tr>
<td>Lake</td>
<td>4/33 (11.9%)</td>
<td>3/33 (9.1%)</td>
<td>1/10 (10%)</td>
<td>0/23</td>
</tr>
<tr>
<td>Parramatta</td>
<td>2/25 (8.2%)</td>
<td>2/34 (5.9%)</td>
<td>1/9 (11.1%)</td>
<td>0/23</td>
</tr>
<tr>
<td>Darling Mills Creek</td>
<td>4/46 (8.7%)</td>
<td>1/9 (11.1%)</td>
<td>1/34 (5.4%)</td>
<td>0/17</td>
</tr>
<tr>
<td>Marsden</td>
<td>9/64 (14.1%)</td>
<td>5/15 (33.3%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weir</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
6.3 Species abundance and water quality

Site characteristics were demonstrated using ellipses to visualise the physiochemical and species data and provide an indication of the level of homogeneity among sites (Figure 6.2). Three of the sites possessed ellipses with a mean value located close to the overall mean of combined water quality factors. This indicates that Toongabbie Creek and Marsden Weir have a high degree of homogeneity, whereas because the ellipse for Lake Parramatta is least in overlap with other sites, it is least similar in terms of its water quality parameters and species abundance.

Generally greater numbers of all species were netted in warmer (November, December and January), than cooler (June, July) months. In the coolest month (June) no turtles were caught and only low numbers of eels (see Figure 6.1). Differences in total capture abundance for all species was significant among seasons ($\chi^2_{3,0.001} = 49.30$).

Of the species investigated, *E. m. dharruk* distribution was more commonly correlated with turbid waters and suspended solids. This species was also more tolerant of salinity, nutrients, dissolved oxygen, faecal coliforms and pH, but distribution was less dependent upon water temperature and chlorophyll-a than *C. longicollis*. In contrast, *C. longicollis* distribution was most strongly correlated with water temperature and chlorophyll-a, while other water quality parameters had less influence on their distribution (Figure 6.2).

*Anguilla reinhardtii* were most commonly found in areas that possessed relatively high levels of dissolved oxygen, while salinity and nutrients appeared to have little impact on
distribution. Fewer individuals were recorded in areas that possessed higher than average water temperature, turbidity and suspended solids (Figure 6.2).

*Anguilla australis* distribution was observed to be more dependent upon elevated levels of dissolved oxygen than *A. reinhardtii*. It was also recorded in areas that possessed relatively higher levels of salinity, turbidity, suspended solids, nutrients, faecal coliforms and pH. However, both eel species were less commonly recorded in areas that possessed lower water temperature and chlorophyll-a levels, than the other species investigated (Figure 6.2).

*Tandanus tandanus* was correlated most strongly with above mean salinity and was more prevalent in areas with relatively higher levels of turbidity, suspended solids, nutrients, dissolved oxygen, faecal coliforms and pH. It was less commonly collected in the presence of relatively high water temperature and elevated levels of chlorophyll-a (Figure 6.2).

*Cyprinus carpio* was found to be environmentally the most indiscriminant of the species investigated. Although more commonly encountered in areas with higher than mean chlorophyll-a and water temperature, the correlation was weaker than for any other species and appeared to have little influence on their abundance (Figure 6.2).

As expected, nutrients were closely correlated with water quality factors, such as turbidity and suspended solids. In contrast, faecal coliforms were more closely correlated with chlorophyll-a and water temperature. Salinity and pH may also influence dissolved oxygen (Figure 6.2).
Figure 6.2 Comparison of water quality factors and species abundance

(T=Toongabbie Ck  P=Lake Parramatta  D=Darling Mills Ck, M=Marsden Weir)
(n = 831, species = 6, water quality parameters = 9, outliers retained)
Lake Parramatta possessed fewer *E. m. dharruk* and above average numbers of *A. australis* and *A. reinhardtii*. This indicates that there were relatively fewer *E. m. dharruk* compared to the other sites and relatively more eels (Figure 6.2). The overall species mean was similar across the other three sites. This is demonstrated by the three ellipses being around, or close to, the mean overall values.

### 6.4 Relationship between vertebrate length and weight

Measurement of vertebrate length against weight was taken for all species to determine the extent, if any, of a linear relationship between these two factors.

Values by site for all species are represented in Figures 6.3 – 6.8. There was a strong linear relationship between weight and length for all species studied. $r^2$ range values varied from 0.8215 (*A. australis*: Figure 6.3), to 0.944 (*C. longicollis*: Figure 6.4).

The two eel species in the study possessed $r^2$ values of 0.8215 (*A. australis*) and 0.8585 (*A. reinhardtii*). The remaining fish species possessed $r^2$ values of 0.8675 (*T. tandanus*) and 0.9015 (*C. carpio*).

In terms of the turtle species in the study, *E. m. dharruk* possessed an $r^2$ value of 0.8755, while *C. longicollis* possessed the highest $r^2$ value of 0.944.
Figure 6.3 Relationship between weight and body length of *Anguilla australis* captured between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.8215$)

- Toongabbie Creek
- Lake Parramatta
- Darling Mills Creek
- Marsden Weir
Figure 6.4 Comparison of weight and length of *Chelodina longicollis* caught between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.944$).

- Toongabbie Creek
- Lake Parramatta
- Darling Mills Creek
- Marsden Weir
Figure 6.5 Relationship between weight and body length of *Anguilla reinhardtii* captured between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.8585$)

- ■ Toongabbie Creek
- □ Lake Parramatta
- ■ Darling Mills Creek
- □ Marsden Weir
Figure 6.6 Relationship between weight and body length of *Tandanus Tandanus* captured between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.8675$).

- Toongabbie Creek
- Lake Parramatta
- Darling Mills Creek
- Marsden Weir
Figure 6.7 Relationship between weight and body length of *Cyprinus carpio* captured between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.9015$).

- ■ Toongabbie Creek
- □ Lake Parramatta
- ■ Darling Mills Creek
- ■ Marsden Weir
Figure 6.8: Relationship between weight and body length of *Emydra macquarii dharruk* captured between November 1997 and December 1998 in the Upper Parramatta River Catchment ($r^2 = 0.8755$).

- Toongabbie Creek
- Lake Parramatta
- Darling Mills Creek
- Marsden Weir
6.5 Fish population profile

Of the four species of fish netted one, *C. carpio*, was feral and the other three were endemic. All species were caught at all sites and in all seasons (Figure 6.1). There were differences in the size distribution between eel species (*A. australis*: 280-1750mm; *A. reinhardtii*: 1010-2150mm).

Recapture rate did not vary greatly among fish species. There was a 13% overall recapture rate for *A. australis* (range = 11.1 – 14.6% among sites, n=138), compared with 14.3% for *A. reinhardtii* (range = 8.1 – 17.8% among sites, n=70), 8% (range = 2.6 – 14.4%, n=113) for *C. carpio* and 11.1% (4.9 - 16.1%, n=63) for *T. tandanus*.

*Anguilla australis* was the only eel species with any captures measuring <500mm, although most (84.7%) were between 501-1300mm in body length. In contrast, most *A. reinhardtii* captured were larger than *A. australis*; with 78.6% of captures between 901 and 1700mm. Only *A. reinhardtii* individuals >1700mm body length were caught (Table 6.4).

Capture numbers for *A. australis* and *A. reinhardtii* increased during summer months and subsequently declined in winter (Figure 6.1). *Anguilla australis* accounted for 62.9% of the summer eel catch (n=62) and 66.7% of winter catch (n=30), while during the same period *A. reinhardtii* accounted for 37.1% and 33.3% of the catch respectively. Seasonal variation was significantly different between seasons for both species ($\chi^2 = 29.50$; $\chi^2 = 22.91$).

In terms of the other two fish species in the study, both *T. tandanus* and *C. carpio* exhibited similar trends across sites. Both species were captured more frequently at Toongabbie Creek
than at any other site. Fewest of both species were caught at Darling Mills Creek, while low numbers of *T. tandanus* were also caught at Marsden Weir, the site where most *C. carpio* captures occurred.

Data for the size and weight of both species is presented in Table 6.5. *Tandanus tandanus* captured from Toongabbie Creek possessed the lowest mean body length (225.9mm) and the greatest variability, as well as the lightest individual (350g). More animals were captured from this site than any other.

In contrast, *C. carpio* were found in highest numbers at Marsden Weir, however, both the shortest individual (240mm) and largest range for captures by weight (range = 1250g) were found at Toongabbie Creek, the same result as *T. tandanus*.

Estimated annual growth for fish species varied, with *A. australis* possessing a growth range between 4.1mm and 26.3mm, \( n = 18 \). *Anguilla reinhardtii* possessed the largest results for growth with variation between 0.3mm and 45.2mm \( n = 10 \).

For the other two fish species in the study, values ranged from 0.2- 11.2mm \( n = 7 \) for *T. tandanus* and 1.3 – 12.3mm for *C. carpio* \( n = 9 \). In general, juvenile animals grew faster than adults, however recapture numbers for all four species were too low for valid statistical analysis.
Table 6.4 *Anguilla australis* and *Anguilla reinhardtii* length and weight data caught between November 1997 and December 1998 at four sites in the Upper Parramatta River Catchment

<table>
<thead>
<tr>
<th>Figures</th>
<th>Toongabbie Ck.</th>
<th>Darling Mills Ck.</th>
<th>Lake Parramatta</th>
<th>Marsden Weir</th>
</tr>
</thead>
<tbody>
<tr>
<td>n</td>
<td>28</td>
<td>46</td>
<td>33</td>
<td>31</td>
</tr>
<tr>
<td>mean length (mm)</td>
<td>975</td>
<td>1278.5</td>
<td>977.9</td>
<td>1255.1</td>
</tr>
<tr>
<td>sd</td>
<td>133.2</td>
<td>142.3</td>
<td>120.1</td>
<td>279.2</td>
</tr>
<tr>
<td>range</td>
<td>520-1380</td>
<td>1010-1770</td>
<td>470-1750</td>
<td>1170-2150</td>
</tr>
<tr>
<td>mean weight (g)</td>
<td>1100</td>
<td>1450</td>
<td>1240</td>
<td>1530</td>
</tr>
<tr>
<td>sd</td>
<td>245.3</td>
<td>213.3</td>
<td>232.8</td>
<td>242.8</td>
</tr>
<tr>
<td>range</td>
<td>91-1215</td>
<td>1100-2200</td>
<td>61-704.1</td>
<td>1100-2400</td>
</tr>
</tbody>
</table>
Table 6.5 *Tandanus tandanus* and *Cyprinus carpio* length and weight data caught between November 1997 and December 1998 at four sites in the Upper Parramatta River Catchment

<table>
<thead>
<tr>
<th>Figures</th>
<th>Toongabbie Ck.</th>
<th>Darling Mills Ck.</th>
<th>Lake Parramatta</th>
<th>Marsden Weir</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td><em>T. tandanus</em></td>
<td><em>C. carpio</em></td>
<td><em>T. tandanus</em></td>
<td><em>C. carpio</em></td>
</tr>
<tr>
<td><strong>n</strong></td>
<td>22</td>
<td>33</td>
<td>9</td>
<td>20</td>
</tr>
<tr>
<td><strong>mean length (mm)</strong></td>
<td>225.9</td>
<td>331.0</td>
<td>231.1</td>
<td>341.1</td>
</tr>
<tr>
<td><strong>sd</strong></td>
<td>13.2</td>
<td>16.4</td>
<td>13.1</td>
<td>19.3</td>
</tr>
<tr>
<td><strong>range</strong></td>
<td>140-260</td>
<td>240-480</td>
<td>150-270</td>
<td>270-470</td>
</tr>
<tr>
<td><strong>mean weight (g)</strong></td>
<td>717.5</td>
<td>1044.5</td>
<td>725.1</td>
<td>1104.1</td>
</tr>
<tr>
<td><strong>sd</strong></td>
<td>24.3</td>
<td>33.3</td>
<td>22.8</td>
<td>34.8</td>
</tr>
<tr>
<td><strong>range</strong></td>
<td>350-1100</td>
<td>600-1850</td>
<td>450-1050</td>
<td>800-1800</td>
</tr>
</tbody>
</table>
6.6 Turtle population profile

Three species of turtle, two endemic (332 *C. longicollis*, 70 *E. m. dharruk*) and one exotic (*T. scripta elegans*) were collected during sampling (Table 6.1). Both *C. longicollis* and *E. m. dharruk* were collected at all four sampling locations. The population structure for turtles is presented in Table 6.7. Seasonally, numbers for both *C. longicollis* and *E. m. dharruk* differed significantly ($\chi^2_{3, 0.05} = 15.10; \chi^2_{3, 0.05} = 22.50$) with higher numbers caught in warmer sampling months (December to February: Figure 6.1).

Juvenile and sub-adults for both *C. longicollis* and *E. m. dharruk* were captured in approximately the same proportions (Table 6.7). The sex ratio also did not differ significantly from a 1:1 male:female ratio ($\chi^2_{1, 0.05} = 3.10; \chi^2_{1, 0.05} = 3.24$; Table 6.1). Adult *C. longicollis* represented 74.4% of the sampled population (39.2% male, 35.2% female) and adult *E. m. dharruk* accounted for 75.7% of those caught (41.4% male, 34.3% female). Females of both species possessed a higher mean weight and carapace length than males (Table 6.7).

Male and female capture rate varied seasonally (Figures 6.9; 6.11). Female *C. longicollis* tended to be caught in larger numbers in warmer months (72 in summer) and the catch declined in winter (n = 13). Males made up a smaller proportion of the catch in spring and summer (35.3%, n = 181) but represented a larger proportion of the catch in winter (66.7%, n = 69: Figure 6.11).
Turtle recapture rate did not vary with recapture rates for *C. longciollis* of 16.9% (range = 14.3 - 18.1%, n = 284) and *E. m. dharruk* of 14.8% (range = 9.8 - 17.3%, n = 61 : Figure 6.10). The ratio of males to females was equivalent in the sub-adult range (110-145 mm), skewed towards males in the adult range (145-165mm) and skewed towards females as carapace length increased beyond 165mm. The catch of females >145mm was similar throughout the year, whereas among larger animals the capture rate of males declined (>165mm).

Estimated annual growth for *C. longciollis* varied between −2.1mm and 25.4mm (n = 48), while *E. m. dharruk* varied between 1.2mm and 36.1mm (n = 9). In general, juvenile and sub-adult animals grew faster than adults, however recapture numbers, particularly for *E. m. dharruk*, were too low for valid statistical analysis.
Figure 6.9 Monthly turtle catch by gender caught between November 1997 and December 1998 in the Upper Parramatta River Catchment

- male E. macquarii
- female E. macquarii
- male C. longicollis
- female C. longicollis
- male T. s. elegans
Table 6.6: Population profile of turtles caught between November 1997 and December 1998 in the Upper Parramatta River Catchment

<table>
<thead>
<tr>
<th>Species</th>
<th>Juvenile Male</th>
<th>Sub-adult Male</th>
<th>Sub-adult Female</th>
<th>Adult Male</th>
<th>Adult Female</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>C. longicollis</em></td>
<td>19 (5.7%)</td>
<td>19 (5.7%)</td>
<td>47 (14.2%)</td>
<td>130 (39.2%)</td>
<td>117 (35.2%)</td>
<td>332</td>
</tr>
<tr>
<td><em>E. m. dharruk</em></td>
<td>2 (2.9%)</td>
<td>3 (4.3%)</td>
<td>12 (17.1%)</td>
<td>29 (41.4%)</td>
<td>24 (34.3%)</td>
<td>70</td>
</tr>
<tr>
<td><em>T. elegans</em></td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>1</td>
</tr>
</tbody>
</table>

Table 6.7: Carapace length/weight of *C. longicollis* and *E. m. dharruk* collected between November 1997 and December 1998 in the Upper Parramatta River Catchment (sd=standard deviation).

<table>
<thead>
<tr>
<th></th>
<th><em>C. longicollis</em></th>
<th><em>C. longicollis</em></th>
<th><em>E. m. dharruk</em></th>
<th><em>E. m. dharruk</em></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Male</td>
<td>Female</td>
<td>Male</td>
<td>Female</td>
</tr>
<tr>
<td>mean (x) carapace length (mm)</td>
<td>157.0 (sd = 36.8)</td>
<td>158.6 (sd = 44.6)</td>
<td>165.0 (sd = 21.5)</td>
<td>169.8 (sd = 27.3)</td>
</tr>
<tr>
<td>weight (g)</td>
<td>466.5 (sd = 220.1)</td>
<td>479.1 (sd = 258.7)</td>
<td>512.8 (sd = 183.6)</td>
<td>551.4 (sd = 212.3)</td>
</tr>
<tr>
<td>number (n)</td>
<td>151</td>
<td>162</td>
<td>34</td>
<td>34</td>
</tr>
</tbody>
</table>
Figure 6.10 *Chelodina longicollis* population structure caught between November 1997 and December 1998 across four sites in the Upper Parramatta River Catchment

- Toongabbie Creek
- Darling Mills Creek
- Lake Parramatta
- Marsden Weir
Figure 6.11 Seasonal variation in *Chelodina longicollis* catch by gender between November 1997 and December 1998 in the Upper Parramatta River Catchment.
CHAPTER VII - DISCUSSION

7.1 Effectiveness of water quality parameters to predict distribution of large aquatic vertebrates

Historically, physical and chemical water quality parameters have been used as a basis for determining aquatic environmental health (e.g. Arthington, 1996; Heath, 1987; Harris, 1998) and the use of macroinvertebrates as bioindicators has become more widespread (Chessman, 1988). However, there have been few studies that have assessed the use of aquatic vertebrates as bioindicators. The main exception has been the use of introduced fish species (Bales, 1992; Harris, 1996).

The use of aquatic vertebrates was thought to provide potential as a bioindicator of water quality, due to the causal relationship between habitat, biological factors and species stressors at the community level. In this study, I compared common water quality parameters with vertebrate species abundance. It was observed that the water quality parameters used to determine aquatic health, did not necessarily correlate with the six large aquatic vertebrates of the Upper Parramatta River Catchment. The assumption that ‘good’ water quality, based on such physiochemical parameters, are the definitive determinant of aquatic health is, therefore, erroneous.

Although the full suite of water quality factors assessed did not broadly correlate with species abundance and population profile, specific parameters were good indicators of the presence of particular species. For example, both eel species (A. australis and A. reinhardtii) were most abundant in the presence of high levels of dissolved oxygen. They are, therefore, a
good potential indicator for $O^2$ levels. In contrast, *T. tandanus* was a strong indicator of salinity levels, while *C. carpio* (one of the feral species most often used as a bioindicator: Bales, 1992) was the most indiscriminant of all species studied and its distribution did not closely correlate with the water quality parameters commonly used to assess environmental health. This demonstrates that this species is of little use as a bioindicator of water quality (Figure 6.2).

The data presented here provides the first assessment of the recently described *E. m. dharruk*. Although *E. m. dharruk* were unrecorded in the literature until described by Cann (1998), viable populations were found at all four sites sampled. They were collected across all size cohorts and juvenile recruitment rates were similar to the endemic *C. longicollis* (Table 6.6).

Endemic turtles were demonstrated to provide useful information on different parameters to the endemic fish: *C. longicollis* abundance correlated with water temperature and chlorophyll-a while *E. m. dharruk* was a strong indicator for turbidity and suspended solids and, to a lesser extent, had the potential to provide information on salinity (Figure 6.2).

*Emydura macquarii*, of which *E. m. dharruk* is a sub-species, have been reported to have a preference for habitation of larger rivers and waterholes than *C. longicollis* (Parmenter, 1976; Chessman 1988), although Crome (1986) suggested that both were present in ephemeral, temporarily nutrient-rich environments. He reported that they were attracted to such areas, particularly after complete drying and refilling of water bodies (Crome, 1986).
Diel or water level separation within the water column has also been observed between two of the turtle species with *E. macquarii* preferring deeper waters than *C. longicollis* and subsequent surface basking to maintain body temperature (Chessman, 1984). Such niche separation may have impacted on catchability of *E. m. dharruk*. Fyke nets set at the water/land interface have been shown to effectively sample *C. longicollis* (Burgin et al., 1999) but no such analysis exists for their relative effectiveness for *E. macquarii* taxa. No conclusions can therefore be drawn on the relative abundance of the two turtle species. In the same terms, *C. carpio* a school fish, may be caught in greater abundance than the solitary natured *T. tandanus* (Merrick and Schimda, 1984), while niche separation has also been observed in both eel species in the study (Sloane, 1984).

In addition to vertebrates correlating with high levels of specific physiochemical attributes, they were also useful to indicate low levels of others. There was a negative correlation between abundance of specific species and individual water parameters. Both eel species were found to be indicators of low chlorophyll-a levels, with *A. reinhardtii* also correlated with low turbidity levels. Abundance of *C. longicollis* has the potential to indicate low pH and, as previously indicated, the distribution of *C. carpio* were uninformative. The assumption, therefore, that ‘improved environmental health’ necessarily means greater species abundance, is fallacious since some species thrive under ‘imperfect’ conditions.

Since the four species of fish and two turtle species (*T. s. elegans* was represented by a single catch) were captured at all sites and in all seasons, diversity as an issue proved uninformative. In addition, the capture rate, injury/deformity frequency, length to weight, and growth data all proved uninformative.
7.2 Potential influence of ferals on other vertebrates

The feral *C. carpio* was well established in all four waterways sampled and there were indications that there were breeding populations at all sites (i.e. individuals were present in a range of sizes from juvenile to adult). They provided little useful information of specific water quality parameters, however, since they can play a major in-stream role in the degradation of such ecosystems, monitoring their distribution and status provides an ‘early warning’ of the potential for further degradation of the local waterways, because of their destructive habits (see Section 2.1.3).

The capacity of *C. carpio* to cause increased turbidity may be detrimental to other species in the study, who, demonstrated a preference for less turbid waters. The exception, *E. m. dharruk*, which showed a preference for turbid waters. Monitoring change in the presence of increasing number of *C. carpio* stock, the dynamic of these ecosystems may rapidly change. Observations on endemic species abundance thus has the potential to provide a quick, economical and relatively simple method of assessment of specific parameters while providing information on the ‘health’ of the ecosystem for the resident endemic species.

Since there is a lack of correlation between *C. carpio* and specific water quality parameters, alteration of ecosystem characteristics to enhance water quality is least likely to impact on this feral species and will undoubtedly impact upon the endemic species first. This is supported, at least in part, by previous studies that have indicated that *C. carpio* is able to out-compete endemic species in regulated rivers, including the adjacent Hawkesbury – Nepean River (Gehrke *et al.*, 1999). The problems associated with the management of this
feral are exacerbated by their high fecundity (Merrick and Schmida, 1984). It is therefore likely that the only potential long-term management of this species is their eradication. One potential method is electrofishing. This technique is currently being trialed by Blacktown City Council (N.S.W.), which adjoins the Upper Parramatta River catchment. Other techniques include the introduction of pathogens which are also currently being trialed but unlikely to be available in the foreseeable future.

Although only a single T. s. elegans was captured, Burgin and Emerton (2000) have provided evidence that this species is breeding in at least one waterbody within the Sydney region. Their devastating impact elsewhere in the world indicates the potential for this species to become a major problem in the future. Wider spread surveying to establish status, establish management strategies and ongoing monitoring is highly recommended.

7.3 Health and recruitment of large aquatic vertebrates in the Upper Parramatta River Catchment

Although all four sampling sites failed to meet ANZEEC guidelines for sustainable aquatic ecosystems, viable populations (Section 6.1), exhibiting low injury/deformity levels (Section 6.3), were demonstrated for all vertebrates studied across the catchment. A total catch of 831, including 730 individuals, was recorded (Figure 6.1), which equated to 0.74 catches per net/unit effort. These catches, at least for turtles, were lower than previous studies (Dalem, 1998; Parmenter, 1976) although differing sampling methodology was used in Parmenter’s study.
Capture rates were of equivalent ratio, but lower in number for *T. tandanus* and *C. carpio* compared with Harris’s (1996) study, while results for eel captures were consistent with Raadik’s (1991) finding of higher capture numbers for *A. australis* and *A. reinhardtii* compared with *C. carpio* and *T. tandanus*.

In terms of recruitment, the four fish species studied possessed ranges that indicated recruitment to viable populations, however, the two commonly caught turtle species, *C. longicollis* and *E. m. dharruk*, possessed only 5.7% and 2.9% of juveniles (<110mm in carapace length) respectively. When compared against the cohort ranges used in other studies, such as Chessman (1978) (where juveniles were considered <140mm) the current study yielded a result of 9.8% and 7.1% which was slightly lower than Dalem’s (1998) study (10%) of *C. longicollis* in Richmond, N.S.W., but much lower than Parmenter’s (1976). He recorded 24.67% and 14% juveniles in his catch at Laureldale Research Station, Armidale, N.S.W and a nearby lagoon. This provides evidence that while viable numbers of turtles may exist across the catchment, recruitment is low.

When combining juvenile and sub-adult numbers combined, there were 25.6% of total captures across all sites for *C. longicollis* and 24.3% for *E. m. dharruk*. This compared similarly to Dalem’s (1998) study (26%), but was much lower than the 52% recorded for *C. longicollis* by Chessman (1978). This demonstrates a skewing towards larger animals in the urban areas and inidcates ongoing low recruitment which may have a longer term impact on populations.
The numbers of *E. m. dharruk* were highest in larger waterbodies (i.e. Toongabbie Creek and Lake Parramatta), while fewest were caught in Darling Mills Creek, which frequently is restricted to a series of billabongs, thereby demonstrating the same trends established by Chessman (1984) for *E. macquarii*.

### 7.4 Seasonal variation in distribution across the Upper Parramatta River Catchment

Across the four sampling sites of the current study, overall capture numbers increased between September/October for most species and peaked during summer (December to February: Figure 4.2). Capture numbers increased threefold for *A. australis* between September and October, doubled for *A. reinhardtii* during the same period and the catch rate of species remained at relatively high levels during summer. Captures doubled for *T. tandanus* between September/October and remained high between November and February, as did *E. m. dharruk*. The impacts of environmental conditions on abundance, despite sub-optimal physiochemical parameters, means that erroneous interpretations can be made when the ecology of the species that are being dealt with is unknown, or sampling occurs over a short period, thereby failing to provide a good indication of local populations.

The catch of both *A. australis* and *A. reinhardtii* was relatively low compared to previous studies and may be reflective of marginal habitat for these species induced by modification of the waterways, such that recruitment may be inhibited by weirs. Recapture rates for *A. australis* were 13% which fell within the range of previous studies (15% Helfman, 1984; 11% Vladykov, 1957; 18.5% Beumer, 1979), while *A. reinhardtii* recapture (14.3%) was lower than the 20% recorded by Jellyman (1996).
Since 1888, 29 dams of >7m height and an additional 52 smaller water supply structures have been built across urban Sydney, Wollongong and the Blue Mountains (N.S.W.). The current study confirmed that there were differences in abundance between regulated and unregulated rivers and this was also observed by Gehrke et al. (1999). *Anguilla reinhardtii* were more abundant in regulated reaches (Darling Mills Creek), in contrast to the feral *C. carpio* that were more common in lowland reaches (Lake Parramatta) as in the Hawkesbury-Nepean. While there is clear size dimorphism between *T. tandanus* and *C. carpio*, the range and comparative ratio between captures by length (shortest to longest) and weight (lightest to heaviest) were the same for both species (Table 6.5), showing similar recruitment for both species.

### 7.5 Predation, injury and deformities observed on specimens captured across the Upper Parramatta River Catchment

The fox *Vulpes vulpes*, which is known to be widespread throughout Sydney and has been recorded at two of the sampling sites (Darling Mills Creek and Lake Parramatta) was not directly observed during the study period. Previous research in the Murray River has indicated that this feral species is having a major impact on turtle nests (Thompson, 1983) and it has been recorded to have destroyed nests of *C. longicollis, E. m. dharruk* and *T. s. elegans* elsewhere in the Sydney area (Burgin and Emerton, 2000). However, since foxes are widespread throughout Eastern Australia, it does not account for the apparent disparity in recruitment and it is therefore assumed that other environmental impacts are at least partly responsible for the observed low recruitment.
Dalem's (1998) results seem consistent with the present study in which no examples of direct trauma on turtles were noted when individuals were caught with eels in over 224 netting days. Moreover, given the total capture/re-capture numbers for both turtle species (402 total, 332 C. longicollis and 70 E. m. dharruk) and eel species (236 total, 156 A. australis, 80 A. reinhardtii), meant that eels and turtles were more likely to be captured in the same net, than separately.

In the present study, overall injury and trauma levels for species were consistent between all sites with limited instances of trauma between eel captures. Apart from smaller sized motile food sources, A. reinhardtii are noted to attack and sometimes consume comparatively smaller sized A. australis (Merrick and Schmida, 1984). For this reason, in many aquacultural farming crops where both species are grown to adult size for human consumption, A. reinhardtii and A. australis are sorted by size to prevent such predation.

Within the current study however, there were only limited attacks between these species. From a total 236 captures and re-captures, on only one occasion was any A. australis actually attacked to the point of mortality and partially-consumed. This occurred however when three A. reinhardtii were caught in the same net as the single A. australis. Such attacks in the study were uncommon and thought to be due to A. reinhardtii being a sporadic feeder, as it sometimes may go without food for up to two months (Merrick and Schmida, 1984).

Injury and abnormalities in the fish species of the current study was mainly confined to physical injury and attack. While eels are known to possess a large number of internal diseases and parasites, only one individual A. australis was noted with what was thought to
be ‘red fin’ disease (*Aeromoniasis*), thought more common on larger eels (Egusa, 1979), while no other individuals possessed external signs of infection.

*Aguilla reinhardtii* injuries were the lowest of any of the fish species in the current study (4.3%). This species is an aggressive predator and possesses few, if any, predators. *Aguilla australis* possessed attack injuries at three of the four sites sampled during the study period, however overall injury levels were still low (6.5%). *Cyprinus carpio* were only found with injury at two of the four sites, however one site, Toongabbie Creek possessed abnormal injury levels (17%). *Tandanus tandanus* injuries were comparable with results for *C. carpio* (8%) but were more consistent between locations, with injury recorded at three of the four sites and a range of 8-3-10% across the remaining three sites.

The proportion of injured *C. longicollis* and *E. m. dharruk* in the present study was 8.5% and 14.5% respectively which was comparable to Dalem (1998) in which 8.8% of *C. longicollis* were found to be injured. Injuries were more common among juvenile and sub-adults with male sub-adults comprising higher injury numbers than sub-adult females. This is consistent with previous studies (Dalem, 1998; Chessman, 1978). The most common type of injury observed for both *C. longicollis* and *E. macquarii* was carapace damage, however, the second highest number of injuries reported for *E. m. dharruk* specimens were missing limbs, presumably caused by predation although scarring was not always detected.

### 7.6 Conclusions

Continued degradation of waterways has been hypothesised to be the most significant environmental threat to the viability of freshwater turtles (Cann, 1993). Algal blooms are
common in the local area, particularly at Marsden Weir and Lake Parramatta, the least turbid sites. Additionally, plants such as the South American weed *Salvinia molesta* can cover the surface of Lake Parramatta (as occurred in 1994) and delete the lake’s oxygen (Anon., 1999), in turn impacting on local aquatic species, especially eels.

The current study has provided baseline data for future monitoring studies to determine the impact of changing management regimes. It has also provided the first assessment of some species, for example, *E. m. dharruk* and *C. longicollis* in the area and has updated information on other species, such as *A. australis* and *A. reinhardtii*.

There are apparently healthy populations of most species in the area but indications are that at least some species are under threat. For example, turtle recruitment appears to be suppressed and since there is a paucity of data on most of the other species, there is insufficient information to draw conclusions.

A significant component of this study was the investigation of the correlation between water quality parameters and the abundance and general health of large vertebrate populations. It is concluded that these parameters, taken generically, are poor surrogates. While specific parameters provide valuable information for a particular species (eg., salinity for *T. tandanus*, dissolved oxygen for *A. australis* and *A. reinhardtii*), it is necessary to investigate more broadly to ensure that healthy populations of all the major species exist. Given that such biological assessment is simple and cost effective, a reassessment of the heavy reliance on physiochemical data alone should be undertaken.
The problems associated with using macroinvertebrates as bioindicators includes the difficulty of accurate identification, the time consuming nature of such studies, and the lack of knowledge of their taxonomy and lifecycles. Large vertebrates do not pose the same problems because of relative ease of identification, the cost effective nature of such research and the ease of catch, compared to macroinvertebrates.

Due to natural environmental fluctuations and changes in flow rates, physical and chemical parameters have limitations for analysis. Analyses of macro-invertebrates is time consuming but monitoring of large vertebrates provides quick and effective monitoring tools and should be better developed over time to supplement chemical analysis. Results from such analysis can better be articulated to the general public and/or community who can provide input and therefore raise awareness which helps both the local environment more broadly, and the aquatic environment specifically.

Residents surveyed by Parramatta Council area, which has jurisdiction over the waterbodies encompassed within the current study, identified healthy animal populations as a key component of local waterway restoration. The Upper Parramatta River Catchment Trust, along with the broader community identified specific water quality goals, together with strategies for achieving them. To date efforts have been targeted at improving water quality in an attempt to meet ANZEEC guidelines (Anon., 1998) without assessment of the status or requirements of the larger vertebrates inhabiting the system.

The target to attempt to meet the ANZEEC guidelines for a number of key areas may have a detrimental impact on the endemic fauna and advantage the feral species. For example,
higher dissolved oxygen levels could prove favourable for eels and *C. longicollis*, potentially detrimental to *T. tandanus* and have no direct impact on *C. carpio*. In reality, if the focus of such management was to impact on endemic species, it may further benefit *C. carpio* and with their tendency to increase turbidity by removing rooted vegetation and reworking the sediments (Lake, 1978; Roberts, 1995), effort may actually be counterproductive. There is therefore a need to ensure that on-going monitoring of vertebrates is maintained to provide an indication of the impact on these species.

7.7 Recommendations

As a result of this study the following recommendations are made:

- Vertebrate sampling should be implemented as part of on-going monitoring of waterways. In developing protocols such characteristics as differences in catchability at different times of the year should be addressed.

- In light of the impact of *T. s. elegans* in many countries overseas and the confirmation that they are capable of breeding in Sydney’s waterways and were identified in the Upper Parramatta, there should be an assessment of their current distribution and an investigation of the likely impact and management strategies developed for their early eradication.

- Integrated management strategies should also be developed and implemented as a matter of urgency for *C. carpio*. 
- Ongoing monitoring and an investigation of the apparent decline of turtles in the area and an assessment of the ecological reasons, as well as any physiochemical factors impacting on *C. longicollis* and *E. m. dharruk* should be undertaken.

- Continued monitoring of populations of the vulnerable *T. tandanus*.

- Broader assessment of the waterways of Sydney to gather baseline data and identify threats to endemic species more generally.

- Assessment of the impact of eradication techniques, based on a full suite of data.

- Improved techniques should be developed for conveying research outcomes to the public to encourage appropriate environmental restoration for healthy functioning ecosystems, incorporating current human recreational activities, both at primary (swimming) and secondary (boating/kayaking) levels.
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**O2** = Dissolved Oxygen  
**SS** = suspended solids  
**Nur** = nutrient levels  
**Sal** = salinity  
**pH** = pH  
**Turb** = turbidity  
**Chl-A** = Chlorophyll-A  
**FC** = Faecal Coliform