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Conservation Biology of the Golden Galaxias
(*Galaxias auratus*) (Pisces: Galaxiidae)

Scott Anthony Hardie

A thesis submitted for the degree of Doctor of Philosophy at the School of Zoology,
University of Tasmania, Hobart, Tasmania, Australia

2007

Declaration of Originality

This thesis does not contain any material which has been accepted for the award of any other degree or diploma in the University of Tasmania nor any other university or institution. The material this thesis contains is, to the best of my knowledge, original except where due acknowledgement is made.

Mr Scott A. Hardie

June 2007

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June 2007

Statement of Co-authorship

The following people and institutions contributed to the publication (or submission for publication) of some of the work undertaken as part of this thesis:

- Preface Candidate*† (96%), Leon A. Barmuta (2%), Robert W. G. White* (2%)
- Chapter 2. Candidate*† (82%), Jean E. Jackson† (6%), Leon A. Barmuta* (10%), Robert W. G. White* (2%)
- Chapter 3. Candidate* (82%), Leon A. Barmuta* (14%), Robert W. G. White* (4%)
- Chapter 4. Candidate*† (85%), Leon A. Barmuta* (13%), Robert W. G. White* (2%)
- Chapter 6. Candidate*† (85%), Leon A. Barmuta* (13%), Robert W. G. White* (2%)
- Chapter 7. Candidate*† (80%), Stephen B. Pyecroft‡ (10%), Leon A. Barmuta* (8%), Robert W. G. White* (2%)

**School of Zoology and Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Private Bag 5, Hobart, Tasmania 7001, Australia*

†Inland Fisheries Service, PO Box 288, Moonah, Tasmania 7009, Australia

‡Animal Health Laboratory, Department of Primary Industries and Water, PO Box 46, Kings Meadows, Tasmania 7249, Australia

L.A.B. supervised the administration of these studies, helped develop the key concepts, and assisted with data analyses for each paper. R.W.G.W. helped refine the content of these papers. J.E.J. and S.B.P. contributed to the development of the content of Chapters 2 and 7, respectively, and S.B.P. also provided laboratory assistance for Chapter 7.

We the undersigned agree with the above stated “proportion of work undertaken” for each of the above published (or submitted) peer-reviewed manuscripts contributing to this thesis:

Signed:

Dr Leon A. Barmuta

Dr Susan M. Jones

Supervisor

Head of School

School of Zoology & TAFI

School of Zoology

University of Tasmania

University of Tasmania

Date: June 2007

Summary

1. The golden galaxias (*Galaxias auratus* Johnston 1883) is a threatened, non-diadromous galaxiid that is endemic to lakes Crescent and Sorell in central Tasmania, Australia. Similar to the lacustrine habitats of other threatened galaxiids in this region, the Crescent-Sorell system has altered hydrology and contains alien fishes. The objective of this study was to examine crucial aspects of the biology and life history of *G. auratus* (including age, growth, population structure, reproductive biology and recruitment dynamics) in these impounded highland lakes that are likely to be influenced by hydrology and lake management. An investigation of population monitoring methods for this species was also conducted. This work aimed to assess the vulnerability of *G. auratus* populations, and gain insight into the ecological attributes of other threatened galaxiids, threats to this family and other similar small-sized lentic species.

2. Analysis of a 5-year, monthly record of population and spawning attributes of both populations, along with detailed environmental data, including water levels and water temperatures, showed water levels and the access they provide to spawning habitat strongly limit the reproduction and recruitment of *G. auratus* in Lake Crescent. Detailed life history attributes of this species support these conclusions and further illustrate the vulnerability of this species to hydrological manipulations.

3. Gonad development began in mid-summer and spawning was spread over late autumn – early spring, peaking in winter. Demersal adhesive eggs (~1.5 mm diameter) were found on cobble substrates (c. 20-250 mm diameter) in littoral areas (c. 0.2-0.6 m deep). Patterns in larval emergence and abundance were associated with the timing of inundation of spawning habitats. In Lake Crescent, seasonal abundances of larvae were strongly related to the

magnitude of water level rises during spawning and egg incubation (i.e. May-September).

Despite the occurrence of larvae in pelagic habitats in both lakes during winter, they did not grow until spring; thus, coupling of water level and water temperature regimes is important for recruitment. *Galaxias auratus* reached up to 240 mm in length and ~10 years of age; however, most grew to <130 mm and the 0+, 1+ and 2+ year classes dominated the age structure of both populations. There were more >2+ fish in Lake Crescent where predatory introduced salmonids were less abundant than in Lake Sorell. Growth of these larger fish was slower due to limited access to complex littoral habitats in Lake Crescent.

4. In a translocated population, fyke netting at night was the most effective sampling method owing to increased activity at night and cover-seeking behaviour by this species. In the lakes, monthly catches of *G. auratus* increased substantially during spawning, suggesting that knowledge of the reproductive biology of target species can aid population monitoring programs for other galaxiids.

5. Water level fluctuations play a key role in the life cycle of *G. auratus* which relies on access to complex littoral habitats for spawning, feeding and refuge. Seasonal hydrological cycles (i.e. rises during late autumn – winter) and a minimum water level of 802.20 m AHD in Lake Crescent during autumn (above which littoral areas of cobble substrate are inundated) are critical to *G. auratus* populations. Because many lacustrine galaxiids use littoral habitats during several life stages, alterations to water levels and seasonal hydrological regimes may impact on their populations by restricting access to these habitats at critical times. To assist the management of other threatened galaxiid species, further studies should identify habitats that limit populations based on species biology and examine ecological traits that provide resilience to major perturbations.

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Dedication

I hope the knowledge gained from this volume of work helps raise awareness of the uniqueness and imperilled status of small-sized inconspicuous freshwater fishes. I believe the naive and careless views expressed in the following quote* regarding European fishes still hold true today in several sectors; to the detriment of many species which are of little or no commercial or recreational value.

‘There is also a little fish called a Stickleback: a fish without scales, but hath his body fenced with several prickles. I know not where he dwells in winter, nor what he is good for in summer, but only to make sport for boys and women-anglers, and to feed other fish that be fish of prey, as Trouts in particular, who will bite at him as at a Pink, and better, if your hook be rightly baited with him, for he may be so baited, as his tail turning like the sail of a wind-mill will make him turn more quick than any Pink or Minnow can. For note, that the nimble turning of that or the Minnow is the perfection of Minnow-fishing.’

*Taken from modern day reprint ‘Walton, I. (1995). *The Complete Angler*. London: J. M. Dent.’. This is the fifth edition of this classic angling companion which was first printed in 1676. Izaak Walton was a seventeenth-century devotee of the art of angling, nostalgic royalist and friend of bishops.

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Preface: Threatened fishes of the world: *Galaxias auratus*

Johnston, 1883 (Galaxiidae)*

*Published as Hardie S.A., Barmuta L.A. & White R.W.G. (2004) in *Environmental Biology of Fishes* 71, 126.

Common name: Golden galaxias.

Conservation status: Rare – (Tasmanian *Threatened Species Protection Act 1995*); Endangered – (2003); Endangered – (Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*).

Identification: D 7-10, A 11-12, P 14-18, vertebral count 53-56 (McDowall & Frankenberg, 1981). Small scaleless salmoniform fish, maximum size: 240 mm TFL, 130 g (Hardie, 2003). Colouration: golden to olive-green on dorsal surface and sides, silvery-grey on ventral surface. Back and sides are covered with round to oval black spots (McDowall & Frankenberg, 1981) (Fig. 1).

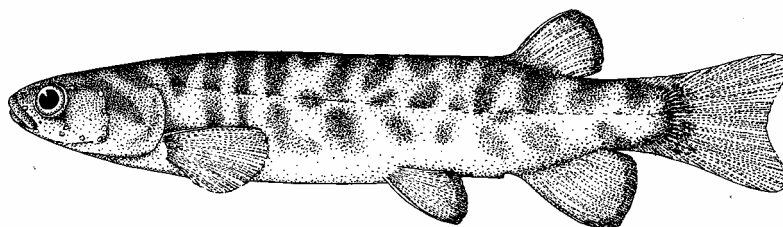


Fig. 1. Golden galaxias (*Galaxias auratus*). Drawing by Carol Kroger in Fulton (1990).

Distribution: Endemic to the interconnecting lakes Crescent and Sorell and their associated creeks and wetlands in the headwaters of the Clyde River catchment central Tasmania, Australia. Two translocated populations were established in farm dams in the Clyde River catchment, between 1996 and 1998. Currently, only four breeding populations exist including two natural and two translocated populations.

Abundance: Natural populations in lakes Crescent and Sorell are currently abundant, although relative abundance differs significantly between the two with the Lake Crescent population being an order of magnitude greater than the Lake Sorell population. The two translocated populations of golden galaxias are currently abundant (Hardie, 2003).

Habitat and ecology: A non-diadromous species preferring lentic waters. Adults frequently feed in the water column but tend to be benthic and prefer the shelter of rocky lake shores, in-lake macrophyte beds and wetland habitat. The adult diet consists of aquatic and terrestrial insects, small crustaceans, molluscs and cannibalism of eggs and juvenile fish is common (Hardie, 2003). Juveniles are pelagic and feed on zooplankton and small insect larvae (Frijlink, 1999).

Reproduction: Spawning takes place in late autumn – winter on rocky shores and possibly in wetland habitat when available (Hardie, 2003). Spawning occurs at approximately 4°C (range 2-7°C) and appears to be triggered by rising lake levels. Fecundity ranges from 1000 to 15 000 eggs. Fertilised eggs are ~1.5 mm in diameter, transparent and adhesive. Spawmed eggs are scattered over cobble substrate or aquatic vegetation at a depth of 200-600 mm. Fertilized eggs are thought to incubate for 30-45 days in the wild. Larval hatching peaks

during late winter – early spring. Newly hatched larvae are 5-7 mm in length and are pelagic until 4-5 months of age (40 mm TFL).

Threats: Low water levels in lakes Crescent and Sorell are the primary threat to natural populations of golden galaxias (Hardie, 2003). In Lake Crescent, low water levels can de-water rocky shorelines which are a critical habitat for spawning and refuge. Competition and predation from introduced fish is also a significant threat. Four introduced species currently inhabit lakes Crescent and Sorell (*Salmo trutta*, *Oncorhynchus mykiss*, *Cyprinus carpio* and *Galaxias maculatus*). Brown trout are known to predate heavily on golden galaxias in both lakes (Stuart-Smith, 2001).

Conservation action: The golden galaxias is protected under State legislation and may only be collected under permit. The findings of the recent work of Hardie (2003) have been incorporated into a water management planning process for lakes Crescent and Sorell and have also provided baseline data for future monitoring of populations. The translocated population of golden galaxias in the Jericho area has been formally reserved.

Conservation recommendations: The significance of adjacent wetlands and in-flowing creeks and their associated wetlands needs to be determined. Wild and translocated populations of golden galaxias should continue to be monitored on an annual basis.

Remarks: Water level management, including the allocation of water to maintain base ‘environmental’ levels in lakes Crescent and Sorell, is critical to the survival and health of natural populations of golden galaxias.

1. General Introduction

1.1. Conservation of freshwater fishes

Freshwater fishes are increasingly being considered threatened at regional (Bruton, 1990; Moyle, 1995; Fu *et al.*, 2003) and international (Cambray & Giorgio Bianco, 1998) levels. In order to maintain their associated fishery resources, aquatic biodiversity, and the health of the systems which they inhabit, freshwater fish conservation has been recognised as an important action globally (Lowe-McConnell, 1990; Cowx & Collares-Pereira, 2002). A broad range of threats, which are mostly linked to anthropogenic impacts and biological invasions, increase the vulnerability of fishes (Bruton, 1995). Additionally, rare species which have restricted distributions, occur infrequently or have low abundances, are often at greater risk of extinction (Economidis, 2002; Fagan *et al.*, 2002). Recent studies (Angermeier, 1995; Parent & Schriml, 1995; Duncan & Lockwood, 2001; Reynolds *et al.*, 2005) have tried to determine if phylogeny and life history traits of freshwater fishes correlate with extinction risk; however, their results have been largely inconclusive. Whilst some biological and ecological attributes, such as small size (Reynolds *et al.*, 2005) and diadromy (Angermeier, 1995), do increase the risk of extinction, identification of extrinsic factors which affect families or individual species is paramount to their conservation (Duncan & Lockwood, 2001).

Strategies to manage threatened species take many forms and include population monitoring, risk assessments, habitat restoration and reservation, captive propagation, and translocation of taxa. Techniques to monitor wildlife populations are needed to estimate abundance and guide decisions regarding their management (Hauser *et al.*, 2006). Additionally, knowledge of the biology and ecology of target species is essential to their conservation as these data underpin methods used to assess status (IUCN, 2003), identify threats and facilitate recovery. Recently, management of threatened taxa has shifted from the traditional single-species focus to multi-

species approaches due to the inclusion of increasing numbers of taxa on threatened species lists, and the economic and resource constraints of management agencies. Species-specific biological understanding is often limited under multi-species recovery plans in comparison to those for single species (Clark & Harvey, 2002). However, data for well studied species can provide insight into the biological attributes and key threats of other closely related taxa, which have yet to be examined.

Knowledge of population dynamics and relationships between species and their habitats is required to perform quantitative risk assessments (Lindenmayer & Burgman, 2005). Such data often exist prior to the decline of commercially exploited species, but for taxa like small-sized freshwater fishes which are of no commercial or recreational value, these data are usually scarce. The life histories of many fishes rely on the spatial and temporal alignment of access to certain habitats. Therefore, the lack of favourable habitats for different life stages can have significant consequences for fish populations (Naiman & Latterell, 2005). In modified environments, this variability may be the primary threat to populations. Thus, identification of biology-habitat linkages can aid conservation efforts (Rosenfeld & Hatfield, 2006).

1.2. Galaxiid fishes and the golden galaxias

Fish of the salmoniform family Galaxiidae are relatively small (adults generally <300 mm long), scaleless and often cryptic (McDowall & Frankenberg, 1981). They occupy freshwater and estuarine environments in mostly cool temperate regions on several land masses in the Southern Hemisphere, but are particularly prominent in the freshwater fish faunas of southern Australia and New Zealand (McDowall & Fulton, 1996; McDowall, 2000). Members of the family, of which there are around 50 species, exhibit both diadromous and non-diadromous

life histories. This enables them to occupy a diverse range of habitats. Whilst some migratory species contribute to regional whitebait fisheries (McDowall & Fulton, 1996; McDowall, 2000), most galaxiid species are not of recreational or commercial value and more than 50% of species are threatened (i.e. protected under legislation in their region(s) of occurrence or listed on conservation awareness lists) (McDowall, 2006).

Despite the broad distribution of the family, particularly some species (e.g. *Galaxias maculatus* Jenyns (Waters & Burridge, 1999)), many non-diadromous species have highly restricted distributions. Several galaxiid species only occur in a single river system or a few lentic waters. Similar to many small-sized endangered fishes, galaxiids face threats from exotic species, particularly those associated with introduced salmonids which have been well documented (Tilzey, 1976; Crowl *et al.*, 1992; Ault & White, 1994; McIntosh, 2000; McDowall, 2003). However, anthropogenic alterations of their habitats are also likely to have contributed to the decline of some galaxiids, and data regarding these impacts are limited (Hanchet, 1990; Eikaas & McIntosh, 2006). For example, impoundment can dramatically alter hydrological regimes and habitat landscapes in natural lentic waters that are occupied by lacustrine galaxiids. However, at present little is known of the roles of hydrological variables in the life histories of galaxiids or the strategies by which they use different habitats.

The island of Tasmania, southern Australia, has a diverse galaxiid fauna, with 16 species, of which 11 are endemic. Seven endemic non-migratory species occur in discrete populations in highland lakes and lagoons in the central highland district known as the Tasmanian Central Plateau (TCP). Six of these species occur in lakes that have been impounded during the past century for hydro-electric power generation and municipal and agricultural water storage purposes. Despite conservation efforts under multi-species recovery plans (Crook & Sanger, 1997; Threatened Species Section, 2006), some Tasmanian lacustrine species (e.g. *Galaxias*

pedderensis Frankenberg (Hamr, 1995) and *Paragalaxias mesotes* McDowall and Fulton (Threatened Species Section, 2006)) in impounded lakes have recently undergone significant declines. Whilst there is conjecture surrounding the reasons for this, a lack of biological and ecological data and effective population monitoring methods for these and other local species make management and recovery of their populations difficult.

The golden galaxias, *Galaxias auratus* Johnston, is a typical example of a localised lacustrine galaxiid (Fig. 1). This species is endemic to the interconnected Lake Crescent and Lake Sorell (Fig. 2) in the south-east of the TCP in the upper reaches of the Clyde River catchment (McDowall & Fulton, 1996). These shallow lakes (mean depths <3.5 m) are very similar chemically and physically and have been impounded for agricultural and municipal water storage purposes. Since being described over 120 years ago (Johnston, 1883), the biology of *G. auratus* has remained largely unstudied. In Australia, this species is listed under State and national threatened species legislation, but unlike many threatened galaxiids, and despite predation from introduced salmonids (Stuart-Smith *et al.*, 2004), wild populations of *G. auratus* are abundant and juvenile and adult fish are reasonably easy to collect (Fig. 3). Two translocated ‘refuge’ populations of *G. auratus* have also recently been established in nearby, small off-stream agricultural water storages (Hardie, 2003), one of which is also relatively abundant. For these reasons, *G. auratus* is a good model species to use to investigate population monitoring methods for threatened lacustrine galaxiids and the biology and ecology of these fishes in waters that have altered hydrology.



Fig. 1. Female *Galaxias auratus* (183 mm fork length). This individual has well developed gonads; thus, the abdominal wall is reasonably distended.



Fig. 2. View from a north-western shore of Lake Sorell looking south towards Table Mountain during 2000.



Fig. 3. Hauling a catch of *Galaxias auratus* in a fine-meshed fyke net in Lake Sorell during 2000 (samplers: Scott Hardie (left), Brett Mawbey (right)). Overnight fyke netting was the main technique used to sample juvenile and adult fish in this study.

1.3. Lakes, littoral habitats and hydrology

Water level fluctuations are an important hydrological variable in lacustrine systems, particularly as they control the availability and condition of habitats in littoral areas (Gasith & Gafny, 1990). Many lacustrine fishes use littoral habitats at some stage of their life cycle (e.g. for spawning or during a juvenile phase (Winfield, 2004a)); therefore, access to these areas can greatly influence fish production (Gafny *et al.*, 1992; Rowe *et al.*, 2002b). Anthropogenic manipulation of water levels in impounded natural lentic waters (i.e. natural lakes that have been dammed for water storage purposes) can alter the timing, magnitude, duration and periodicity of fluctuations. For these reasons, alterations to hydrological regimes are likely to be a major threat to some freshwater fishes. Additionally, fish play important ecological roles in lake ecosystems, particularly as many species pass through a zooplanktivorous stage during their ontogeny (Winfield, 2004b). Their ontogenetic habitat shifts can link different areas of

lakes by providing pathways for energy flow between habitats (Schindler & Scheuerell, 2002; Vander Zanden & Vadeboncoeur, 2002). Therefore, if fish are reliant upon certain hydrological conditions and the habitats they provide to complete their life cycle, lake hydrology may also indirectly control connectivity between littoral, pelagic and benthic habitats.

Lakes high in the local hydrological landscape (e.g. highland lakes) often have relatively small catchments and, thus, receive a greater proportion of their water from precipitation than lakes lower in the hydrological landscape (Warren & French, 2001). Because the hydrology of these systems depends heavily on local climatic conditions, these waters can be greatly influenced by climatic extremes, such as droughts. Whilst some researchers have documented the affects of droughts on riverine fishes (Matthews & Marsh-Matthews, 2003), few have investigated the response of lacustrine fishes to these events. In southern Australia, one of the main drivers of climatic variability is the El Niño Southern Oscillation (ENSO) (Kershaw *et al.*, 2003). The periodic occurrence of El Niño-induced droughts causes high temporal variability in the hydrological regimes of rivers in the region (McMahon & Finlayson, 2003). Coupled with increasing demands for water resources, El Niño-induced droughts may represent a significant threat to the freshwater fish fauna of southern Australia. Therefore, knowledge of the processes by which these events affect fish populations is needed, particularly for species with restricted distributions where refuge habitats may be scarce or unavailable.

1.4. Research objectives and approach

The objective of this thesis was to examine the crucial aspects of the biology and ecology of *G. auratus* that are likely to be vulnerable to the two main anthropogenic disturbances in lakes

Sorell and Crescent: (1) changes in habitat quality and quantity that are associated with altered water levels, and (2) predation and competition from introduced salmonids. By studying this species, I hoped to gain insight into the ecological attributes of other threatened galaxiids, threats to this family and other similar small-sized lentic species.

I focused on the likely stages of this species' life cycle and their habitat associations using data for other lacustrine galaxiids as a guide (Fig. 4). A field-based (rather than a laboratory-based) approach that involved examining all five suspected life stages of *G. auratus* (see Fig. 4) was taken by studying wild populations in lakes Crescent and Sorell. Over a temporal scale of several years (up to 5 years for some data), *G. auratus* populations were subject to seasonal hydrological and climatic cycles which allowed the roles of these factors to be explored. However, at a smaller spatial and temporal scale, a translocated population of *G. auratus* in an agricultural water storage was used to evaluate methods to monitor populations as this small closed system enabled more robust comparisons to be made.

The approach I used for this research followed a logical series of steps which helped develop the concepts for each chapter of this thesis. Firstly, due to a lack of information regarding the Tasmanian galaxiid fauna, I reviewed its status and identified threats and knowledge gaps using existing literature (Chapter 2). This highlighted the lack of data regarding the life cycles and habitat requirements of the Tasmanian fauna and illustrated the significance of the non-migratory species which inhabit highland lakes on the TCP. Secondly, because several population sampling methods are routinely used to monitor galaxiid populations in lentic waters but none had been formally evaluated, I conducted a study to do this using a translocated population of *G. auratus* (Chapter 3). Thirdly, I focused on the wild *G. auratus* populations in lakes Crescent and Sorell and sampled them intensively (i.e. principal sampling

at monthly intervals) over 2.5 years, to examine: *i*) seasonal variation in catches (Chapter 4), *ii*) age and growth, and population structure (Chapter 5), *iii*) reproduction (Chapter 6) and its associated mortality (Chapter 7), and *iv*) recruitment dynamics (Chapter 8). During this time, I also identified and surveyed the littoral spawning habitats of *G. auratus* in Lake Crescent which have a limited extent. This exercise showed that the availability of this critical habitat is heavily influence by water level fluctuations in Lake Crescent, whereas in Lake Sorell, access to this habitat is not reliant upon hydrology. Because of these findings, I continued to monitor the recruitment of these populations for a further 2.5 years (5 breeding seasons in total) to examine the roles of hydrology and habitat availability (Chapter 8).

1.5. Thesis structure and format

The structure of this thesis reflects the logical approach followed for this research. Consequently, chapters are presented in a sequence which allows them to build upon the findings of previous ones and, in some cases, investigate previously posed questions. All chapters, which vary in length, have been prepared as papers for publication in scientific journals; some have been published, accepted or submitted for publication and others have not yet been submitted. Because of this formatting and the multiple use of sampled material (i.e. fish specimens) and other data (e.g. water levels), there is some repetition within the chapters. However, each chapter has a unique focus and seeks to address a different set of aims which relate to the conservation biology of *G. auratus*.

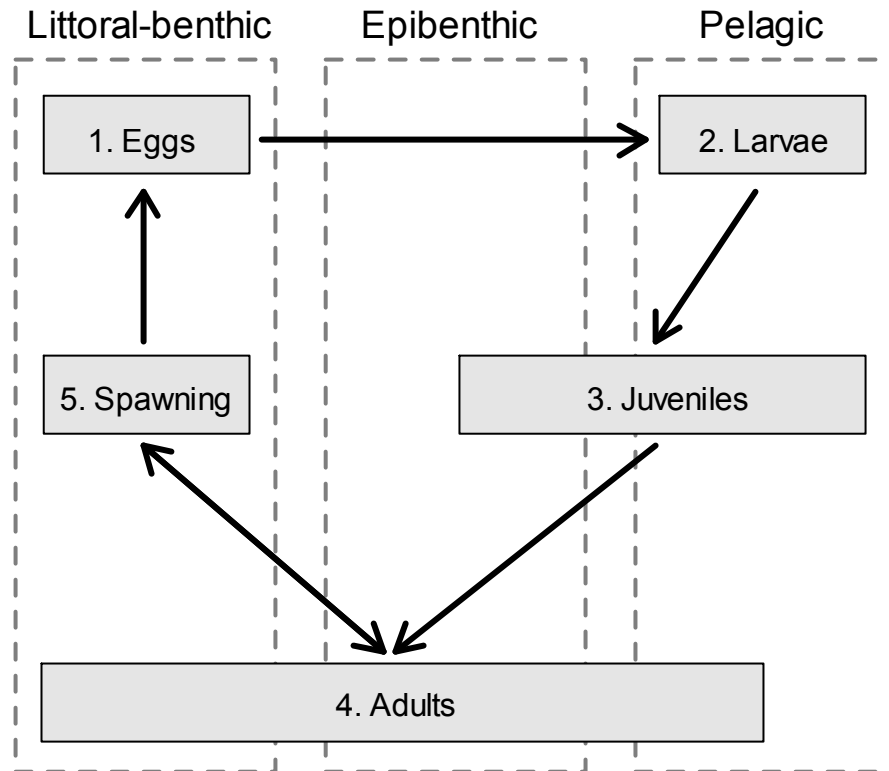


Fig. 4. Five typical stages in the life cycle of a non-diadromous lacustrine galaxiid and the main habitats used during each stage based on published data for other similar species (Pollard, 1971; Humphries, 1989; Rowe & Chisnall, 1996a; Barriga *et al.*, 2002; Rowe *et al.*, 2002a; Morgan, 2003). The relative proportion of time each life stage occupies the three habitats is represented by the position of the horizontal bars.

2. Status of galaxiid fishes in Tasmania, Australia: conservation listings, threats and management issues*

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2.1. Abstract

1. Fishes of the family Galaxiidae are restricted to the Southern Hemisphere where they occupy a diverse array of habitats ranging from over 2000 m in elevation to sea level. Some species are diadromous and hence freshwater, estuarine and marine habitats are used during their life cycle; other species complete their entire life cycles in freshwater environments.
2. Tasmania has a diverse galaxiid fauna that accounts for 64% of native freshwater fish species found on the island. The Tasmanian galaxiid fauna is characterised by high species richness (5 genera and 16 species), endemism (11 species), restricted distributions (11 species), and non-diadromous life histories (11 species).
3. The galaxiid fauna of Tasmania has significant conservation status with 69% of species considered 'threatened'. The conservation status of the fauna is recognised at State, National and International levels.
4. The key threats to galaxiids in Tasmania are exotic species, hydrological manipulations, restricted distributions, general habitat degradation, and exploitation of stocks.
5. While work has recently been undertaken to conserve and manage Tasmanian galaxiid populations, the fauna is still thought to be imperilled. Knowledge gaps needing to be addressed include the biology and ecology of most species (e.g. reproductive biology, life histories, and habitat use and requirements), impacts of habitat manipulations, as well as

mechanisms and impacts of interactions with exotic species. Techniques to accurately monitor the status of galaxiid species and their populations need to be developed and the coexistence of some galaxiids with introduced salmonids should also be examined.

Key words: Galaxiidae; freshwater fish; southern Australia; threatened species; endemism; introduced species; conservation; fisheries management

2.2. Introduction

Fishes of the salmoniform family Galaxiidae are relatively small (adults generally <300 mm long) and scaleless. They are generally regarded as freshwater species, although a few members of the family are diadromous, and some species have both diadromous and non-diadromous life histories. Diadromous species may occupy estuarine and/or freshwater habitats, but are characterised by having a marine juvenile stage in their life cycle (Andrews, 1976). The family is restricted to the Southern Hemisphere, where approximately 50 species (McDowall, 1996) occur largely in cool temperate regions including Australia, New Zealand, Chile, Patagonian Argentina and South Africa (McDowall & Frankenberg, 1981).

The family is particularly diverse in southern Australia where 5 genera and 22 species occur in freshwater and estuarine lotic and lentic habitats, 91% of which are endemic to the continent (Allen *et al.*, 2002). Galaxiids commonly occur from sea level up to elevations of around 1200 m a.s.l. in Australia, although one species (*Galaxias olidus*) inhabits lake and creek habitats up to approximately 2000 m a.s.l. in the Kosciuszko area of southern New South Wales (Raadik & Kuitert, 2002).

The main island of Tasmania and its numerous smaller offshore islands (total land area of 55 000 km² (Williams, 1974)) lie at the south-eastern tip of mainland Australia (between 143°48'-148°30'E and 39°34'-43°41'S) (Fig. 1). The landform of central Tasmania is dominated by the Tasmanian Central Plateau (TCP) (Fig. 1) where there are over 4000 lentic waterbodies (Lynch, 1972) many of which are interconnected. Currently, 34 freshwater fish species (25 native and 9 exotic) are known to occur in Tasmanian inland waters (Table 1), which provide a regional stronghold for galaxiids in Australia with 5 genera and 16 species, 11 of which are endemic (Table 1). Galaxiids account for 64% of native freshwater fish species in Tasmania (Table 1) and the majority of the island's threatened freshwater fish species, with only one other freshwater species, *Prototroctes maraena*, being listed under State and National legislation. The galaxiid fauna comprises two subfamilies with Galaxiinae being dominant (15 species) while Aplochitoninae is represented by *Lovettia sealii*. Galaxiids are distributed throughout Tasmania in subalpine lakes, rivers, wetlands, coastal lagoons and estuaries. The TCP is regarded as a localised 'hotspot' of freshwater fish endemism in comparison to other regions of Australia (Allen *et al.*, 2002), due to the occurrence of seven endemic non-migratory galaxiid species in highland lakes and lagoons.

Galaxiids have significant conservation value with over 50% of species globally being 'threatened' (i.e. protected under legislation in their region(s) of occurrence or listed on conservation awareness lists). In Australia, 16 of the 22 species (73%) are considered to be 'threatened' (ASFB, 2003). Common threats in Australian waters include barriers to fish movement (Harris, 1984; Koehn & O'Connor, 1990; Gehrke *et al.*, 2002) and competition and/or predation from introduced salmonids (Tilzey, 1976; Raadik *et al.*, 1996; Lintermans, 2000), while overexploitation (i.e. harvesting) may have an impact on galaxiid populations in some systems in Tasmania (Blackburn, 1950).

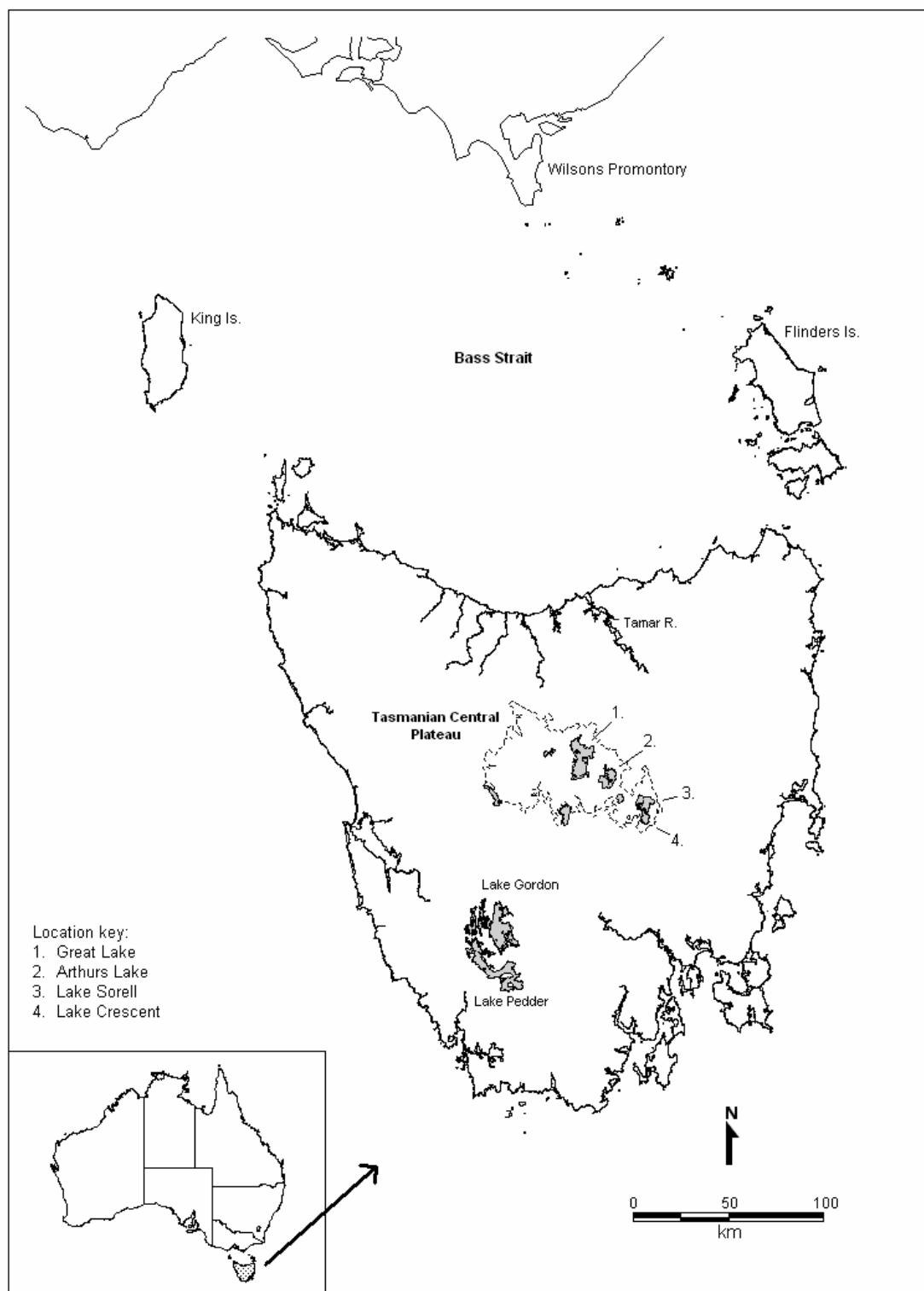


Fig. 1. Tasmanian region showing the Central Plateau and some major rivers and lakes.

Table 1. Freshwater fish fauna of Tasmania.

Native species	Introduced species
Pedder galaxias (<i>Galaxias pedderensis</i>)*	Brown trout (<i>Salmo trutta</i>)
Swan galaxias (<i>Galaxias fontanus</i>)*	Atlantic salmon (<i>Salmo salar</i>)
Clarence galaxias (<i>Galaxias johnstoni</i>)*	Rainbow trout (<i>Oncorhynchus mykiss</i>)
Saddled galaxias (<i>Galaxias tanycephalus</i>)*	Brook trout (<i>Salvelinus fontinalis</i>)
Golden galaxias (<i>Galaxias auratus</i>)*	Redfin perch (<i>Perca fluviatilis</i>) [†]
Swamp galaxias (<i>Galaxias parvus</i>)*	Goldfish (<i>Carassius auratus</i>) [†]
Climbing galaxias (<i>Galaxias brevipinnis</i>)	Tench (<i>Tinca tinca</i>) [†]
Spotted galaxias (<i>Galaxias truttaceus</i>)	European carp (<i>Cyprinus carpio</i>) ^{†‡}
Common galaxias (<i>Galaxias maculatus</i>)	Eastern gambusia (<i>Gambusia holbrooki</i>) ^{†‡}
Arthurs paragalaxias (<i>Paragalaxias mesotes</i>)*	
Shannon paragalaxias (<i>Paragalaxias dissimilis</i>)*	
Great Lake paragalaxias (<i>Paragalaxias eleotroides</i>)*	
Western paragalaxias (<i>Paragalaxias julianus</i>)*	
Australian mudfish (<i>Neochanna cleaveri</i>)	
Dwarf galaxias (<i>Galaxiella pusilla</i>)	
Tasmanian whitebait (<i>Lovettia sealii</i>)*	
Tasmanian smelt (<i>Retropinna tasmanica</i>)*	
River blackfish (<i>Gadopsis marmoratus</i>)	
Australian grayling (<i>Prototroctes maraena</i>)	
Short-headed lamprey (<i>Mordacia mordax</i>)	
Pouched lamprey (<i>Geotria australis</i>)	
Long-finned eel (<i>Anguilla reinhardtii</i>)	
Short-finned eel (<i>Anguilla australis</i>)	
Southern pygmy perch (<i>Nannoperca australis</i>)	
Freshwater flathead (<i>Pseudaphritis urvillii</i>)	

*Endemic to Tasmania.

[†]Considered a 'pest' species by State authorities.

[‡]Listed as a 'controlled' species under State legislation.

The imperilled status of freshwater fishes and their conservation has received significant attention in recent years in Australia (Cadwallader, 1978; Ingram *et al.*, 1990; Pollard *et al.*, 1990; Faragher & Harris, 1994; Cadwallader, 1996) and on a global scale (Cambray & Giorgio Bianco, 1998; Duncan & Lockwood, 2001). Given the significance of the Tasmanian galaxiid fauna and its high endemism, management of these fishes is of utmost importance to conserving the region's freshwater fish biodiversity and protecting the species diversity of the family. This chapter briefly comments on the taxonomy of galaxiids in Tasmania, summarises the family's conservation status in Tasmania, defines threatening processes, and discusses management issues.

2.3. Taxonomy

Until recently, only three Galaxiinae genera were recognised in Tasmania; however, studies have warranted a change in the taxonomy of the mudfish previously known as *Galaxias cleaveri* to *Neochanna cleaveri* (McDowall, 1997a; Waters & White, 1997). This species was considered endemic to Tasmania until 1980 when it was found on mainland Australia at Wilsons Promontory in Victoria (Fig. 1) (Jackson & Davies, 1982). It has since been collected on Flinders Island in Bass Strait (Green, 1984) and along the western coast of Victoria (Koehn & Raadik, 1991), further confirming its occurrence outside Tasmania. It is therefore proposed that the common name 'Australian mudfish' (as used by McDowall (1997a)), rather than the currently used 'Tasmanian mudfish', be adopted for *N. cleaveri*, as it is currently the only *Neochanna* species recognised in Australia.

An additional Tasmanian endemic species, *G. niger*, described by Andrews (1985), has not been included in this paper (Table 1) as its validity is uncertain (Fulton 1990; R. M. McDowall, pers. comm.). A current genetic and morphometric study on some galaxiids

(Raadik, 2001), including this species, should help resolve its taxonomic status (T. A. Raadik, pers. comm.).

2.4. Conservation listings of Tasmanian galaxiids

The galaxiid fauna of Tasmania has significant conservation value (Table 2). Eleven (10 endemic species plus *Galaxiella pusilla*) of the 16 species (69%) that occur in Tasmania are currently listed under the Tasmanian *Threatened Species Protection Act 1995* and at a national level by the Australian Society for Fish Biology (ASFB, 2003). Five are also listed under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* and seven at an international level (IUCN, 2003).

Tasmania is home to arguably the most ‘endangered’ freshwater fish in Australia, *G. pedderensis*, which is no longer found in its remaining natural habitat (tributaries of Lake Pedder impoundment (Fig. 1)) and now occurs only in two translocated populations (Hamr, 1995; Threatened Species Section, 2006). It is also important to note that all four species of the endemic genus *Paragalaxias*, which are confined to the TCP, are listed under threatened species legislation.

Table 2. Conservation status of galaxiids in Tasmania.

Species	Conservation status			
	TTSP ^a	ASFB ^b	EPBC ^c	IUCN ^d
<i>Galaxias pedderensis</i> *	endangered	critically endangered	endangered	critically endangered
<i>Galaxias fontanus</i> *	endangered	endangered	endangered	critically endangered
<i>Galaxias johnstoni</i> *	endangered	endangered	endangered	critically endangered
<i>Galaxias tanycephalus</i> *	vulnerable	vulnerable	vulnerable	vulnerable
<i>Galaxias auratus</i> *	rare	endangered	-	-
<i>Galaxias parvus</i> *	rare	endangered	-	data deficient
<i>Galaxias brevipinnis</i>	-	-	-	-
<i>Galaxias truttaceus</i>	-	-	-	-
<i>Galaxias maculatus</i>	-	-	-	-
<i>Paragalaxias mesotes</i> *	endangered	endangered	-	vulnerable
<i>Paragalaxias dissimilis</i> *	vulnerable	endangered	-	-
<i>Paragalaxias eleotroides</i> *	vulnerable	endangered	-	-
<i>Paragalaxias julianus</i> *	rare	low risk	-	-
<i>Neochanna cleaveri</i>	-	-	-	-
<i>Galaxiella pusilla</i>	rare	vulnerable	vulnerable	vulnerable
<i>Lovettia sealii</i> *	-	-	-	-

^aTasmanian *Threatened Species Protection Act 1995*.

^bAustralian Society for Fish Biology Threatened Species Listings 2003 (ASFB, 2003).

^cCommonwealth *Environment Protection and Biodiversity Conservation Act 1999*.

^dInternational Union for Conservation of Nature and Natural Resources Red List 2003 (IUCN, 2003).

*Endemic to Tasmania.

2.5. Threats to Tasmanian galaxiids

While the threats facing some Tasmanian galaxiid species have been reported (Hamr, 1995; Crook & Sanger, 1997; Crook & Sanger, 1998a, b; Koster, 2003; Hardie *et al.*, 2004; Threatened Species Section, 2006), mechanisms and the scale of impacts have yet to be formally identified or quantified.

Major threats to freshwater fish species on a global scale have been discussed and summarised by Bruton (1995), who proposed that factors causing fishes to become threatened can be summarised into eight broad categories. While he did not include a category to account for the naturally confined or disturbance-induced fragmented distribution of fishes (i.e. ‘restricted distribution’), it can be viewed as a threat to the survival of a species. Although not a process itself, the ‘restricted distribution’ of a species leaves it vulnerable to environmental events (e.g. drought or flood) or human-induced habitat changes, which could have genetic implications for small or isolated populations (Shaffer, 1981; Vrijenhoek, 1998). As Tasmanian endemic galaxiids have limited natural distributions, ‘restricted distribution’ is considered a significant threat to the fauna and it has been included in recent nominations for listing several species under threatened species legislation.

We assessed threats facing all galaxiid fishes of Tasmania (including those without conservation listings) against the eight categories proposed by Bruton (1995), with the addition of a ninth category, ‘restricted distribution’.

Summary of threats

The three key threats to each galaxiid species were determined from current available information (published data, government reports and unpublished data), summed, and

expressed as a percentage of the number of species affected by each threat (Fig. 2; Appendix 1). Five threat categories were found to be relevant to the current status of galaxiids in Tasmania. ‘Global effects’, a category proposed by Bruton (1995), was thought to be the next most significant, although it was not believed to be a major priority for any species at present.

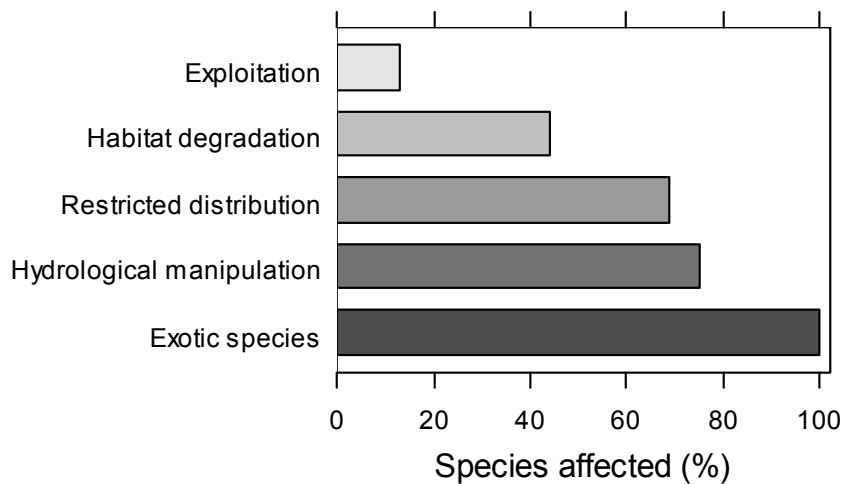


Fig. 2. Threats to galaxiids in Tasmania. Threat categories have been modified from Bruton (1995).

The impacts associated with the introduction of ‘exotic species’ (i.e. competition and predation) were found to be a significant threat to all 16 galaxiid species. This relates primarily to salmonid species, particularly *Salmo trutta* which is widespread in Tasmania. ‘Hydrological manipulation’ and the ‘restricted distribution’ of many species were the next most significant threats, with these categories impacting on 75% and 69% of species respectively. General ‘habitat degradation’ affects 44% of species and ‘exploitation’ (i.e. harvesting) is a major threat to two species, *N. cleaveri* and *L. sealii*, which form significant components of whitebait stocks that are targeted by recreational fishers.

Threats in detail

The primary threat to Tasmanian galaxiids is thought to be competition and predation by introduced species (both exotic and translocated native species). The salmonids *S. trutta* and *Oncorhynchus mykiss*, which form the basis of the State's recreational trout fishery, are of greatest concern. *S. trutta* has further restricted the already naturally confined distributions of *G. fontanus* in the Swan River (148°04'E, 41°57'S) and Macquarie River (147°18'E, 41°50'S) catchments (Crook & Sanger, 1998a) and *G. johnstoni* in the Clarence River catchment (146°19'E, 42°06'S) (Crook & Sanger, 1998b). *S. trutta* also limits the abundance of *G. auratus* in lakes Crescent and Sorell (Fig. 1) (Hardie *et al.*, 2005), *G. truttaceus* (Ault & White, 1994) and *G. tanycephalus* (Inland Fisheries Service, Tasmania, unpubl. data). Much of the early spread of salmonids in Tasmania occurred before most galaxiid species were recognised. *S. trutta* is now found in all habitat types containing galaxiids in Tasmania and is likely to have had varying impacts on all species. Salmonids have also been reported to impact on galaxiids through competition and predation in other regions (Crown *et al.*, 1992; Raadik *et al.*, 1996; Macchi *et al.*, 1999; Raadik & Kuiter, 2002; Cambray, 2003; McDowall, 2003).

Perca fluviatilis are generally viewed as an undesirable 'pest' species in Tasmania, although it is not listed as such in legislation. *P. fluviatilis* impacts upon populations of *G. fontanus* (Crook & Sanger, 1998a) and is present in catchments with several other galaxiid species. Its spread into other systems, particularly those containing endemic galaxiids, could have devastating effects. Tasmanian galaxiids have evolved in isolation from larger freshwater piscivores and are thought to be particularly vulnerable to competition and direct predation from aggressive introduced species such as *P. fluviatilis*. Large *P. fluviatilis* are piscivorous

(Collette *et al.*, 1977) and it has been suggested that they could threaten small fish species, including galaxiids, in southern Western Australia (Pen & Potter, 1992; Morgan *et al.*, 2002). The present restricted populations of *Cyprinus carpio* and *Gambusia holbrooki* in Tasmania stem from unauthorised introductions during the 1980s. Both species are also viewed as undesirable ‘pest’ species and are listed as ‘controlled’ species under the Tasmanian *Inland Fisheries Act 1995*. In Tasmania, *C. carpio* are only found in lakes Crescent and Sorell which is also the entire natural range of the endemic *G. auratus*. Currently *C. carpio* are in low abundance due to an intensive containment and eradication program (Diggle *et al.*, 2004; IFS, 2004). Elsewhere in Australia, *C. carpio* has been shown to degrade habitats (Fletcher *et al.*, 1985; King *et al.*, 1997; Robertson *et al.*, 1997) and alter the trophic structure of freshwater ecosystems (Khan, 2003; Khan *et al.*, 2003; Parkos *et al.*, 2003). While interactions between cyprinids and galaxiids remain largely unstudied, *C. carpio* is thought to pose a threat to *G. auratus* through competition for resources, and possibly habitat degradation if the species was to become abundant.

Gambusia holbrooki is currently restricted to parts of the Tamar River catchment (Fig. 1) in northern Tasmania (Keane & Neira, 2004) and a few small isolated populations in southern Tasmania, which are currently being subjected to eradication efforts. The recent introduction of *Gambusia holbrooki* to Tasmania is of concern as this species has had negative impacts on small indigenous fish species (Myers, 1965; Howe *et al.*, 1997). Its close relative, *Gambusia affinis*, is an aggressive predator of juvenile galaxiids in laboratory experiments (Barrier & Hicks, 1994). It therefore has the potential to impact on galaxiid species in the wild.

Gambusia holbrooki has been identified as a significant threat to the nationally threatened *Galaxiella pusilla* (Koster, 2003) as both species prefer wetland-type habitats.

Translocated native fish species also pose a threat to galaxiids. For example, the native *G. brevipinnis* which was translocated via hydroelectric development from the Gordon River catchment (site of Lake Gordon: Fig. 1) into habitats that were previously only occupied by the endemic galaxiids *G. pedderensis* and *G. parvus*, in the south-west of the State (Hamr, 1992b). *G. brevipinnis* may have contributed to the decline of *G. pedderensis* in Lake Pedder after it was flooded in the 1970s (Hamr, 1995). It has been hypothesised that *G. brevipinnis* has similarly displaced smaller non-migratory galaxiid species in streams above impoundments in New Zealand (McDowall & Allibone, 1994; Allibone, 1999). This species has also been translocated between catchments by the Snowy Mountains hydroelectric scheme on mainland Australia (Morison & Anderson, 1991; Waters *et al.*, 2002).

Habitat alteration associated with the hydrological manipulation of inland waters is threatening 75% of Tasmanian galaxiids. Impoundments and water diversions created by State-wide hydroelectric power schemes that were largely constructed in the 1960-70s are the main contributors. Catchments occupied by 10 of the 11 endemic galaxiids have been altered by hydrological developments. Threats associated with these developments in riverine habitats include barriers to migrations, changes in flow regimes, and alterations to water levels. Habitats of impounded natural lakes are impacted by unseasonal and, in some cases highly variable, fluctuations in water levels. These alterations to hydrological regimes can degrade habitat attributes that are critical in the life cycle of galaxiids (e.g. suitable spawning habitat (Hardie *et al.*, in press)).

The restricted distributions of 10 endemic galaxiid species, and the indigenous *Galaxiella pusilla*, makes their survival precarious. In some cases the already confined natural distributions of these species have been further reduced by human-induced changes or

through exclusion from habitats by introduced species (i.e. *S. trutta*). For several species (e.g. *G. johnstoni*), the decline probably started with the introduction of salmonids in the 1860s, and the original distributions are not known. Future reduction in the range of these species and isolation of their populations is of great concern.

Habitat degradation, including impacts such as sedimentation and reductions in water quality caused by urbanisation, and agriculture and forestry practices, are considered to be significant threats. Land use and catchment and riparian vegetation types have been found to influence the distribution and abundance of galaxiids in New Zealand (McDowall, 1997b; Rowe *et al.*, 1999; Eikaas *et al.*, 2005). The preservation of wetland-type habitats is considered critical for the conservation of *Galaxiella pusilla* and *N. cleaveri* (Koehn & Raadik, 1991; Koster, 2003). Approximately 20 of the 89 Tasmanian wetlands listed under the Directory of Important Wetlands in Australia are declining in condition, and only a few significant sites currently have any sort of short- or long-term security (e.g. through reservation) (Anon, 2003).

The recreational whitebait fishery in Tasmania may pose a threat to galaxiid stocks in some waters. This fishery, which dates back to the 1930s, historically focused on *L. sealii*, the dominant species in the early spring-time migratory runs. However, the reopening of the recreational season in 1990 for four weeks during spring in a limited number of rivers has targeted other species which contribute to the migrations, including *G. maculatus*, *G. brevipinnis*, *G. truttaceus* and *N. cleaveri*. The fishery was studied in detail in the 1940s (Blackburn, 1950), and the status of stocks was examined briefly during the 1980s (Fulton & Pavuk, 1988), but no recent surveys have taken place. Although information is limited, recreational harvesting of whitebait migrations could potentially threaten *L. sealii* and *N. cleaveri* populations in some systems with relatively small catchments; it could also be

impacting on other galaxiid populations by regularly depleting populations at levels that are unsustainable.

Global climate changes at a number of temporal scales (Shulmeister *et al.*, 2004) would no doubt have implications for all galaxiid species in the State. However, at greatest risk are non-migratory species which have restricted distributions in the east of the State (*Galaxiella pusilla* and *G. fontanus*) where rainfall is historically low and highly variable. Eastern populations of migratory whitebait species may also be affected by climatic changes. Climatic change may alter levels of discharge and seasonality of flows in coastal rivers which are important to the reproduction of these species (Blackburn, 1950), thus, limit production of their populations.

Direct effects of climate change include elevated water temperatures and decreases in habitat availability including total loss of water from small headwater streams occupied by *G. fontanus*. Recent low water levels in lakes Crescent and Sorell have been found to have a significant negative impact on the recruitment of *G. auratus* populations through the dewatering of spawning substrate (Hardie, 2003). Impacts on other species around Tasmania may also be a result of changes to hydrological regimes of storages and waterways that are components of the State's hydroelectric scheme. Increases in natural ultraviolet radiation (UVR) may also affect recruitment of galaxiid populations in elevated waters on the TCP because exposure to UVR may increase mortality of galaxiid eggs (Battini *et al.*, 2000).

The effect of climate change on the freshwater fish fauna of New Zealand (which has a similar climate to Tasmania), has been examined by McDowall (1992a). He postulated, amongst other things, a shift from north to south in the distributions of widespread species,

but concluded that human activities were more likely to influence the condition of water courses than climate change.

2.6. Management issues for galaxiids in Tasmania

Recent conservation work

Over the past 20 years, fisheries managers in Tasmania have begun to direct effort towards conserving the State's native freshwater fish fauna, particularly galaxiids. There has been recognition of the significance of galaxiids, and the imperilled status of many species, particularly with the progressive listing of 10 of the endemic species under State legislation.

Work conducted under galaxiid recovery plans (Sanger & Fulton, 1991; Crook & Sanger, 1997) has focused largely on monitoring the status of five endemic species (*G. pedderensis*, *G. fontanus*, *G. johnstoni*, *G. tanycephalus* and *G. parvus*), briefly examining their biology and, in some cases, instigating on-ground works to protect populations. It has also included a public education program. A new recovery plan has recently been prepared (Threatened Species Section, 2006) including all 11 galaxiid species that are listed under State threatened species legislation. If funded, the new recovery plan will continue conservation actions for at least another five years. A management plan for the Tasmanian whitebait fishery has also recently been prepared (IFS, 2006) which includes management strategies to protect galaxiid species that contribute to whitebait migrations. Intensive research on the status, biology and ecology of *G. auratus* has also been undertaken (Hardie, 2003; Hardie *et al.*, 2004, 2005, 2006a; Hardie *et al.*, 2006b; Hardie *et al.*, 2007; Hardie *et al.*, in press).

Recent conservation management actions implemented by these projects include establishment and protection of translocated populations, efforts to eradicate exotic fish,

construction of barriers to prevent exotic species invasions and captive breeding trials (Hardie, 2003; Threatened Species Section, 2006). The work on *G. auratus* also aims to better define the species' life history and determine the effects of lake level management. Pest species eradication and control, and education programs have also been conducted. Eradication and containment efforts have focused on confined populations of feral salmonids, *C. carpio* and *Gambusia holbrooki*, while education about the negative impacts of all five introduced 'pest' species has also taken place (R. Walker, pers. comm.). Public awareness of impacts caused by introduced species is critical to the conservation of the State's galaxiid fauna. This is illustrated by recent unauthorised introductions of *C. carpio* and *Gambusia holbrooki* onto the island and illegal translocations of *Gambusia holbrooki* and salmonids within Tasmania.

Some basic conservation strategies have been implemented by recreational fisheries managers, including no longer stocking salmonids into natural waters that do not already contain them (i.e. no deliberate extension of the distribution of salmonids in natural waters) and generally constraining stocking rates within the historical limits for receiving waters. However, habitats containing threatened galaxiid species continue to be stocked with salmonids even though the impact of stocking strategies has not been examined. Clearly, work is needed in this area if Tasmanian fisheries managers wish to balance the recreational and conservational components of there freshwater fish resources.

Current data and knowledge gaps

The first endemic Tasmanian galaxiids to be described were *G. auratus* and *L. sealii* (Johnston, 1883). Species continued to be described over the proceeding years with the last major period of taxonomic work occurring in the late 1970s when five species were described

(*G. fontanus* (Fulton, 1978b), *G. tanycephalus* (Fulton, 1978a), *P. mesotes* and *P. eleotroides* (McDowall & Fulton, 1978b), *P. julianus* (McDowall & Fulton, 1978a)), while *G. niger* was described in 1985 (Andrews, 1985). Given the generally cryptic nature of galaxiids and the large number of waters in areas of remote wilderness in Tasmania, it is possible that more species remain undiscovered. Therefore it is important that faunal surveys continue in unsampled areas. The use of genetic techniques may enable the recognition of further species belonging to species complexes which have evolved in the isolation of naturally fragmented populations. This has been the case in New Zealand, where genetic analyses have recently discovered galaxiid species complexes that were previously thought to belong to a single species (Allibone *et al.*, 1996; Allibone & Townsend, 1997; Ling *et al.*, 2001; Waters & Wallis, 2001); this also seems likely on mainland Australia (Raadik, 2001).

Ecological data on Tasmanian galaxiids are limited. Information available includes status and distribution (Fulton, 1990; Sanger & Fulton, 1991; Chilcott & Humphries, 1996; Crook & Sanger, 1997; Hardie, 2003; Nelson, 2004; Threatened Species Section, 2006) and evolutionary history and genetics of the galaxiid fauna (Johnson *et al.*, 1981; Johnson *et al.*, 1983; White *et al.*, 1987; Ovenden & White, 1990; Ovenden *et al.*, 1993; Waters, 1996; Waters & White, 1997; McDowall, 1998), as well as the biology and ecology of some species (Fulton, 1982; Humphries, 1989, 1990; Sanger & Fulton, 1991; Hamr, 1992a; Frijlink, 1999; Hardie, 2003; Davis, 2004). However, many aspects of the galaxiid fauna critical for conservation management have not been examined. These include the reproductive biology, life histories, habitat use and requirements of most species, impacts of habitat manipulations (such as water levels), as well as mechanisms and impacts of interactions with exotic species. There is also a lack of genetic information, particularly for those species that have isolated

populations and fragmented distributions. The threats previously outlined also need to be examined in detail to enable their mitigation.

Information on galaxiid biology is lacking on the TCP, where there are seven endemic species including all four species of *Paragalaxias*. Lentic waters on the TCP also support Tasmania's most significant salmonid fisheries. Arthurs Lake and Great Lake (Fig. 1) each contain two endemic galaxiid species (Arthurs Lake: *P. mesotes* and *G. tanycephalus*; Great Lake: *P. dissimilis* and *P. eleotroides*), which have highly restricted distributions and are thought to be under considerable pressure from the salmonid stocks which are actively managed and monitored. However, knowledge of the life cycles and habitat requirements of *Paragalaxias* spp. (Fulton, 1982) and *G. tanycephalus* (Davis, 2004) is limited. The fact that these galaxiids are currently coexisting with introduced salmonids should not be viewed as indicating their resilience to threats posed by salmonids. Salmonids pose well-documented threats to galaxiids (Crowl *et al.*, 1992), and alterations to the condition of these lakes could compound to induce a sudden decrease in the abundance of galaxiids, from which there maybe long-term genetic consequences.

The recent years of very low water levels in Great Lake, a hydro-managed impounded natural lake complex, are an example of a significant habitat change that could have direct or indirect effects on galaxiid populations. These low water levels impact upon the charophyte algal beds, which are at times prolific in the benthic habitats of Great Lake. *Paragalaxias* spp. appear to favour these weed beds, but the impact of altered water levels on their populations is unknown. The rapid disappearance of *G. pedderensis* from the Lake Pedder impoundment less than a decade after the natural lake was inundated (Lake, 1998) illustrates how the status

of a lake-dwelling population can change due to compounding impacts (i.e. introduction of non-native fish and habitat alterations).

The recreational salmonid fishery in Tasmania is important locally and internationally, and it provides financial benefits to many sectors of the community (Anon, 2003). Historical evidence suggests that introduced salmonids have been detrimental to at least a few Tasmanian galaxiid species. For example, *G. johnstoni* has disappeared from waters to which *S. trutta* have been introduced or invaded (Threatened Species Section, 2006). Conversely, some species have maintained populations with introduced salmonids for over 100 years. Again, *G. johnstoni* provides an example; it persists in Clarence Lagoon (146°19'E, 42°06'S) despite the stocking of *Salvelinus fontinalis* since 1963. Maintenance of the *S. fontinalis* fishery discourages the illegal introduction of other salmonid species to the Lagoon (Threatened Species Section, 2006). Clearly, there is still an urgent need for further work to identify both the impacts of introduced salmonids and the mechanisms that operate to allow some galaxiids to coexist with them.

Waters that contain existing salmonid fisheries and coexisting galaxiids continue to be restocked. These translocations are aimed at maintaining recreational salmonid fisheries, even though cost-benefit analyses of these management strategies have not been performed, let alone assessments of the impacts they maybe having on indigenous species. Illegal restocking or translocation of salmonids is highly likely to occur if given waters were destocked. Illegal translocations affecting endangered species have been detected in recent years (e.g. introduction of *O. mykiss* into a lagoon containing *G. johnstoni* (Jackson *et al.*, 2004)), therefore, any future strategic removals or destocking of salmonids will need to be planned and executed carefully.

Translocations of fish species via water transfers associated with hydro-electric schemes on the TCP also threaten galaxiids. Examples include transfers from the Western Lakes area (146°22'E, 41°52'S) into Great Lake, and between Great Lake and Arthurs Lake, which have the potential to translocate *Paragalaxias* spp. and *G. tanycephalus* into waters outside of their natural range. The likelihood of these translocations and their potential impacts on galaxiid populations have not been fully investigated.

The status of the Tasmanian whitebait fishery requires assessment. Investigations should focus on the size of runs, species composition in recreational catches, and determine the effectiveness of current management regulations in protecting populations. Comparisons could then be drawn with previous studies (Blackburn, 1950; Fulton & Pavuk, 1988).

Short-term actions

In the short-term, several actions are considered important for the conservation of Tasmanian galaxiids. The status of populations of threatened species should be monitored on a regular basis and rigorous quantitative or standardised qualitative monitoring methods need to be developed. The question “what do the catches mean?” needs to be formally addressed to aid programs monitoring galaxiid populations.

A state-wide GIS-based freshwater fish distribution database should be developed. Currently the recording of catches by various collectors (public and private sectors) is poorly coordinated and management of the data is *ad hoc*. A database of this kind would have benefits for the management of galaxiid fauna and also other natives, recreational species, and exotic pests. Monitoring of the spread of exotic species is critical to the management of the State's freshwater fisheries as a whole.

The ecology and biology of all galaxiid species need to be examined, especially for species under greatest threat. Mitigation strategies to address identified threats should be investigated and implemented, and faunal surveys in unsampled areas should be given priority. New populations of galaxiids should be investigated by taxonomists and, where necessary, genetic analyses be conducted to resolve taxonomic uncertainty.

Long-term issues

Successful long-term management of the Tasmanian galaxiid fauna depends upon several issues being addressed. These include management of fish passage, protection and rehabilitation of aquatic and riparian habitats, control over translocations of exotic and native species and containment of the distribution of exotic pest species, and adherence to protective legislation for threatened species management.

Clearly the use of freshwater, estuarine and marine habitats by diadromous galaxiids during their life cycles makes management of their 'habitat' difficult. Problems associated with the conservation of diadromous species have been highlighted by McDowall (1992b). In Tasmania, diadromous species occur below and above seaward limits, meaning that their protection and management rely on at least two separate management agencies and different government legislation. However, currently most diadromous galaxiid species in Tasmania are common, widespread, and are not considered to be significantly threatened. Non-diadromous species with restricted distributions are of greatest concern. Management issues associated with these species, which generally occupy lentic habitats (i.e. lakes and lagoons), are more localised and, therefore, should be more straightforward to address.

Control over the translocation of exotic and native fish into and within Tasmania is critical to the conservation of the galaxiid fauna. Management of the State's recreational and commercial freshwater fisheries should aim to conserve the indigenous fish fauna. The distributions of salmonids should be monitored, particularly as they are a valued recreational species in Tasmania and illegal translocations are known to occur. In New Zealand it has been speculated that *S. trutta* is continuing to spread 130 years after their initial introduction into the country (McDowall, 2000). This is also the case in Tasmania (Inland Fisheries Service, Tasmania, unpubl. data).

The distribution of other introduced species within the State should continue to be monitored. Management authorities also need to be aware of the further spread of exotic species in southern mainland Australia and of the potential success or otherwise of translocations of species that are currently not found in Tasmania. Authorities should also take action to prevent the introduction and establishment of further freshwater fish species into the State by enforcing greater control over aquarium and aquaculture industries.

Resources to enforce conservation legislation are often lacking, thus freshwater fishes are not always afforded the protection they are designated under law (Collares-Pereira & Cowx, 2004). Conservation listings of Tasmanian galaxiids should be kept up to date to ensure that threatened species have appropriate protection provided by legislation when management issues arise. The level of protection given to 'threatened' species in Tasmania needs to be considered by management authorities. State (*Threatened Species Protection Act 1995*) and Commonwealth (*Environment Protection and Biodiversity Conservation Act 1999*) legislation is in place for the protection of threatened species in Tasmania; however, the legislation is, in some cases, not implemented. Under State legislation, any impact upon a single individual of

a listed species is not acceptable. However, current fisheries management strategies in waters containing threatened galaxiids do not have sufficient data to ensure the security of these fishes via their management actions. Nor is there, in many cases, quantitative evidence to prove that these strategies are actually beneficial to recreational fisheries.

Current policies that allow the continued stocking of salmonids into waters containing listed galaxiids are being implemented without assessments of their impacts or benefit to recreational fisheries. However, removal of salmonids from waters containing threatened species is being implemented and is recognised as a management option for other waters where threatened species exist. While programs to eradicate salmonids from some waters of conservation significance will likely be resisted or reversed illegally, we feel strongly that for the long-term security of Tasmanian galaxiids three things need to happen. First, monitoring of the entire fish communities within which vulnerable galaxiid populations occur needs to continue, so that illegal or accidental introductions of exotic species can be speedily dealt with. Second, the impacts of introduced salmonids need to be further researched, prioritising the most vulnerable galaxiids, so that *ad hoc* hypotheses about the potential long-term coexistence of galaxiids and salmonids can be evaluated, and the benefits of changes to stocking regimes can be clearly identified. Third, for those waters where the risk posed by continued salmonid stocking is untenable, substantial educational and enforcement campaigns should be undertaken to ensure the enduring success of any attempt at salmonid eradication or destocking.

3. Comparison of day and night fyke netting, electro-fishing and snorkelling for monitoring a population of the threatened golden galaxias (*Galaxias auratus*)*

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3.1. Abstract

The littoral zone of small off-stream water storage containing a translocated population of *Galaxias auratus* was sampled fortnightly at day and night with fyke nets, electrofishing and snorkelling over 3 months. Variation in population data provided by each method, including relative abundance indices, size structure, and habitat preferences, were examined. Aspects of behaviour and activity patterns were also investigated. Night sampling using all methods consistently yielded larger catches than day sampling. The size structure of catches varied, with electrofishing at night and fyke netting during the day having higher proportions of juveniles, whilst snorkelling at night and electrofishing during the day had higher proportions of adults. Fyke netting at night yielded by far the largest catches (~3-fold more than other methods) and also captured good numbers of both juveniles and adults. *Galaxias auratus* had a strong diel activity pattern and were most active at night. The majority of the population moved into the littoral zone during the night and back into deeper water during the day. A small number of juveniles remained in the littoral zone and some adults sheltered in the dense cover of species-rich littoral vegetation during the day. Shores with shallow depth profiles appeared to be preferred due to higher catches in these areas using all methods. Based on the results of this study, fyke netting at night in littoral habitats is recommended for monitoring

populations of *G. auratus*. Fyke netting is likely to be an effective method for monitoring other lacustrine galaxiid species; however, further work is required to investigate the effects of habitat variables and fish community structure on activity patterns of galaxiids, and hence their catchability with various methods, in more extensive lentic environments.

Key words: Galaxiidae; Tasmania; threatened fish; population monitoring; sampling methods; diel activity patterns

3.2. Introduction

Conservation-based management efforts for threatened freshwater fishes are increasingly being conducted under single or multi-species recovery plans (Brown *et al.*, 1998; Department of Conservation, 2004). Accurate monitoring of the status of threatened fish populations is a critical component of research that is required to underpin management decisions relating to threatened species (Cowx, 2002); it is also fundamental to the implementation of recovery programs which aim to prevent further declines and facilitate recovery. Abundance and size structure data are required to assess the impact of identified threats, the effectiveness of management strategies, and the long and short-term viability of populations.

Fishes of the Galaxiidae are found in freshwater and estuarine environments in cool temperate regions of several land masses in the Southern Hemisphere (McDowall & Fulton, 1996).

Many species throughout the family's range of occurrence are considered to be threatened (Hamr, 1995; Gill & Morgan, 1997; David, 2002; Keith, 2002); hence, monitoring populations for conservation purposes is a common management practice. Galaxiids account for 64% of the native freshwater fishes in Tasmania, Australia and comprise 16 species, of

which 11 are endemic to the island. Tasmanian endemic galaxiids are generally lake-dwelling (i.e. non-migratory life histories), have highly restricted distributions and are ‘threatened’ (i.e. listed under State threatened species legislation) (Hardie *et al.*, 2006b) and they have all been included in a recent recovery plan (Threatened Species Section, 2006) which aims to conserve their populations.

The golden galaxias (*Galaxias auratus* (Johnston)) is a typical example of an endemic Tasmanian galaxiid. This species is non-migratory, protected by State and national legislation, and its distribution is confined to the upper reaches of the Clyde River catchment in the south-east of the Tasmanian Central Plateau (Hardie *et al.*, 2004). Two natural populations exist in the interconnected lakes Crescent and Sorell. Two translocated populations have also recently been established in off-stream agricultural water storages within the catchment by authorised transfers of adult fish from natural populations (Hardie, 2003).

A primary requirement when selecting methods to sample threatened fish is to ensure damage to caught fish is minimal, so as to enhance the survivorship of released fish. Techniques for monitoring galaxiid populations need to account for several attributes of these fishes: small size (<300 mm long), scaleless and hence susceptible to abrasion, generally benthic and cryptic (McDowall & Frankenberg, 1981), diel changes in behaviour and activity (Rowe & Chisnall, 1996a; David & Closs, 2003), and occupation of a diverse range of habitats (McDowall, 1990; McDowall & Fulton, 1996).

Methods previously used to monitor galaxiid populations in lakes generally sample littoral habitats and include electrofishing (Fulton, 1982; Humphries, 1989), trap netting (Rowe *et al.*, 2003), seine netting (Pollard, 1971), gill netting (Barriga *et al.*, 2002), fyke netting (Rowe &

Chisnall, 1997a; Rowe *et al.*, 2002a) and snorkel censuses (Inland Fisheries Service, Tasmania, unpubl. data). In-lake hydroacoustic measurements have also been employed in larger waters for galaxiid larvae (Taylor *et al.*, 2000) and adults (Rowe & Chisnall, 1996a).

Catchability for a given method is a measure of the interaction between the abundance of a fish species and the efficiency of the fishing method, which is influenced by habitat parameters, fish behaviour, and community dynamics (Arreguín-Sánchez, 1996). Fish species are usually active during limited periods within the diel cycle; diurnal species in daytime, nocturnal species at night and crepuscular species at dawn and dusk (Wootton, 1998). Littoral habitats are known to be important areas for small-sized fish in lentic waters (Fischer & Eckmann, 1997; Brosse & Lek, 2002), with the existence of different kinds of littoral structure, substrate, or vegetation being critical to the zonation of fish in lakes (Matthews, 1998). Given variations in morphology, behaviour and habitat use in fish of different age and size, all sampling methods have limitations and degrees of bias towards certain size-classes of a species. To reduce this bias, a combination of fishing methods is often employed to monitor populations. However, use of a single technique, to which the target species has a high degree of catchability across varying habitats, enables direct comparisons of relative abundance and sometimes population structure.

In Tasmania, monitoring regimes for lacustrine galaxiid populations have relied largely upon shore-based electrofishing and to a lesser extent littoral fyke netting, despite suspected unquantified influences on catchability from climatic variables (e.g. wind induced turbulence), habitat parameters (e.g. substrate types), and behavioural patterns of target species. Recent day time littoral sampling of *G. auratus* in lakes Crescent and Sorell showed that catches using both electrofishing and fyke netting are greater in shallow (depths <1.0 m)

areas which contain complex habitats such as rocky substrate or aquatic vegetation (Hardie, 2003). Electrofished catches of several fish species have been shown to vary across diel periods in littoral habitats (Pierce *et al.*, 2001), as have fyke net catches of adult galaxiids (Rowe & Chisnall, 1996a). To date, the efficiency of sampling regimes for monitoring lacustrine galaxiid populations has not been critically examined. Evaluation of commonly used methods will improve the accuracy of monitoring techniques and, therefore, the ability to detect changes in the status of threatened species and effectiveness of conservation strategies.

The aim of this study is to assess which of three commonly used sampling methods (fyke netting, electrofishing and snorkel censuses) best describes the status of a small population of *G. auratus*. To do this we compared catches of *G. auratus* collected from the littoral zone of a small closed system (off-stream water storage) at day and night using these sampling methods. Given the habitats present at the study site, each of these methods was deemed suitable for sampling the galaxiid population. Variation in population data provided by each sampling method including relative abundance indices, size structure, and habitat preferences were examined. Aspects of behaviour and activity patterns were also investigated. The hypotheses analysed were that (1) *G. auratus* have strong diel movement patterns and are nocturnally active, hence, catches will be greater at night, (2) fyke nets are the least size-biased and therefore capture the broadest size range of fish, and (3) vegetated cover and shallow margins with gentle gradient profiles are preferred habitats for *G. auratus*. The results of this study should provide insight into broader methodological issues for monitoring lacustrine galaxiid populations.

3.3. Materials and methods

Study site

The translocated population of *G. auratus* sampled during this experiment was located in a small (0.6 ha) human-made, off-stream agricultural water storage (referred to as a ‘dam’ for the remainder of this paper) on the south-eastern fringe of the Tasmanian Central Plateau, Australia at 550 m a.s.l. (42°17’S, 147°8’E) (Fig. 1). The dam which is situated in the Fordell Creek catchment, a small sub-catchment of the Clyde River catchment, was stocked with *ca* 700 adult *G. auratus* from Lake Crescent during 1998 by the Tasmanian Inland Fisheries Service (Hardie, 2003). An abundant self-sustaining population of *G. auratus* has since established (Hardie, 2003) and no other fish species have been recorded in the dam (S. A. Hardie, unpubl. data). The dam was constructed in 1972, is rectangular in shape, has a capacity of 11 300 m³ at full supply level (but was approximately 0.4 m below full supply level during the study) and had a mean depth of 2-2.5 m (maximum of 3.2 m) at the time of the experiment. Native dry sclerophyll forest covers the dam’s catchment (30 ha), 80% of which has been fenced to exclude stock.

Habitat variables

Physico-chemical water variables (conductivity, dissolved oxygen, pH and turbidity) were measured at four littoral sites once during each sampling occasion ($n = 20$) using the following instruments respectively: WTW Conductivity Meter (LF 330), WTW Oximeter (Oxi 330), WTW pH Meter (pH 320), and HACH Turbidimeter (2100P). Water level was also measured at the beginning of each sampling occasion ($n = 5$) from a reference point marked at the start of the experiment. Three water temperature loggers (OnSolution Pty Ltd) were deployed at a depth of 0.5 m in the littoral zone and were downloaded periodically.

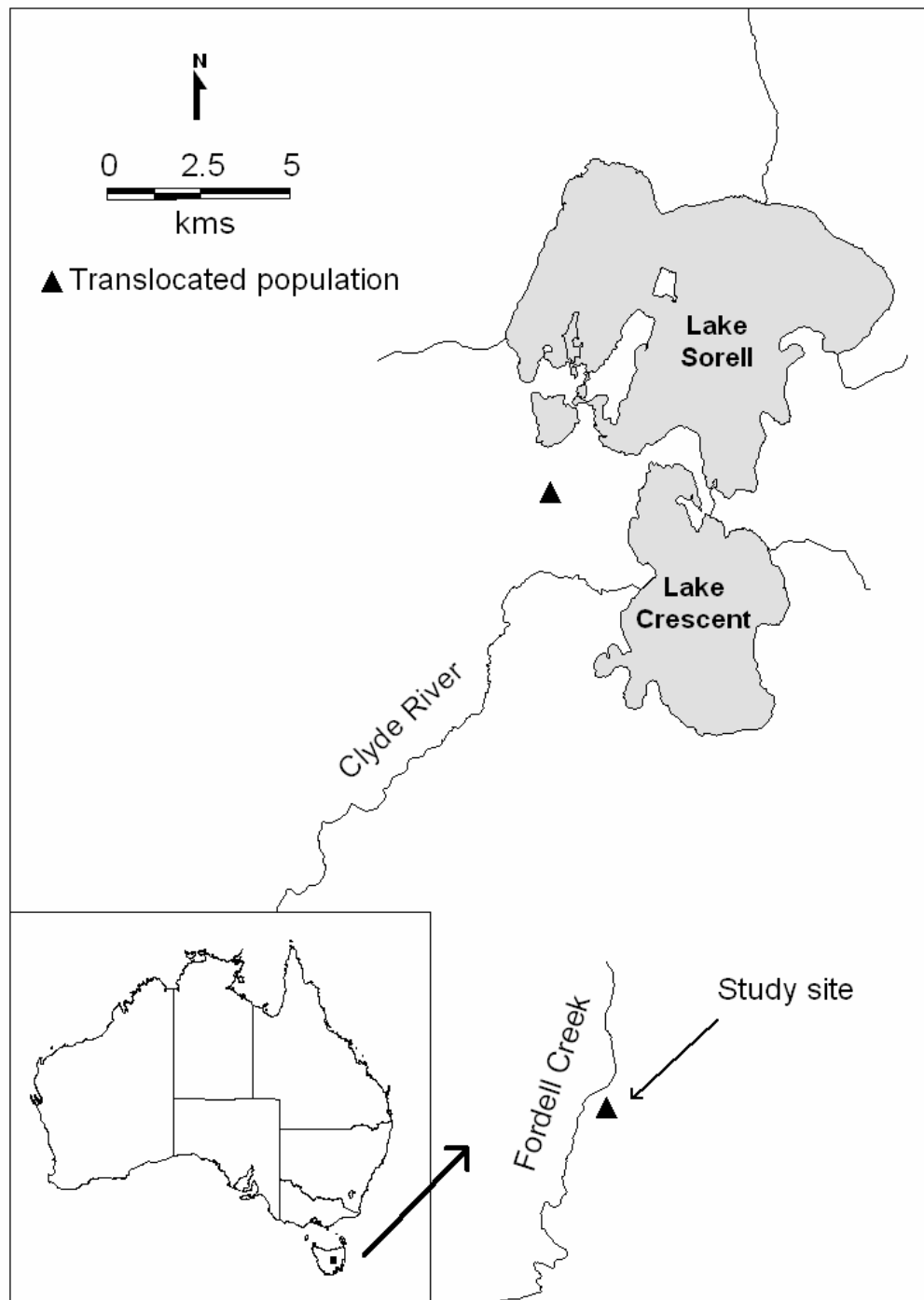


Fig. 1. Location of study site and other populations of *Galaxias auratus* in the lakes Crescent and Sorell region, Tasmania, Australia.

Shoreline depth profiles were surveyed along 18 transects at 90° to the shore. Depth readings were taken at 1 m intervals from the shore until a depth of ~1.2 m was reached (within 7 m of shallower southern and eastern shores and 4 m of deeper northern and western shores). The outer edge of beds of emergent vegetation was also recorded to the nearest 1 m along transects.

Aquatic vegetation in the littoral zone was surveyed at 16 sites (sites were positioned in clockwise order starting in the north-east corner). Plant species, species percent cover, and water depth were measured within a 1 m × 1 m quadrat at each site. The Braun-Blanquet cover scale (Moore & Chapman, 1986) was used to define the area of cover provided by each plant species at the water surface and at the surface of the substrate.

Fish sampling

Fish were sampled between March and May 2004. The time of year selected for this study (autumn) was based on knowledge of the reproductive cycle and recruitment dynamics of *G. auratus* at the study site and in natural populations. After autumn, catchability of adults in littoral habitats becomes highly variable and may increase significantly during spawning (winter – early spring) (Hardie *et al.*, 2005) while juveniles recruit into adult habitats during summer (Chapter 8).

The sampling regime of this study consisted of five fortnightly sampling occasions (each lasting 3-4 d) within which each of the fishing methods was employed at day and night (within a 24 h period). The sequence of methods in each sampling occasion varied and accounted for 5 of the 6 possible sequences. The starting photoperiod (night or day) for each sampling occasion was randomly selected.

Fish were sampled using 12 single wing fine-meshed fyke nets (3 m × 0.6 m wing, 570 mm D-shaped entrance, 5 mm stretched mesh), backpack electrofishing (Smith and Root Inc, model 12-B POW, 250 mm anode ring) and snorkel censuses (14 W torch at night). Fyke nets also had 80 mm stretched mesh screens on the entrance to avoid capture of platypus (Grant *et al.*, 2004) and water birds (Beumer *et al.*, 1981). Sunrise occurred at 0624-0713 h and sunset at 1703-1936 h during the experiment. Fyke nets were set and pulled within the 2 h of light either side of sunrise and sunset and electrofishing and snorkelling were performed at midday and 2 h after sunset for day and night sampling, respectively.

Netting sites (mean depth range 370-930 mm) were spatially distributed around the littoral perimeter of the dam. Nets were set individually at alternating 45° angles to the shore. The same sites were used for netting sessions throughout the study and soak time averaged 8.5 h and 15.5 h during day and night sessions respectively. Electrofishing was undertaken along zig-zagged transects around the entire littoral zone of the dam (<1.2 m deep) by a team of two, with stunned fish being collected by the dip net of the assistant. Electrical settings were kept constant on each sampling occasion (330 V, DC current of 20 Hz and pulse frequency of 8 ms). Snorkel censuses involved a single diver swimming around the entire littoral zone (observations taken at depths <1.6 m) (Fig. 2), with the length of observed fish being estimated to the nearest 10 mm and recorded by a shore-based assistant. Snorkelling visibility was determined on a single occasion at both day and night by measuring the distance from which a white sign (300 × 210 mm) with the black letter “A” (140 mm high and 10 mm line thickness) positioned at 0.5 m below the surface could be read. Visibility was found to be 1.5 m during day and 2.4 m at night. Electrofishing and snorkelling effort (time to complete 1 lap of littoral margin) varied from 30-55 min, generally taking longer at night when catches were greater.



Fig. 2. Day time snorkel census of littoral zone of the dam that contains a translocated population of *Galaxias auratus* during 2004 (sampler: Scott Hardie).

Catch shore⁻¹ (i.e. N, E, S and W) for each method was recorded and summed to obtain the total catch for each session. All fish captured by fyke netting and electrofishing were anaesthetised using a clove oil-based anaesthetic (Aqui-s[®]) and the lengths (total fork length (TFL), nearest mm) of a sub-sample of 50 fish were measured. Anaesthetised fish were allowed to recover in fresh water before being released back into the dam. Where large numbers of fish were collected ($n > 50$), approximately equal proportions of the catch were released in each corner of the dam.

Data analysis

Mean depth at 1 m intervals from shore was calculated for each shoreline, and depth profiles were plotted. Linear regression was used to quantify shore gradients using depth profile data. Community composition of littoral vegetation was examined using agglomerative hierarchical

cluster analyses implemented by the unweighted pair-groups method using arithmetic averages (UPGMA) (Legendre & Legendre, 1998) of the survey sites based on Bray-Curtis matrix of dissimilarities (Faith *et al.*, 1987) of species presence/absence. Vegetation data were also analysed using the same methods to compare sites in terms of their structural similarity in total vegetative cover measured at the water surface and at the surface of the substrate. Catch data for each fishing method and diel period were pooled and square root transformed to normalise residuals prior to analysis. Two-way analysis of variance (ANOVA) was used to determine differences in catches during day and night and between fishing methods. Because no fish were recorded by snorkelling during the day, the two-way ANOVA was only carried out including the other two methods, while a one-way ANOVA was used to test for differences between methods at night. Length data for fishing methods and diel periods were pooled and cumulative length-frequency distributions from each method were compared using Kolmogorov-Smirnov two-sample tests (Quinn & Keough, 2002). Diel differences in catches between methods were analysed to investigate habitat use and activity patterns. Catch data for each shore were corrected to account for the different length of shores (fish m⁻¹ of shore) and these corrected values were $\log_{10}(x + 1)$ transformed to normalise residuals. These data were analysed using two-way ANOVA for each shore with two fixed factors: diel period (day or night) crossed with fishing method (fyke netting, electrofishing or snorkelling), followed by Tukey's tests for any unplanned multiple comparisons (the three-way ANOVA crossing diel period, fishing method and shore yielded a highly significant 3-way interaction, hence the two-way analyses). All analyses were carried out in R version 2.1.1 (R Development Core Team, 2005) using the packages cluster (Maechler & based on S original by Peter Rousseeuw (Anja.Struyf@uia.ua.ac.be) (Mia.Hubert@uia.ua.ac.be) and initial R port by Kurt.Hornik@R-project.org, 2005) and vegan (Oksanen *et al.*, 2005) for cluster analyses. The significance level for hypothesis tests was $P = 0.05$.

3.4. Results

Habitat characteristics

Physico-chemical measurements in the littoral zone remained reasonably constant (mean \pm S.E.); turbidity 2.74 ± 0.11 NTU, dissolved oxygen 10.60 ± 0.27 mg L⁻¹, conductivity 278 ± 0.92 μ S cm⁻¹, pH 8.04 ± 0.06 , although there was a gradual decrease in water temperature from a mean of 14.6 °C during the first sampling occasion to 7.1 °C during the fifth occasion. The water level of the dam varied marginally within ± 60 mm during the entire experiment.

Profiles of the dam's shores are depicted in Fig. 3; gradients were all approximately linear with shallower gradients on the southern and eastern shores. Cluster analysis showed that aquatic vegetation in the littoral zone was relatively similar in terms of species composition, except for in the north-east corner (NE_1) of the dam, which had more species and a unique combination of plants compared to all other sites (Fig. 4a; Table 1). The amount of vegetative cover was fairly homogenous with two major groups of sites (Fig. 4b). The larger group, which included the diverse north-east corner site, had similar degrees of cover at both the water surface and the substrate (~80-100% cover in each), whereas the other group consisted of sites on the deeper littoral shores (i.e. sites E_3, E_5, SW_9, SE_6 and S_8) where cover at the substrate was 30-50% less than at the water surface. Inshore littoral areas (<1 m deep and <4 m from shore for southern and eastern shores and <3 m from shore for northern and western shores) were dominated by a mixture of submerged (*Myriophyllum* sp. and *Lilaeopsis polyantha*) and emergent (*Eleocharis acuta*) aquatic vegetation, while offshore littoral areas (1-1.6 m deep and <8 m from shore) were dominated by submerged vegetation (*Myriophyllum* sp. and *L. polyantha*).

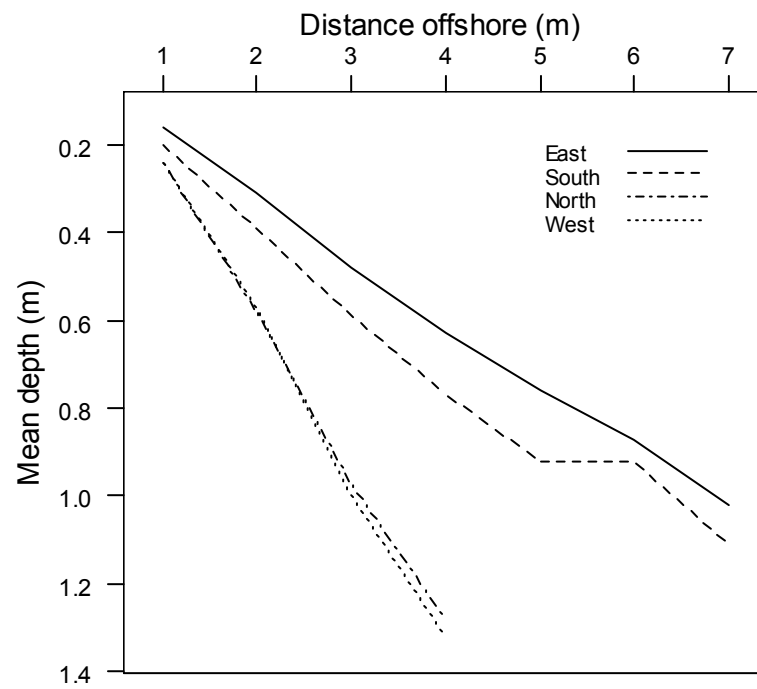


Fig. 3. Depth profiles of dam shorelines.

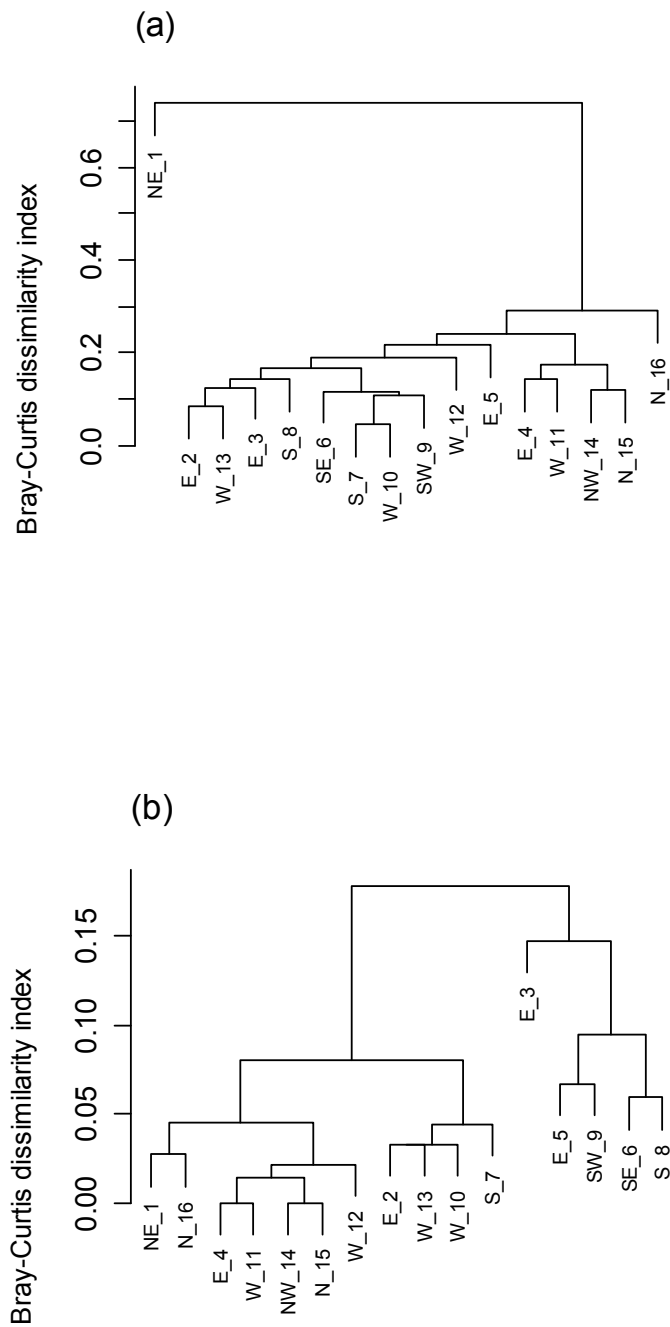


Fig. 4. Dendrograms of agglomerative hierarchical cluster analysis based on Bray-Curtis matrix of dissimilarities in (a) species richness and (b) cover at surface and substrate levels between vegetation survey sites. Sites at abbreviated by shore (E = east, N = north, NE = north-east corner, NW = north-west corner, S = south, SE = south-east corner, SW = south-west corner and W = west) and site number. Note different y-axis scales.

Table 1. Species composition of aquatic vegetation in the littoral zone of the dam.

Site ¹	Macrophyte species ²										No. of spp.
	C	EA	EG	LP	M	NA	P	PT	RA	TP	
NE_1	+	+				+		+	+	+	6
E_2		+		+	+						3
E_3		+		+	+						3
E_4		+		+	+						3
E_5		+		+	+						3
SE_6		+		+	+						3
S_7		+		+	+						3
S_8		+		+	+						3
SW_9		+	+	+	+						4
W_10		+		+	+						3
W_11		+		+	+						3
W_12		+		+	+	+	+				5
W_13		+		+	+	+					4
NW_14		+		+	+	+		+			5
N_15		+	+	+	+						4
N_16		+			+						2

¹Sites are abbreviated by shore (E = east, N = north, NE = north-east corner, NW = north-west corner, S = south, SE = south-east corner, SW = south-west corner and W = west) suffixed with site number.

²Macrophyte names are abbreviated as: *Characeae* sp. (C), *Eleocharis acuta* (EA), *Elatine gratioloides* (EG), *Lilaeopsis polyantha* (LP), *Myriophyllum* sp. (M), *Neopaxia australasica* (NA), *Poaceae* sp. (P), *Potamogeton tricarinatus* (PT), *Ranunculus amphitrichus* (RA), and *Triglochin procerum* (TP).

Comparison of fishing methods

Small numbers of juveniles were periodically observed feeding from the surface of the water at day time during the study. Snorkel observations at night revealed that juveniles generally occurred in shallow, inshore habitats (<0.6 m) containing a mixture of submerged and emergent vegetation, whereas adults were stationed in epibenthic microhabitats in deeper open water (0.6-1.2 m) above submerged vegetation. Adults were generally distributed evenly (approximately 0.4-1.0 m apart) within these littoral habitats. Both adults and juveniles were observed foraging for invertebrates in submerged vegetation at night. Aggressive territorial behaviour was observed between adult and juvenile fish, with larger fish chasing small fish away from their feeding area.

A total of 2388 *G. auratus* were sampled during the study (174 during the day and 2214 at night; pooled length range 60-171 TFL mm). No short-term mortality from sampling was observed and no other fish species were caught. There was a highly significant interaction between diel period and fishing method (omitting snorkelling: $F_{1,16} = 56.7$, $P < 0.001$) with greater catches at night for all other methods. The magnitude of this difference depended on fishing method (Fig. 5). There was no significant difference between day time fyke net catches and electrofishing ($P = 0.85$). At night, fyke netting caught ~3 fold more fish than either of the other methods ($F_{2,12} = 59.4$, $P < 0.001$).

Sampled fish comprised two distinct groups of size classes which were taken to represent juveniles (60-100 TFL mm) and adults (101-170 TFL mm). Pooled catch data for the three fishing methods yielded significantly different cumulative length-frequencies (all $P < 0.01$). Similarly, length-frequencies of catches for each method also varied with diel period (Fig. 6). There were significant differences in length frequencies between the day and night catches

from both electrofishing ($P < 0.05$) and fyke netting ($P < 0.001$). Electrofishing caught more juveniles at night and more adults during the day. Fyke netting caught predominately juveniles during the day, while at night there was a more balanced catch of juvenile and adult fish. No fish were observed by snorkelling during the day and fish observed at night were predominately larger-sized adult fish.

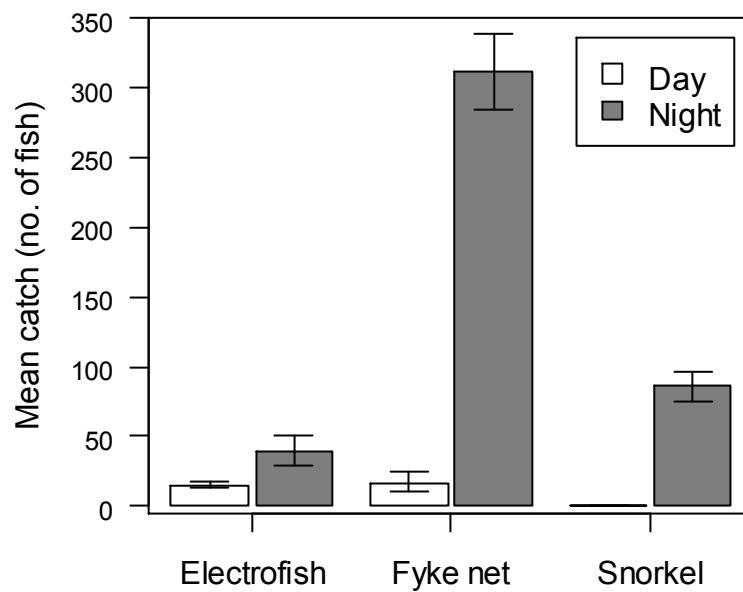


Fig. 5. Mean (± 1 S.E.) diel catches or observations of *Galaxias auratus* by electrofishing, fyke netting and snorkelling (no fish were observed by snorkelling during day time).

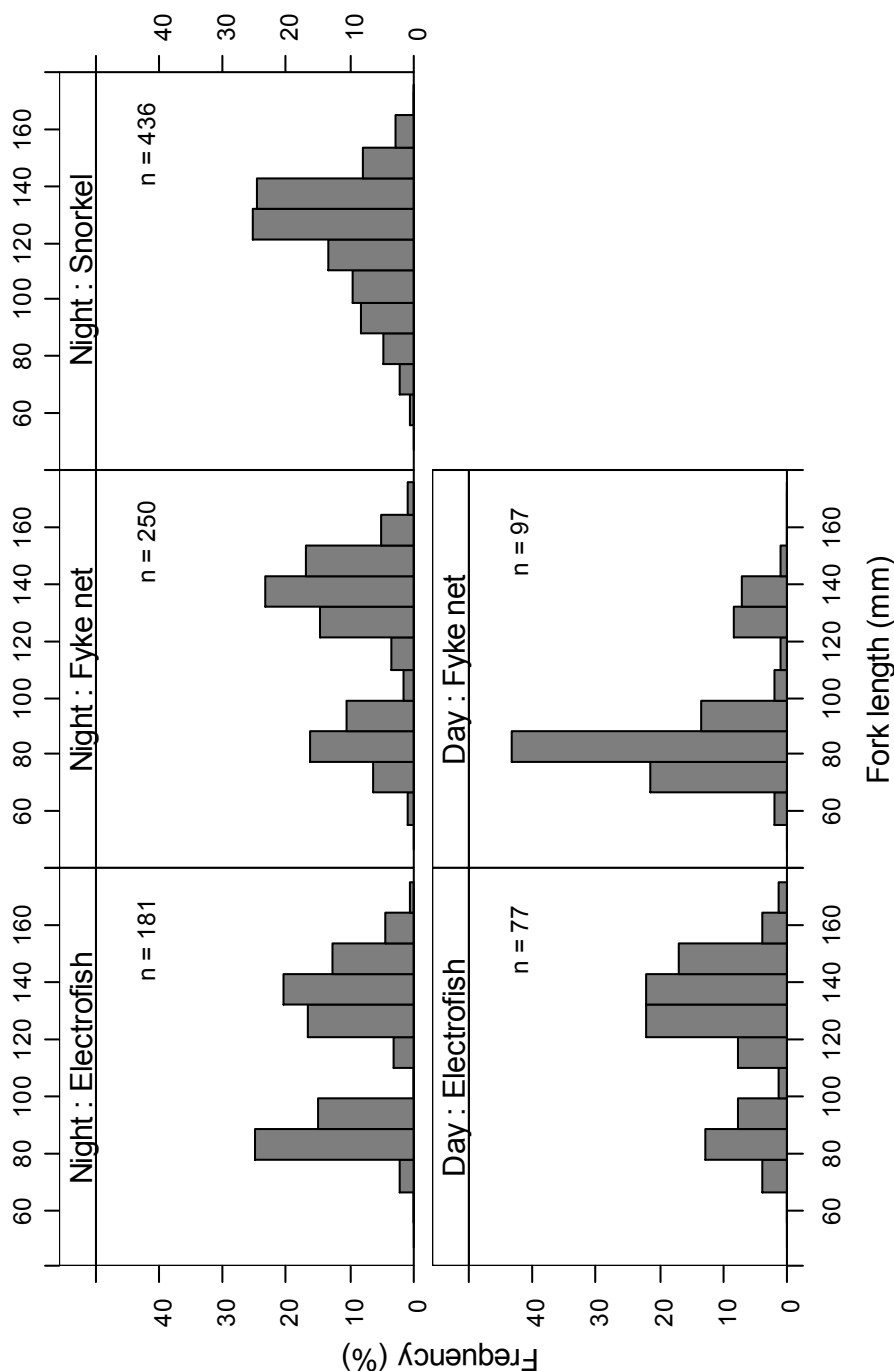


Fig. 6. Cumulative length frequencies of diel catches or observations of *Galaxias auratus* by electrofishing, fyke netting and snorkelling (no fish were observed by snorkelling during day time).

Habitat use and behaviour

Galaxias auratus appears to prefer shallow littoral habitats with gently sloping depth profiles as catches for all fishing methods, except for electrofishing during the day, were generally greater on eastern and southern shores (Fig. 7). The majority of fish (>90%) sampled by electrofishing during the day were captured in the north-east corner of the dam (Fig. 3 and Table 1; region surrounding site NE_1). Analysis of corrected catch data (fish m⁻¹ of shore) for night samples showed there was no significant interaction between method or shore ($F_{6,44} = 0.5$, $P = 0.8$), but there were differences between methods ($F_{2,44} = 112.71$, $P < 0.001$) and shores ($F_{3,44} = 14.56$, $P < 0.001$) (Fig. 7). Tukey's tests for each of these main effects revealed that all pairwise comparisons were significantly different between fishing methods, while amongst the shores, the eastern shore had significantly more fish m⁻¹ than all other shores, and the southern shore had more fish m⁻¹ than the northern shore. For day samples, fish m⁻¹ for each shore differed depending on the method employed (shore \times fishing method interaction: $F_{3,23} = 6.37$, $P = 0.01$). Tukey's tests within each fishing method showed no significant differences between shores for fyke nets, while the eastern shore had significantly more fish m⁻¹ than the western shore (Fig. 7).

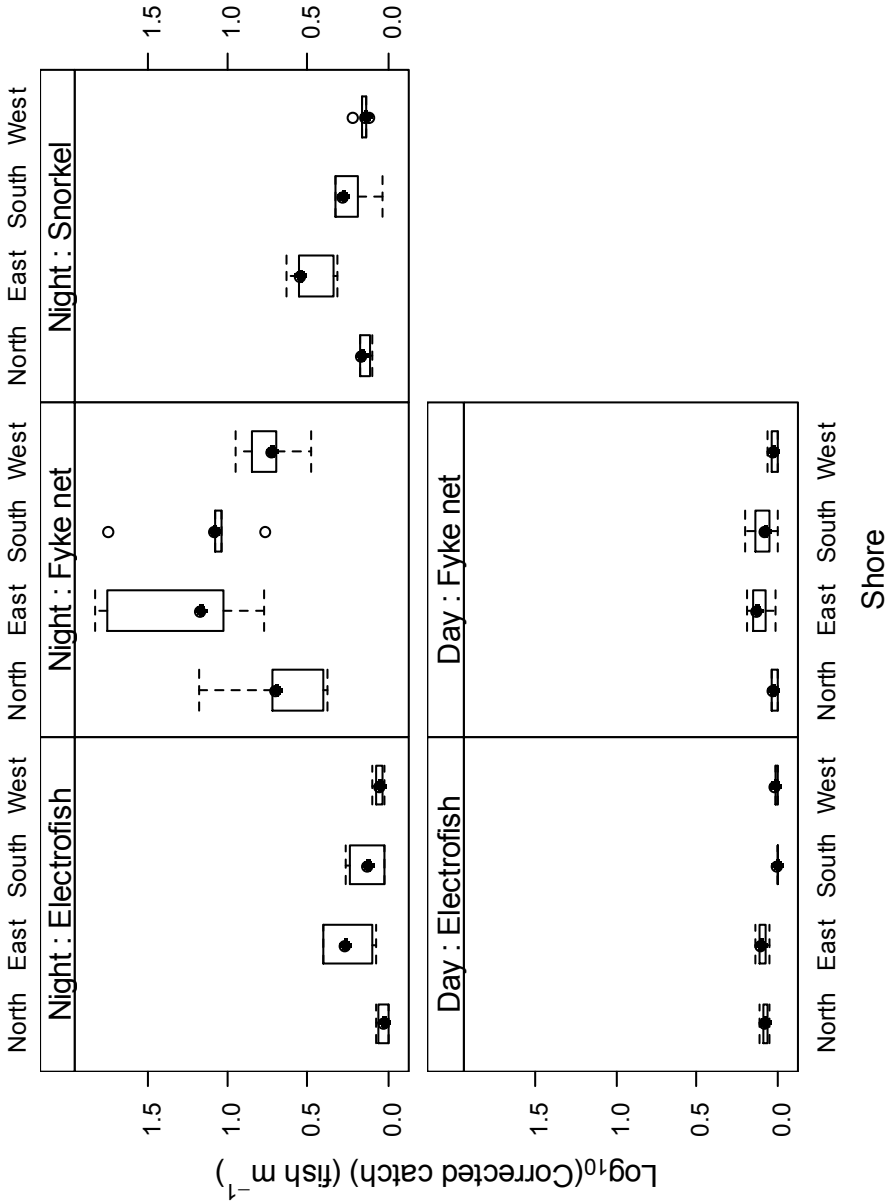


Fig. 7. Boxplots of catches or observations of *Galaxias auratus* per shore by electrofishing, fyke netting and snorkelling. Dots represent the median and open circles denote values that lie outside the range spanned by the whiskers (no fish were observed by snorkelling during day time).

3.5. Discussion

Evaluation of sampling methods

Ideally, regimes for monitoring the status of populations of threatened fishes should use efficient fishing methods which provide catches that: (1) proportionally represent all size classes (i.e. are not selective), and (2) are sufficiently sensitive to enable early detection of alterations in population size. Comparisons of the catches from each fishing method in this study are limited by not knowing the absolute abundance or size structure of the *G. auratus* population sampled. However, given the relatively small size of the dam and the large effort employed for each fishing method (e.g. electrofish and snorkel entire littoral zone), direct comparisons between methods should provide a useful assessment, at least, of the sensitivity of each technique.

This study has shown that sampling at night using either fyke netting, electrofishing or snorkelling yielded larger catches of *G. auratus* than during the day. Fyke netting at night was the most efficient sampling regime as it produced the largest catches (mean = 313 fish session⁻¹); however, the proportion of juveniles was lower in night time fyke catches than in the day time, which may suggest some differential activity patterns between adults and juveniles. Clearly, this needs to be resolved further by more intensive diel sampling experiments. Day time fyke netting captured few adults and only small numbers of juveniles (mean = 19 fish session⁻¹). Therefore, it appears that night time fyke netting is a more efficient method for sampling the *G. auratus* population of the dam. The efficiency of night time fyke netting for *G. auratus* is likely to be due to increased activity in all size classes in the littoral zone at night, but could also be partly attributed to fish seeking the cover provided by nets. The high catchability of cover-oriented fish species has been related to the ability of fyke nets to provide cover in other studies (Krueger *et al.*, 1998; Rogers *et al.*, 2003).

Mesh size can be used to adjust the size selectivity of fyke net catches (Kraft & Johnson, 1992; Chisnall & West, 1996). The fine mesh (5 mm diameter) used in this study enabled the capture of an extensive size range of galaxiids (64-171 TFL mm). Although the effort to set and retrieve fyke nets was greater than that needed for the other methods (i.e. requiring two site visits and time to clean and dry nets before storage), the results obtained by fyke netting at night justify this extra effort.

The design of galaxiid population monitoring regimes incorporating fyke netting need to avoid bias due to seasonal variation in catches. This has occurred in wild populations of *G. auratus* with temporal catches increasing during spawning (winter – early spring) (Hardie *et al.*, 2005). However, based on our results, sampling in late summer – autumn with fyke nets is recommended for monitoring the abundance and size structure of the galaxiid population in the dam sampled. Fine-meshed fyke nets have been effective for assessing community or population structure of benthic fishes in lakes elsewhere (Krueger *et al.*, 1998) and their use in combination with other methods has been recommended for sampling littoral fish assemblages (Weaver *et al.*, 1993; Fago, 1998). Our results further support these recommendations.

Electrofishing caught smaller samples of fish than night time fyke netting at both day (mean = 15 fish session⁻¹) and night (mean = 42 fish session⁻¹). Night time electrofishing was more effective than day time as it captured larger numbers of juvenile and adult fish. However, we observed that the efficiency of electrofishing at day and night was very poor, due to thick vegetation hindering the capture of stunned fish after their initial detection. Therefore, catches under represented the abundance of fish present in the littoral zone.

Night electrofishing has been reported to more efficient than day electrofishing for fish in riverine (Paragamian, 1989) and lake (McInerny & Cross, 2000) habitats. Electrofishing is biased towards larger size fish (Zalewski, 1983) and it is also more effective in shallower water where avoidance of the electric field is more difficult (Reynolds, 1996). In this study, juvenile *G. auratus* were thought to be more susceptible to electrofishing at night, as they occupied the shallower margins (<0.6 m). Day time electrofishing is commonly used to monitor Tasmanian galaxiid populations in lentic waters (Threatened Species Section, 2006). Electrofishing may be effective for presence/absence surveys; however, abundance and size structure data from electrofishing samples should also be treated with caution given this techniques' bias towards larger fish and the effluence of habitat-related variables (i.e. fish habitat use and poor catchability in dense vegetation) had on catches in this study. If backpack electrofishing is to be used to monitor galaxiid populations, then sampling at night should be considered.

Snorkelling in the day was ineffective for monitoring the *G. auratus* population, but night sampling encountered many fish (mean = 87 fish session⁻¹). No fish were observed by snorkelling during the day, which, based on the results of the other methods, accurately reflects the small number of galaxiids in open water and sparsely vegetated areas in the littoral zone during the day. We also suspect that the small number of fish that may have occupied these habitats during the day were more likely to be frightened by the diver in day light and sought shelter before detection.

The size structure of fish observed when snorkelling at night was strongly biased towards adults (>100 TFL mm). Underwater magnification of the actual size of observed fish (Mullner *et al.*, 1998) was not accounted for in this study and may be partly responsible for this bias.

However, visual observations indicated that areas of deeper water beyond the littoral beds of emergent vegetation were favoured by larger fish. This region was not sampled as effectively by the other methods, particularly electrofishing. Conversely, shallower areas that contained denser stands of emergent vegetation which are probably occupied predominately by juveniles were poorly sampled by snorkelling due to the cover provided to fish. Based on our results, depending on suitable water clarity, snorkelling at night could be used to assess presence/absence and the abundance of adult fish in a given population, but should not be relied upon to determine the year class strength of juveniles or overall population structure in galaxiid monitoring programs.

Diel activity, habitat use and behaviour

Diel shifts in habitat use are common in small-sized fish in lentic freshwater habitats and are generally thought to be linked to predator avoidance (Fuiman & Magurran, 1994) and feeding strategies (Pitcher, 1993). Differences in the abundance and size structure of catches at day and night in this study enabled the diel activity patterns of *G. auratus* to be examined. Catches were significantly greater at night using all sampling methods (passive and active) so it would appear that *G. auratus* are either nocturnally active or many fish move inshore and concentrate in the littoral zone at this time. Several other galaxiid species are also nocturnally active (Glova & Sagar, 1989; Rowe & Chisnall, 1996b; David & Closs, 2003).

Galaxias auratus show a strong diel shift in habitat use. During the day, the majority of fish appeared to retreat to deeper water with small numbers of juveniles remaining in sparsely vegetated littoral habitats and some adults sheltering in the dense species rich vegetation in the north-east corner of the dam. Diel fyke net and electrofishing catches indicate that, at night, larger numbers of both juveniles and adults moved into littoral areas, with much greater

catches of both size classes at this time. Higher catches on shores with shallow depth profiles (southern and eastern shores) indicated a preference for these habitats. This may be a function of increased catchability to the methods used in shallower water. Fyke nets are most efficient in shallow water and are generally set where the float line is at the surface to prevent fish swimming over the wing (Rogers *et al.*, 2003), while the efficiency of backpack electrofishing is known to decrease with an increase in water depth (Reynolds, 1996). However, our snorkel observations indicated *G. auratus* had a preference for shallow shores where submerged vegetation was abundant in comparison to deeper shores where barren substrate was dominant.

Cover provided by aquatic macrophytes influences predator-prey interactions in freshwater fish communities (Savino & Stein, 1989; Eklöv, 1997) and also diel activity patterns of prey fish (Jacobsen & Berg, 1998). The structural complexity of macrophytes may also influence the distribution of fishes in lakes with small-sized fish favouring these habitats (Chick & McIvor, 1994; Weaver *et al.*, 1997). The preference of some adult *G. auratus* during the day for littoral vegetation in the north-east corner of dam is likely to be due to the greater and more structurally complex cover provided by vegetation in this area. Food resources were not assessed in the dam, but vegetated aquatic habitats usually contain higher abundances of macroinvertebrates than adjacent unvegetated areas (Crowder & Cooper, 1982). Adults and juveniles were observed feeding at night when snorkelling with stationary fish foraging for macroinvertebrates in submerged vegetation. Therefore, movement of *G. auratus* from offshore areas to the vegetated littoral zone appears to be feeding related and may take advantage of an increase in the activity of invertebrates which could also seek shelter during day and be more vulnerable to predation at night.

No other fish species occur in the dam used in this study therefore the behaviour and activity patterns observed have occurred without competition and predation pressures from other aquatic species. However, avian predators can alter the abundance and size distribution of teleost prey (Gliwicz & Warsaw, 1992; Steinmetz *et al.*, 2003). The piscivorous waterbird, white-faced heron (*Egretta novaehollandiae*), was frequently observed at the dam during the day, and it is suspected that cormorants (*Phalacrocorax carbo*), also a piscivorous species, may visit the dam periodically. Therefore, the diel movement between habitats observed in this study may be partly due to the threat from avian rather than aquatic predation.

Management implications

This study has compared three fishing methods for sampling *G. auratus* in a small closed lentic system; however, the habitat attributes and fish community structure of the dam may differ from those of larger lakes occupied by other lacustrine galaxiids. For example, many lentic waters on the TCP that are inhabited by threatened galaxiid populations are deeper, have several littoral substrate types (e.g. rocks, sand, aquatic vegetation and woody debris) and also contain abundant populations of introduced salmonids which are known to heavily prey upon galaxiids (Crown *et al.*, 1992; Stuart-Smith *et al.*, 2004). Given these differences in habitat, and competition from and predation by salmonids, the behaviour and activity patterns of other lacustrine galaxiids may vary from those observed here. Therefore, the efficiencies of the sampling methods used in this study are also likely to differ. Further work is required to investigate the effects of habitat and fish community structure on activity patterns of galaxiids in lentic waters. Examination of these factors will help determine the catchability of galaxiids in different habitats in relation to various methods. This information is needed to index galaxiid abundance and population status using catch per unit effort (CPUE) data collected by routine monitoring. Research in this area should lead to predictive models of abundance based

on CPUE data that would enable accurate monitoring of threatened galaxiid populations with minimal effort.

4. Spawning related seasonal variation in fyke net catches of golden galaxias (*Galaxias auratus*): implications for monitoring lacustrine galaxiid populations*

*Published (with minor adjustments) as Hardie S.A., Barmuta L.A. & White R.W.G. (2005) in *Fisheries Management and Ecology* 12, 407-409.

Key words: abundance; catchability; fyke net; Galaxiidae; population monitoring; spawning

4.1. Introduction

Fishes belonging to the family Galaxiidae are relatively small (usually <300 mm long), freshwater-dwelling teleosts which are found on several land masses in the Southern Hemisphere (McDowall & Fulton, 1996). Many galaxiid species have non-diadromous life histories and the distribution of some of these consists of discrete lacustrine populations (McDowall, 2000; Allen *et al.*, 2002). Due to restricted or fragmented distributions, along with other impacts (e.g. introduced fish species), many lacustrine galaxiid species are considered to be threatened; hence, monitoring populations for conservation purposes is a common management practice.

Fyke nets are known to be effective for capturing small-sized benthic or littoral fish in lakes (Fago, 1998; Krueger *et al.*, 1998), including galaxiids (Rowe & Chisnall, 1997a; Rowe *et al.*, 2002a). Fyke net catches of Atlantic salmon, *Salmo salar* L., in lakes vary seasonally, suggesting relative abundance indices based on catch data often require correction for differences in catchability (Ryan, 1984). Seasonal fluctuations in fyke net catches of

European perch, *Perca fluviatilis* L., have also been reported due to spawning-related habitat shifts (Kubecka, 1992). Other studies have shown that cover-orientated fish species tend to have increased catchability by fyke nets because fyke nets provide cover (Krueger *et al.*, 1998; Rogers *et al.*, 2003).

Knowledge of seasonal changes in catchability of target species to a given type of fishing gear is critical to the application of that technique for population monitoring. Populations of threatened freshwater fish are infrequently monitored (i.e. seasonally or annually) to determine their status, therefore, sampling regimes need to ensure catches are not biased by seasonal variations in behaviour or habitat use. While fyke nets have previously been used to monitor lacustrine galaxiid populations (Threatened Species Section, 2006), seasonal variation in catches needs to be examined to avoid the confounding influence of such variation.

The golden galaxias, *Galaxias auratus* (Johnston), is endemic to lakes Crescent (42°10'S, 147°10'E) and Sorell (42°6'S, 147°10'E) on the Central Plateau of Tasmania, Australia. These lakes are located at an elevation of *c* 800 m a.s.l., and are relatively shallow (<3.5 m) and turbid (mean of 130 NTU during study). *Galaxias auratus* is threatened by alterations to water level regimes, and competition and predation from introduced fish species (Hardie *et al.*, 2004).

4.2. Materials and methods

As a component of recent work to examine the current status and ecology of *G. auratus* populations in lakes Crescent and Sorell, both populations were sampled monthly using fyke nets between October 2000 and December 2002 (Hardie *et al.*, in press; Chapters 5 and 8).

The data obtained enabled seasonal variation in catches to be investigated. The same sampling techniques and effort were applied in each lake on each sampling occasion. Twelve fine-meshed fyke nets (2 mm stretched mesh) were set overnight (mean soak time of 18 h) among three littoral sites which represented the dominant substratum types (cobble, sand, fine sediment) in each lake (i.e. four nets per site). The entrances of fyke nets were fitted with screens (84 × 70 mm aluminium mesh) to avoid the capture of platypus (Grant *et al.*, 2004) and waterbirds (Beumer *et al.*, 1981). Differences in catches between lakes were examined using mixed model ANOVA and *a priori* contrasts were used to test whether catches during spawning periods differed from intervening non-spawning periods. Inspection of residuals suggested that the count data be log transformed; this transformation was successful in removing heteroscedasticity and yielding approximately normally-distributed residuals.

4.3. Results and discussion

Catches of *G. auratus* were always greater in Lake Crescent than in Lake Sorell (Fig. 1). During late autumn – early spring (May-September in Lake Sorell and June-October in Lake Crescent), *G. auratus* deposited eggs, which were approximately 1.5 mm in diameter and adhesive, on the fine mesh of fyke nets during sampling (Figs. 1 & 2). To verify that eggs were deposited due to spawning activity (i.e. fertilisation took place) a sample of eggs was incubated in an aquarium until hatching occurred. During the 2001 and 2002 spawning periods in Lake Crescent and in the 2002 period in Lake Sorell, catches increased substantially and differed significantly from intervening non-spawning periods (Fig. 1: Lake Crescent (all $P < 0.01$), Lake Sorell (all $P < 0.05$)). This increase in catches also coincides with other measures of the reproductive cycle of *G. auratus* (Hardie *et al.*, in press): during winter – early spring the gonadal somatic index (ratio of gonad mass to body mass) of *G. auratus*

peaks, fish with gonads classed as post-spawning were present in both populations at this time, and eggs were found on rocky shores.

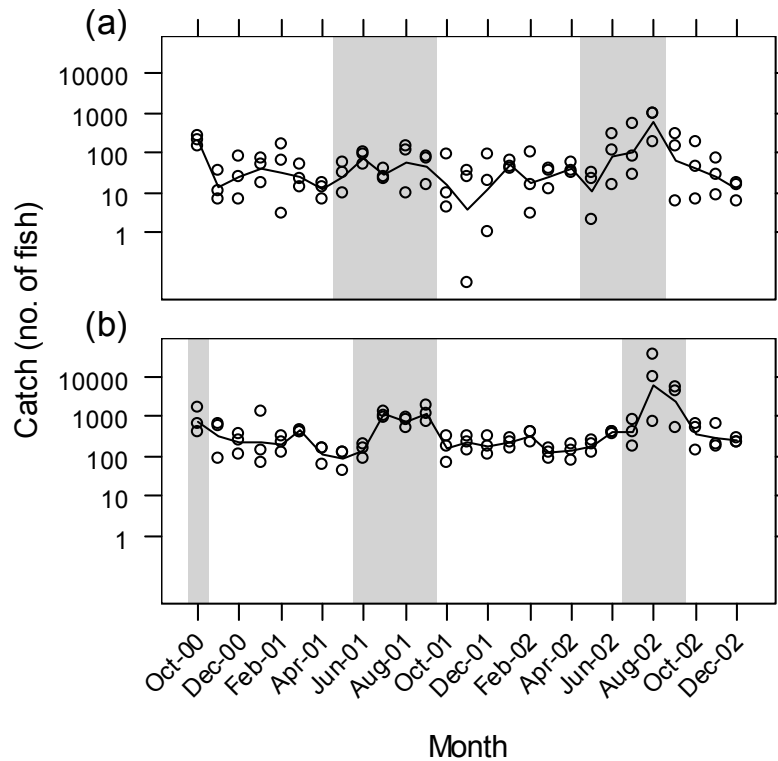


Fig. 1. Monthly fyke net catches of *Galaxias auratus* from each site in (a) Lake Sorell and (b) Lake Crescent. Catch at each site is indicated with open circles and the mean monthly catch is connected by solid lines. The occurrence of *G. auratus* eggs on fyke net mesh during sampling is indicated by grey shading.

Galaxias auratus probably has a higher catchability with fyke netting than with other sampling methods, including shore-based backpack electrofishing and seine netting (Hardie *et al.*, 2006a). This may partly be due to an attraction towards fyke nets because of the cover they provide. However, the increase in catches that occurred during winter-spring could also result from fish being attracted to the nets as a spawning substrate, in addition to a general increase in the abundance of *G. auratus* in the littoral zone during this time.



Fig. 2. *Galaxias auratus* eggs deposited on a fine-meshed fyke net that was used to sample fish during August 2001 (sampler: Scott Hardie).

It appears that spawning-related seasonal changes in behaviour can alter the catchability of adult *G. auratus* by fine-meshed fyke nets. Consequently future monitoring regimes for *G. auratus* should avoid sampling during spawning periods so that abundances are not overestimated. The results of this study suggest that monitoring programmes should not sample *G. auratus* populations between May and October (inclusive). This will prevent spawning-related increases in catches causing over-estimated relative abundance indices.

Excluding catch data from spawning periods (May to October), there were on average over 7.7 times more galaxiids in Lake Crescent than in Lake Sorell (269 fish month⁻¹ in Lake Crescent vs 34 fish month⁻¹ in Lake Sorell; $F = 62.2$; d.f. = 1 and 4; $P = 0.0014$). This difference in mean catch probably represents the difference in relative abundance between the

two populations, which is suspected to be due to different rates of predation by introduced salmonids and differences in primary productivity between the lakes. Lake Crescent is more productive (Cheng & Tyler, 1976b) and has less abundant salmonid (brown trout, *Salmo trutta* L., and rainbow trout, *Oncorhynchus mykiss* (Walbaum)) populations (Inland Fisheries Service, Tasmania unpubl. data) than Lake Sorell, therefore, the capacity to support a more abundant galaxiid population.

Tasmanian inland waters provide a regional stronghold for galaxiids in Australia, with five genera and 16 species (11 of which are endemic), but 69% of species are considered to be threatened (Hardie *et al.*, 2006b). Nine of the 11 endemic galaxiid species primarily occur in lentic waters and have non-diadromous life cycles. Fyke netting is currently used to monitor populations of some of these species (J. E. Jackson, pers. comm.) and may be an effective monitoring method for most populations. However, fisheries managers in Tasmania and other regions should be cautious when interpreting relative abundance indices based upon fyke net catches of lacustrine galaxiid populations unless seasonal variation in catchability is known. Data on the reproductive biology of target species is one aspect of their life history that may improve the design of monitoring programmes.

5. Age, growth and population structure of the golden galaxias (*Galaxias auratus* Johnston) in lakes Crescent and Sorell, Tasmania, Australia*

*To be submitted to *Marine and Freshwater Research* following minor adjustments.

5.1. Abstract

Age of *Galaxias auratus* ($n = 600$) from lakes Crescent and Sorell, Tasmania was estimated by counts of opaque zones in thin-sections of sagittal otoliths. Age 0+ fish were assigned ages based on theoretical birth dates and sample dates, and the progression of mean cohort lengths was used to describe their growth. The oldest male and female *G. auratus* analysed were 10 and 7 years respectively. Length-at-age data showed two distinct phases in the growth of juveniles and adults (transition at ~60 mm fork length). Von Bertalanffy growth models fitted length-at-age data of adults, but did not accurately describe the growth of juveniles. Females grow faster and have greater longevity than males, and both sexes in Lake Sorell grow faster than those in Lake Crescent. Fish in the 0+, 1+ and 2+ year classes dominated the age structure of both populations, but there were more >2+ in Lake Crescent. Differences in habitat availability, food resources, and competition and predation interactions with introduced salmonids are likely to influence the growth and age structure of *G. auratus* populations. Management strategies should focus on minimising short-term impacts such as reductions in lake levels, recruitment failures or increases in the abundance of salmonids.

Key words: Tasmanian Central Plateau; Galaxiidae; lacustrine populations; non-diadromous life history; threatened species

5.2. Introduction

The Galaxiidae are found in the Southern Hemisphere (Allen *et al.*, 2002), and the family accounts for a significant proportion of the freshwater fish fauna of southern Australia (McDowall & Frankenberg, 1981). The island of Tasmania, in southern Australia, provides a regional stronghold for the family with a diverse galaxiid fauna containing five genera and 16 species (64% of Tasmanian native freshwater species), of which 11 are endemic (Hardie *et al.*, 2006b). Three Tasmanian galaxiid species (*Galaxias maculatus*, *G. truttaceus* and *G. brevipinnis*) are also common in nearby southern mainland Australia and in New Zealand, and some aspects of their biology have been reported (McDowall & Fulton, 1996; McDowall, 2000). However, biological information is limited for most Tasmanian endemic species and 10 of the 11 species are considered to be ‘threatened’ (i.e. protected under State or Federal Australian threatened species legislation) (Hardie *et al.*, 2006b).

The golden galaxias (*Galaxias auratus* Johnston) is endemic to the interconnected lakes Crescent and Sorell and their associated streams and wetlands in the south-east of the Tasmanian Central Plateau (TCP) (Hardie *et al.*, 2004). This species has a non-diadromous life history with spawning occurring during late autumn – winter and larvae emerging in winter-spring and remaining pelagic until 4-5 months of age (Hardie *et al.*, 2004; Hardie *et al.*, in press; Chapter 8). It is also one of the largest galaxiid species of the TCP growing to 279 mm in length (Johnston, 1883). *Galaxias auratus* closely resembles both the spotted galaxias, *G. truttaceus*, and saddled galaxias, *G. tanycephalus*, and is thought to be a land-locked genetic derivative of the ancestral version of *G. truttaceus* (Johnson *et al.*, 1981; Johnson *et al.*, 1983; Ovenden *et al.*, 1993). Since being described over 120 years ago (Johnston, 1883), the biology of *G. auratus* has remained largely unstudied. This species is listed under State (Tasmanian *Threatened Species Protection Act 1995*) and Federal

(Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*) threatened species legislation due to its highly restricted distribution, and threats associated with alterations to water level regimes and introduced fish species (i.e. *Salmo trutta*, *Oncorhynchus mykiss* and *Cyprinus carpio*) (Hardie, 2003).

Clearly, knowledge of growth parameters and age structure is needed to effectively monitor and protect populations of *G. auratus* and other Tasmanian endemic galaxiids, which are currently managed under a threatened species recovery plan (Threatened Species Section, 2006); without this information, the status of populations and their viability can not be accurately assessed. Calcified body parts, such as sagittal otoliths, are often used to age teleost fish species as they incorporate growth-related incremental structure (DeVries & Frie, 1996). Researchers have highlighted the need to validate (Beamish & McFarlane, 1983) and identify the precision and accuracy (Campana, 2001) of age estimates. Several factors including seasonal water temperature cycles, feeding patterns, and reproductive cycles influence somatic and otolith growth as well as the formation of annual growth rings (annuli) (Beckman & Wilson, 1995).

Several studies focusing on both adult (Eldon, 1978; Humphries, 1989; Bonnett, 1990; Pen & Potter, 1991b; Morgan, 2003) and juvenile (Cussac *et al.*, 1992; McDowall *et al.*, 1994; McDowall & Kelly, 1999) galaxiids have examined otoliths to derive age estimates. However, to our knowledge examination of thin-sections, as opposed to whole or cracked otoliths, has not previously been undertaken on galaxiids. This technique is commonly used to estimate the age of larger-sized Australian freshwater fishes (Mallen-Cooper & Stuart, 2003; Brown *et al.*, 2004; McDougall, 2004) and may be applicable to smaller species. In some fishes, otolith measurements (e.g. weight, length and width) are more strongly correlated with age than fish

length (Boehlert, 1985) and, therefore, may also be used to estimate age and growth more cheaply (Pawson, 1990; Francis & Campana, 2004). These alternative methods have not previously been assessed in galaxiid species.

The aims of our study were to: (i) use an otolith-sectioning technique to derive age estimates for *G. auratus* from Lake Crescent and Lake Sorell based on otolith incremental structure, (ii) validate that the number of opaque zones on transverse otolith sections is related to age, (iii) examine age-somatic growth, and age-otolith growth relationships for sexes from both populations, and (iv) determine the age structure of populations. Based on these findings, we discuss implications for conservation-based management strategies for *G. auratus* and assess the feasibility of this technique for examining the age and growth of other galaxiid species.

5.3. Materials and methods

Study site

The interconnected Lake Crescent (42°10'S, 147°10'E) and Lake Sorell (42°6'S, 147°10'E) lie 1 km apart and are situated at ~800 m a.s.l. in the south-east of the TCP, Australia (Fig. 1). Both lakes were natural lentic waters that have had their levels raised on numerous occasions since the 1830s for water storage purposes. Lakes Crescent and Sorell have areas of 23 and 52km², respectively, mean depths <3.5 m at full supply levels), and during this study they had mean turbidities of ~130 NTU (Uytendaal, 2003). The lakes are very similar physically and chemically and are located in an area of uniform geology, climate, soils and vegetation (Cheng & Tyler, 1973). However, the trophic status of these lakes differs, with Lake Sorell being mesotrophic and Lake Crescent moderately eutrophic (Cheng & Tyler, 1976a). Lake Crescent has a phytoplankton standing crop biomass 10 times that of Lake Sorell (Cheng & Tyler, 1976b).

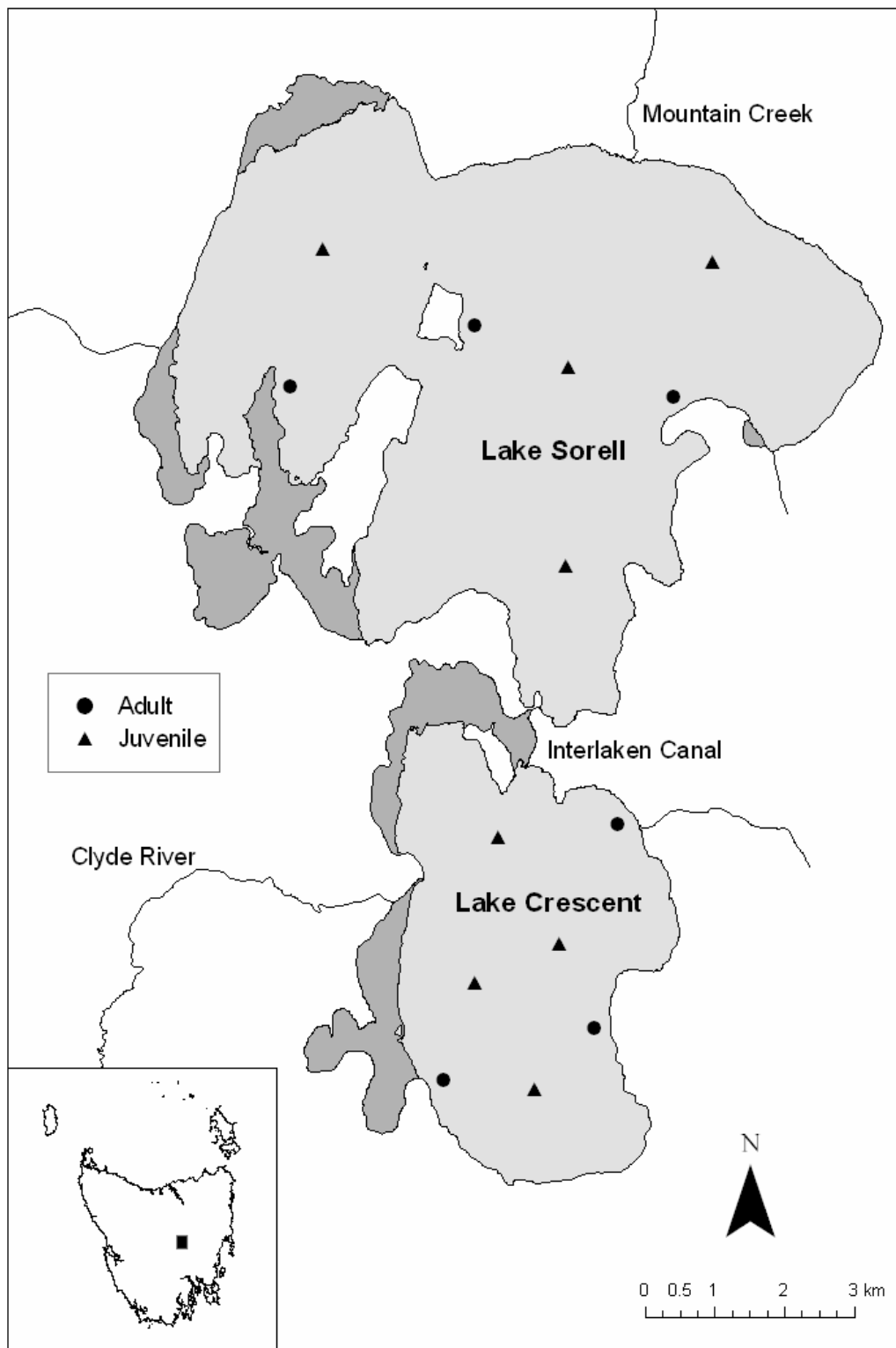


Fig. 1. Location of adult and juvenile *Galaxias auratus* sampling sites in lakes Crescent and Sorell, Tasmania, Australia. Lake perimeters are at full supply levels. Wetland areas are depicted by dark shading.

The fish communities of these lakes comprise two native species, the endemic *G. auratus* and indigenous short-finned eel (*Anguilla australis*), and three exotic species, brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*) and common carp (*Cyprinus carpio*). *Salmo trutta* and *O. mykiss* were first introduced into the lakes in 1868 and 1922 respectively, and *C. carpio* was first discovered in the lakes in 1995 (IFS, 2004). Salmonids are routinely stocked into Lake Crescent and *A. australis* into both lakes to support their respective recreational and commercial fisheries, whilst *C. carpio* are regarded as a pest and currently subject to an eradication program (IFS, 2004). The salmonid populations in Lake Sorell, which are self-sustaining, are several orders of magnitude more abundant than the populations in Lake Crescent (Inland Fisheries Service, Tasmania, unpubl. data).

Water temperature

Mean daily in-lake water temperature was measured from October 1997 to December 2002 in Lake Crescent and Lake Sorell using Optic StowAway Temperature loggers (Model WTA). Temperature loggers were deployed at a depth of 0.5 m at central sites in each lake. The difference between the water temperature regimes of the lakes was examined using a paired *t*-test of the mean monthly temperatures.

Pelagic juvenile fish sampling

Juvenile *G. auratus* were sampled monthly from Lake Crescent and Lake Sorell between July 2000 and December 2002. Juveniles were collected at four sites in each lake (Fig. 1) by towing a conical ichthyoplankton net (400 mm diameter, 1.25 m tail with 500 μ m mesh) for 10 min, approximately 15 m behind a boat at a speed (~ 8 km h⁻¹) that ensured the net was sampling the top 1 m of the water column. Catches from pelagic sites were generally comprised of larvae and juveniles (<70 mm fork length, FL). All material collected in

ichthyoplankton samples was euthanased in an anaesthetic solution (Aqui-s™) and preserved in 70% ethanol.

Littoral adult fish sampling

Juvenile and adult *G. auratus* (>40 mm FL) were sampled monthly from littoral habitats in Lake Crescent and Lake Sorell between July 2000 and December 2002. Fish were collected at three sites in each lake (Fig. 1) which represented one of the dominant littoral habitats (fine sand, cobble rock and sediment) in both lakes at the time of the study. On the first three sampling occasions fish were sampled by electrofishing 100 m of shoreline at each site with a backpack electro-fisher (Smith and Root 12-B POW). Due to the poor efficiency of electrofishing in the turbid water of these lakes (i.e. difficulty in detecting stunned fish), on the remaining sampling occasions fish were sampled using fyke nets. Four fyke nets (two 3 m × 0.6 m wing and two 5 m × 0.6 m wing nets with 600 mm D shaped entrance and 2 mm stretched mesh) were set in tandem at each site (12 nets per lake). All nets had a 84 × 70 mm aluminium screen in the entrance to avoid the capture of platypus (*Ornithorhynchus anatinus*), water birds and larger fish species (e.g. salmonids). Nets were set overnight (mean soak time of 18 hours) in shallow margins (depths ≤1.2 m). Catches of *G. auratus* were anaesthetised in an anaesthetic solution (Aqui-s™) prior to handling. A sub-sample of fish from each lake (Lake Crescent: $n = 43$ to 77 ; Lake Sorell: $n = 10$ to 70) representing the size range of fish captured, was euthanased in an anaesthetic solution (Aqui-s™) and preserved in 70% ethanol. The remaining captured fish were revived in fresh water and released at the site of capture.

Laboratory procedures

The FL (to the nearest 1 mm) of the first 100 *G. auratus* larvae and small juveniles (≤35 mm FL) and all large juveniles (≥36 mm FL) in each ichthyoplankton sample was

measured (Lake Crescent: $n = 3912$, Lake Sorell: $n = 4242$). The FL (to the nearest 1 mm) and total weight (TW, to the nearest 0.1 g) of *G. auratus* collected from the littoral habitats was measured (Lake Crescent: $n = 1624$, Lake Sorell: $n = 1354$), and fish were sexed by microscopic examination of gonads. Dissected fish from both populations covered large size ranges: Lake Crescent, 38-235 mm (FL) and 0.3-173.4 g (TW); Lake Sorell, 31-179 mm (FL) and 0.1-63.3 g (TW). The sagittal otoliths (hereafter referred to as otoliths) of 10 fish representing the size range of each monthly littoral sample from each lake were removed, cleaned in 70% ethanol and dried. Both otoliths from each fish were weighed and the average weight (OW, to the nearest 0.1 mg) was recorded. One otolith from each fish was randomly selected and sectioned using lapidary diamond saw and analysed following the methods of Ewing *et al.* (2003). Up to six transverse thin-sections (widths $\sim 300\ \mu\text{m}$) were obtained from each otolith. All specimen material, except removed otoliths, was lodged at the Tasmanian Museum and Art Gallery, Hobart, Tasmania (catalogue numbers D3756-D3774 and D3778-D3825).

Otolith reading procedures

A primary reader examined all otolith sections ($n = 600$) on a single occasion and a random selection of sections ($n = 200$) on a second occasion. A secondary reader examined half ($n = 100$) of the selected sections that were analysed twice by the primary reader. Completed opaque zones in otolith sections were counted along an axis parallel to the ventral side of the sulcus acusticus at $\times 31$ magnification without reference to size of fish nor date of sampling. Each otolith reading was given a clarity score from 1 to 5, with 1 being 100% reliable and 5 being unreadable. All grade 5 sections were omitted from further analysis (10% of otolith sections). The location of the first opaque ring in older fish (>2 opaque zones) was often difficult to identify as its separation from the primordium was unclear. Therefore, to guide

ring counts in older fish, the radius from the centre of the primordium to the first opaque zone was measured (along the same axis as opaque zone counts) on a sub-sample of sections ($n = 60$) with 1 or 2 opaque zones which had clarity scores of 1. The precision of age estimates (both inter-reader and intra-reader variability) were quantified using the index of average percent error (APE) (Beamish & Fournier, 1981). Reader variability was also examined visually using age-bias plots (Campana *et al.*, 1995) and slope estimates of these relationships were derived by regression analysis.

Validation of aging technique

The periodicity of the formation of opaque and translucent zones in otolith sections was examined along the same axis as opaque zone counts at $\times 63$ magnification using two forms of marginal increment analysis (MIA): (i) qualitative scores of the status of the otolith outer margin (Mallen-Cooper & Stuart, 2003), and (ii) quantitative measures of the marginal increment index of completion (Campana, 2001). Otolith sections were randomised prior to MIA and these analyses were undertaken without reference to size of fish nor date of sampling. The monthly proportions of fish with opaque material on the outer margin of otolith sections were calculated to define the period of annulus formation and its correlation with the water temperature. Temporal patterns in mean monthly marginal increment indices were used to examine the periodicity of annulus completion in different age classes. The periods when increment completion occurred in pooled data for fish with 2-9 opaque zones were examined by analysis of variance (ANOVA) with year and month as fixed factors.

Age and growth analyses

The growth of juvenile *G. auratus* (age 0+) was examined by length frequency plots of fish captured monthly in pelagic habitats in both lakes. Visual inspection of these plots found well

defined recruitment events in both populations; therefore, the monthly progression of mean cohort lengths was used to estimate the growth of 0+ fish. Progressive monthly growth rates were calculated as the difference between consecutive mean lengths of presumed cohorts divided by the number of days between sampling.

The relationships between length (FL) and weight (TW) of sexes in both populations were examined using the function:

$$\log_{10}(FL) = a + b \log_{10}(TW)$$

where a and b are constants. Birth dates of the 1st of September and 1st of October were assigned to the Lake Sorell and Lake Crescent populations, respectively based on peaks in the emergence of larvae in both lakes (S. A. Hardie, unpubl. data). Opaque zone counts were converted to decimal ages using assigned birth dates for each population, the periodicity of increment completion (31st of December was used as the date of opaque zone completion; see Results for rationale) and sample dates. The usually linear relationship between age and OW (Pawson, 1990) was analysed to determine if growth in the otoliths of *G. auratus* was proportional to somatic growth. Few fish <60 mm FL were able to be sexed and aged via otolith analysis; therefore, the age of unsexed pelagic 0+ fish belonging to the presumed 2000 and 2001 cohorts (for the first 7 months in Lake Sorell and first 8 months in Lake Crescent) was calculated using sample dates and the designated birth dates for each population. The growth of sexes in each population was investigated by: (i) fitting standard von Bertalanffy growth models to adult, and (ii) a combination of adult and juvenile length-at-age data, and also (iii) fitting a two-phase version of von Bertalanffy growth model (Hearn & Polacheck, 2003) to the combined adult and juvenile data. The von Bertalanffy equation is:

$$L_t = L_{\infty} \left(1 - e^{-K[t-t_0]} \right)$$

where L_t is the length at age t (years), L_∞ is the asymptotic length, K is the growth coefficient and t_0 is the hypothetical age at which members of a population would have length zero.

Standard von Bertalanffy growth models for each sex and population were compared using likelihood-ratio tests (Kimura, 1980). The age structure of the littoral dwelling proportion of each population (i.e. larger juveniles and adults) was estimated by applying the sex-specific length-at-age relationships (standard von Bertalanffy growth models) for each population to length frequency data of fish captured in littoral habitats. Age frequency distributions for each population were compared using Kolmogorov-Smirnov two-sample tests (Quinn & Keough, 2002). All analyses were carried out in R version 2.2.0 (R Development Core Team, 2005), and growth modelling included functions in the package nlme (linear and nonlinear mixed effects models) (Pinheiro *et al.*, 2005). The significance level for hypothesis tests was $P = 0.05$.

5.4. Results

Growth of juveniles

Length data of monthly catches of *G. auratus* from pelagic sites (Fig. 2) show annual recruitment events in lakes Crescent and Sorell during each year. During 2000 and 2001, *G. auratus* larvae (<10 mm FL) were most abundant in the water columns of both lakes from late winter to spring, whilst during 2002 larvae occurred as early as May in Lake Sorell and were present in relatively low numbers through winter and spring in both lakes. The monthly progression of length-related growth is evident in the strong 2000 (first 14 months) and 2001 (first 16 months) cohorts in Lake Crescent. Mean monthly growth rates of the presumed 2000 and 2001 cohorts in Lake Crescent were greatest during their first summer (0.24-0.48 mm day⁻¹) compared to the proceeding winter where growth basically ceased (0.00-0.07 mm

day⁻¹). Juvenile fish reached ~40 mm FL by the end of summer and ~60 mm FL by the end of their first year.

Fish that clearly belonged to 2000 and 2001 age 0+ cohorts also occurred in littoral fyke net catches during their first summers (generally from January onwards in both lakes). Some of these specimens have been included in the otolith analyses that were performed on larger fish to identify when the first opaque zone forms in the otolith sections of juveniles.

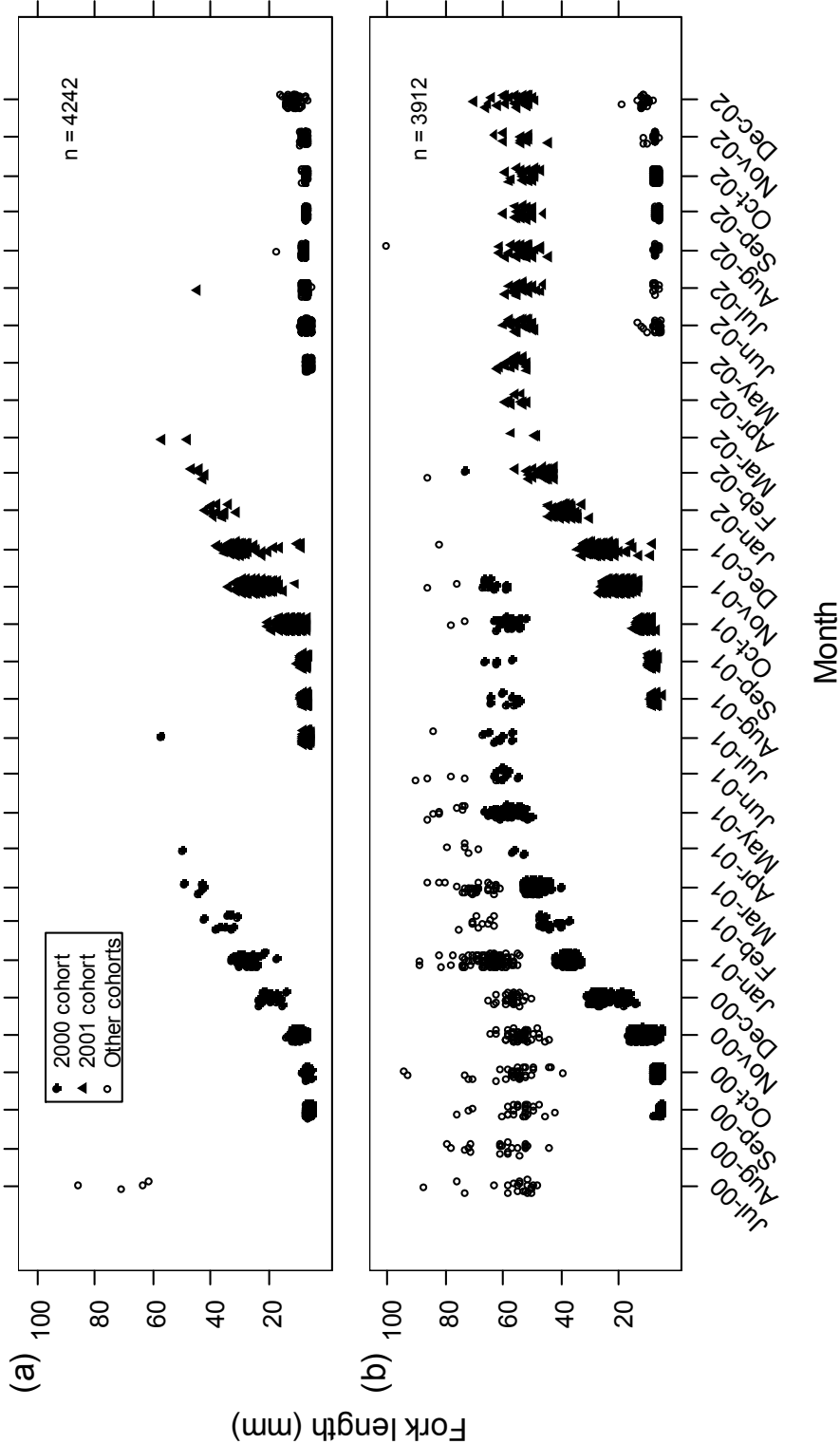


Fig. 2. Length of *Galaxias auratus* collected monthly from pelagic sites in (a) Lake Sorell and (b) Lake Crescent between July 2000 and December 2002. Presumed 2000 and 2001 cohorts of juvenile fish are shown by solid symbols.

Otolith structure

Viewed under transmitted light, otolith thin-sections from *G. auratus* of all sizes had a dense opaque primordium and distinct alternating translucent and opaque zones were clearly visible on otoliths of larger fish (>60 mm FL) (Fig. 3). Opaque zones were generally narrower than the translucent zones; however, the inner most opaque zone was often equal in width to the adjacent inner and outer translucent zones and much wider than the other opaque zones on a given otolith section (see Fig. 3). Examination of the position of the first opaque ring in a selection of otolith sections with 1 or 2 opaque rings found the mean (\pm S.E.) increment radius from the primordium to the first annulus to be $191 \pm 0.9 \mu\text{m}$.

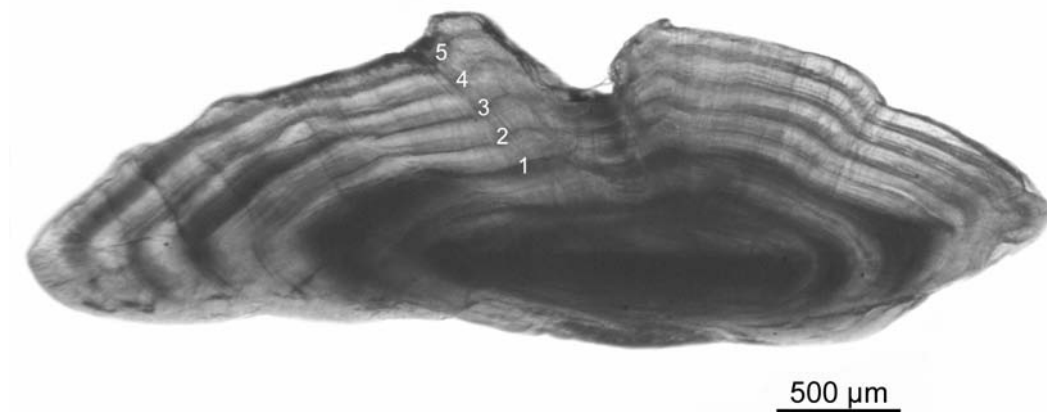


Fig. 3. Transverse thin-section (viewed under transmitted light) of a sagittal otolith from a female *Galaxias auratus* (148 mm FL and 33.6 g TW) from Lake Crescent. Opaque zones are numbered and the alignment shows the axis along which zones counts were read. The estimated age is 5 years.

Timing of opaque zone formation

Lakes Crescent and Sorell had very similar annual water temperature cycles between 1997 and 2002, with only a marginally significant difference between the lakes ($t = 2.025$, d.f. = 41, $P = 0.049$). However, the mean difference in the average monthly temperatures of the lakes

was very small ($<0.1^{\circ}\text{C}$) and not biologically meaningful. Therefore, water temperature data for both lakes and information on the status of otolith section outer margins for both populations were pooled to examine correlations between water temperature regimes and the formation of otolith incremental structure. Mean (\pm S.D.) monthly temperatures showed strong seasonal variation (Fig. 4) with temperatures being $<5^{\circ}\text{C}$ during winter (minimum of $4.0 \pm 1.0^{\circ}\text{C}$ during July) and $>14^{\circ}\text{C}$ during summer (maximum of $17.0^{\circ}\text{C} \pm 2.1^{\circ}\text{C}$ during January). The status of the outer margin was interpretable on 69% of *G. auratus* otolith sections. Opaque material was deposited on the outer margins of $>40\%$ of otolith sections with 0 to 9 opaque zones while mean monthly water temperatures increased from the July minimum to $12.7 \pm 2.3^{\circ}\text{C}$ in November (Fig. 4). The occurrence of opaque material on the outer margins peaked during September and October ($>75\%$) whilst translucent material was present on the outer margin of most ($>93\%$) otolith sections from fish sampled between January and May.

Annual trends in marginal increment indices of completion for otolith sections with ≥ 2 opaque zones (Fig. 5) were consistent with the periodicity of opaque zone formation. Whilst low samples sizes of fish with >4 opaque zones limited this analysis for older age classes, data for fish with 2, 3 and 4 opaque zones show a common pattern. Pooled data for fish with 2-9 opaque zones indicate that opaque zones were completed in early summer during 2000 and 2001 with sharp decreases in the mean indices of completion between November and December-January. In both years, the mean completion index for December was significantly less ($P < 0.01$) than the mean of pooled data for September to November.

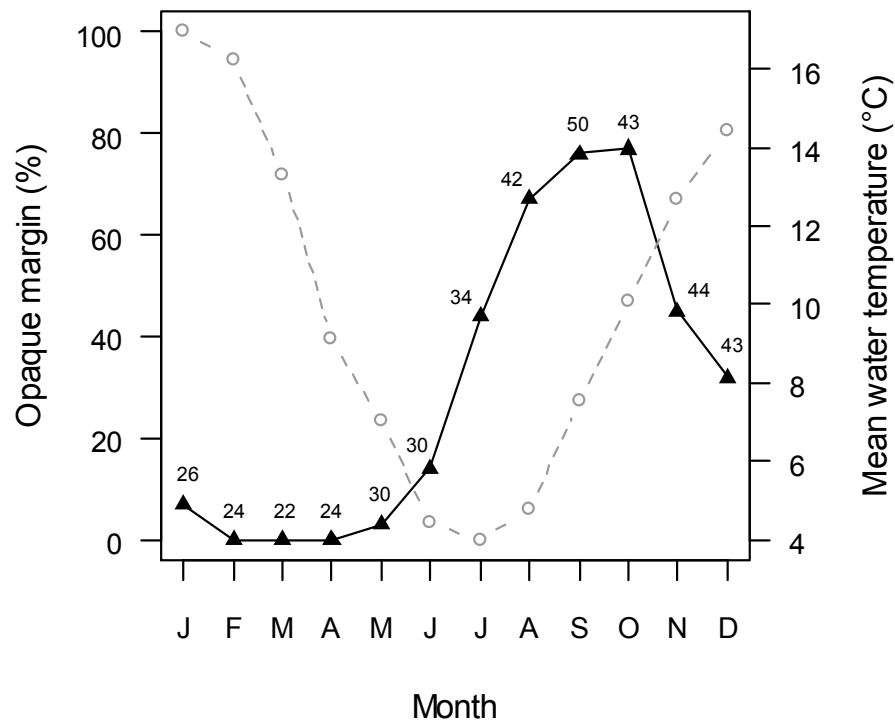


Fig. 4. Proportion of otolith sections with opaque material on the growing edge (solid triangles) and mean monthly water temperature of lakes Crescent and Sorell (open circles). Marginal status data of fish with 0-9 opaque zones from the entire sampling period from both lakes are pooled. Otolith section sample sizes are shown above data points.

The results from both forms of MIA indicate that single opaque and translucent zones are deposited annually in otoliths of *G. auratus* and, hence, opaque ring counts are a valid method for estimating age in this species. The first opaque zone is completed early in the second summer after emergence (i.e. at 14 months of age in Lake Crescent and 15 months in Lake Sorell).

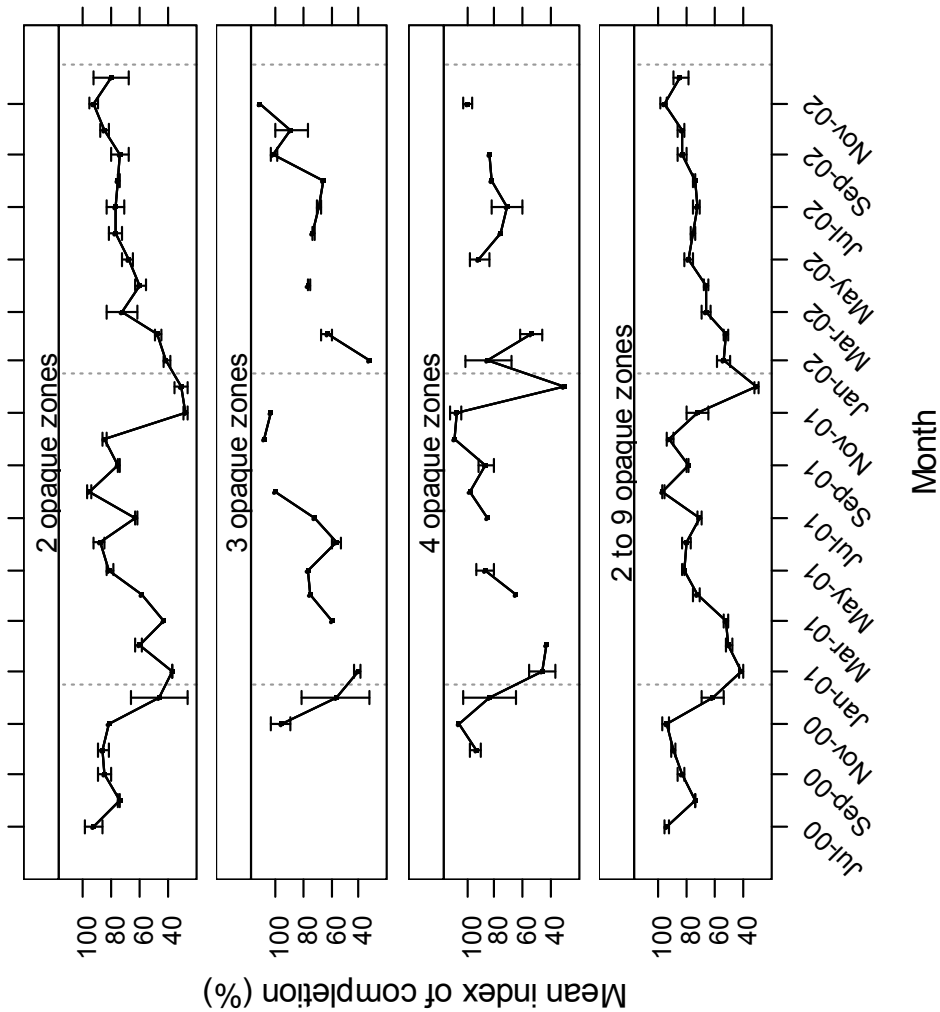


Fig. 5. Trends in mean (± 1 S.E.) monthly marginal increment indices of completion for otolith sections of *Galaxias auratus* (solid circles and lines) collected between July 2000 and December 2002. Data from lakes Crescent and Sorell are pooled. The 31st of December of each year is also shown (vertical dashed lines).

Precision of age estimates

Age estimates were recorded for 90% of the 600 otoliths examined. Otoliths sections that were unreadable had uncertain incremental structure which was caused by poor sectioning. This was particularly evident with otolith sections from smaller fish (<60 mm FL), where sectioning was more likely to not cut through the primordium of smaller-sized otoliths resulting in sections with non-complete incremental structure. The APE for age estimates by the primary reader was 3.4% with 90% of estimates in agreement, and 16.6% for both readers with 75% of estimates in agreement. The maximum difference in estimates within and between readers was 1 and 2 years, respectively. Age-bias plots illustrated a high level of agreement between age estimates (Appendix 2). Slope estimates (\pm S.E.) of the relationships between age readings, both within (slope = 0.95 ± 0.01) and between (slope = 1.00 ± 0.03) readers, closely approximated the 1:1 ratio that would occur if variation was equal for all year classes.

Growth of adults and population structure

There were significant differences between the FL and TW relationships of sexes in Lake Crescent ($P < 0.05$) but not in Lake Sorell ($P = 0.857$), and both sexes differed significantly between the populations ($P < 0.001$) (Table 1; Appendix 3). The oldest female *G. auratus* analysed was 10 years old and 34% of females were older than 3 years, whereas the oldest male was 7 years old and only 13% of males were older than 3 years. Age had a strong association with OW ($r > 0.88$; see Table 1; Appendix 4) and there were no significant inter-population differences between age and OW relationships with sex ($P = 0.245$), but there were significant differences between sexes ($P < 0.001$).

Table 1. Regression parameters and standard errors (S.E.) of fish length (FL)-total weight (TW) and estimated age-otolith weight (OW) relationships for male and female *Galaxias auratus*. Data from lakes Crescent and Sorell are pooled.

Sex	Relationship	<i>n</i>	Intercept \pm S.E.	Slope \pm S.E.	R^2
Male	$\log_{10}(\text{FL}) \sim \log_{10}(\text{TW})$	925	-13.45 ± 0.13	3.41 ± 0.03	0.93
	Age \sim OW	190	-0.46 ± 0.12	1.47 ± 0.05	0.80
Female	$\log_{10}(\text{FL}) \sim \log_{10}(\text{TW})$	1771	-13.79 ± 0.08	3.47 ± 0.02	0.95
	Age \sim OW	308	-0.32 ± 0.13	1.61 ± 0.04	0.84

Combined juvenile and adult length-at-age data for both populations (Fig. 6) showed distinct phases in the growth trajectories of juveniles and adults with rapid growth in 0+ fish up to ~60 mm FL. Growth of adults (>60 mm FL) was slower and more gradual and did not approach asymptotic lengths, with the exception of females in Lake Sorell. Because of this growth pattern, both the standard and two-phase von Bertalanffy growth functions fitted poorly to combined juvenile and adult length-at-age data. The two-phase function did not converge on estimates, whilst the standard function fitted the growth of juveniles, but not the growth of adults. The standard von Bertalanffy function did, however, fit length-at-age data of adults on their own (Fig. 6), but with the exception of females in Lake Sorell, L_{∞} estimates were large (compared to maximum lengths: male = 179 mm FL, and female = 235 mm FL) and had significant errors (>25%) (Table 2). These growth models were sufficient to enable comparisons between sexes, which differed significantly both between ($P < 0.001$) and within ($P < 0.001$) the populations. The growth models indicated that females grow faster than males, and both sexes in Lake Sorell grow faster than those in Lake Crescent where the growth of adults followed a near-linear trajectory.

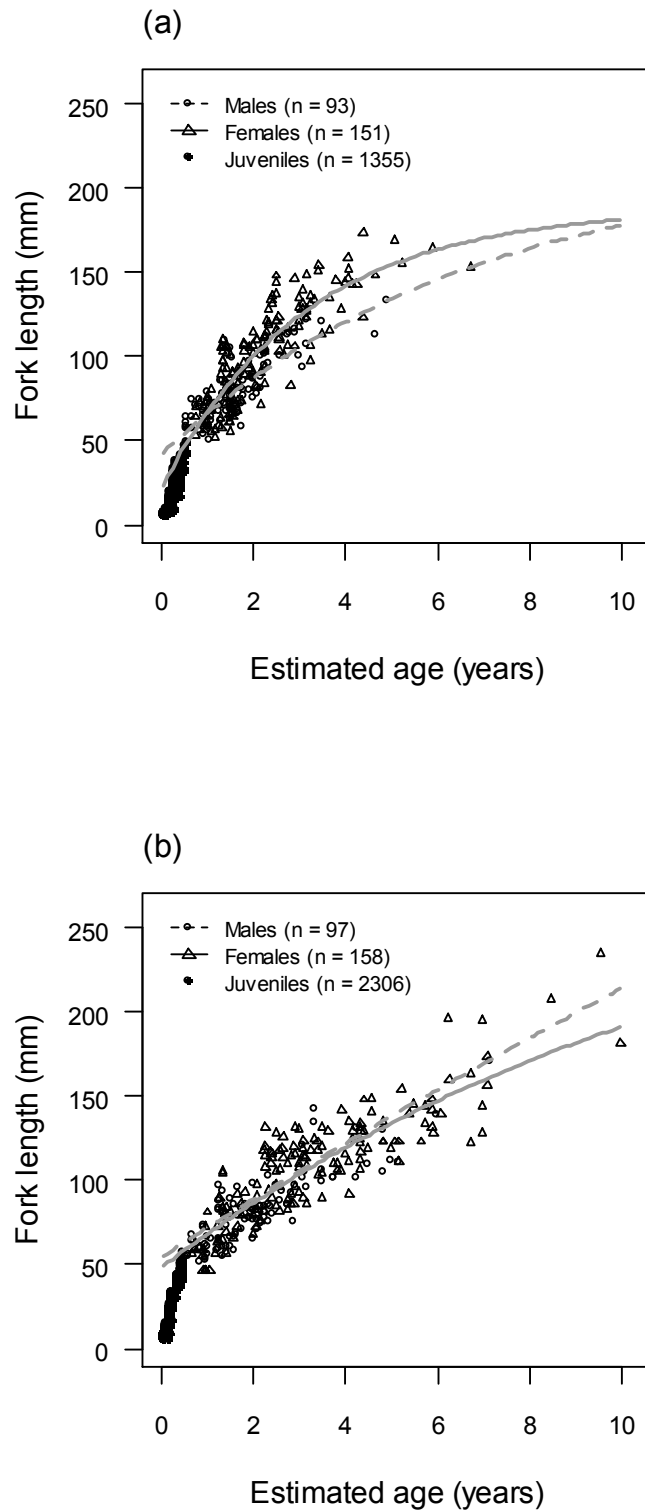


Fig. 6. Estimated age and length of *Galaxias auratus* from (a) Lake Sorell and (b) Lake Crescent with fitted von Bertalanffy growth models for females and males (fitted to adult data only). Growth model parameter estimates are given in Table 2.

Table 2. Von Bertalanffy growth model parameters and standard errors (S.E.) for male and female *Galaxias auratus* from lakes Crescent and Sorell. Growth models were fitted to adult length-at-age data only.

Lake	Sex	<i>n</i>	$L_{\infty} \pm \text{S.E. (mm)}$	$K \pm \text{S.E. (years}^{-1}\text{)}$	$t_0 \pm \text{S.E. (years)}$
Crescent	Male	97	934.98 ± 3673.25	0.02 ± 0.08	-3.00 ± 1.35
	Female	158	308.14 ± 77.82	0.08 ± 0.03	-2.14 ± 0.50
Sorell	Male	93	216.69 ± 125.54	0.15 ± 0.15	-1.44 ± 0.77
	Female	151	187.96 ± 17.18	0.32 ± 0.07	-0.39 ± 0.21

Whilst *G. auratus* were found to attain ages of up to ~10 years, fish in the 0+, 1+ and 2+ year classes dominated the estimated age structure of populations in Lake Crescent (76%) and Lake Sorell (94%) based on fish captured by electrofishing and fyke netting in littoral habitats (Fig. 7). Age distributions of the populations differed significantly ($P < 0.001$) with the Lake Crescent population having fewer 1+ fish and greater proportions of older fish (i.e. 3+, 4+ and 5+).

5.5. Discussion

Evaluation of aging technique

Transverse thin-sections of *G. auratus* otoliths provided an interpretable view of incremental structure which enabled robust age estimates for a large size-range of fish to be derived with reasonable precision. To our knowledge, this is the first time otolith thin-sections have been used to study the age and growth of a galaxiid fish as previous studies have examined whole otoliths (Humphries, 1989; Bonnett, 1990; Pen & Potter, 1991b; McDowall & Kelly, 1999; Morgan, 2003). During initial trials using whole, ground and polished *G. auratus* otoliths, their thickness obscured incremental structure and the status of the growing margin was unclear. The use of otolith sections not only provided a clear view of incremental structure,

but also enabled the position of the first annulus to be quantified. This protocol greatly improved the precision of opaque zone counts as age under-estimation was likely without this measurement owing to the first zone being overlooked in otolith sections from older fish. Other researchers have also found similar protocols to be useful in fish aging (Ewing *et al.*, 2003; Stevens *et al.*, 2005; Tracey & Lyle, 2005). The status of the growing margin could also be accurately assessed on *G. auratus* otolith sections, hence, the timing and duration of the formation of opaque and translucent material was able to be determined.

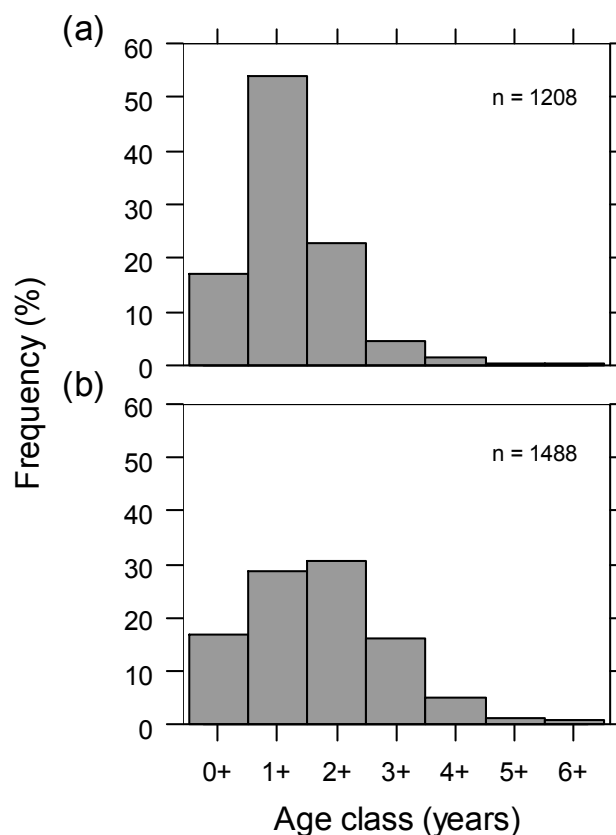


Fig. 7. Age frequency distributions of *Galaxias auratus* in (a) Lake Sorell and (b) Lake Crescent based on fish captured in littoral habitats between July 2000 and December 2002. Age classes were assigned using estimated length-at-age relationships for males and females in each lake. The 6+ year class includes fish aged ≥ 6 years.

Otoliths from *G. auratus* <60 mm FL were difficult to section successfully due to their small size (OWs ≤ 1.3 mg) with only 20% of otolith section readings from fish <60 mm FL being recorded as 100% reliable. Sectioning often missed cutting through the primordium of these smaller otoliths; therefore, incremental structure was not complete. Most *G. auratus* <60 mm FL that were aged successfully (89%) were age 0+, so this limitation did not hinder our growth analyses. However, this limitation may reduce the sectioning technique's applicability to age and growth studies on small-sized galaxiid species (such as *Paragalaxias* spp.) which may have annual incremental structure in small-sized otoliths. We recommend further studies on other larger-sized galaxiids (i.e. commonly >80 mm FL) examine the incremental structure of otolith sections to derive age estimates. In the case of smaller-sized species (i.e. commonly <80 mm FL), the use of otolith weight or dimensions may provide adequate relationships with somatic growth to conduct age and growth analyses. Techniques commonly used in daily aging studies of age 0+ fish, such as grinding (Humphries, 2005) and scanning electron microscopy (Smith & Walker, 2003), may also be effective for examining annual increments in the otoliths of small-sized galaxiids.

Age and growth

While several exogenous and endogenous factors effect fish growth (Wootton, 1998), water temperature is important in both somatic (Goolish & Adelman, 1984; Forseth & Jonsson, 1994) and otolith (Lombarte & Lleonart, 1993; Barber & Jenkins, 2001) growth. In our study, trends in both somatic growth of 0+ fish and increment formation in otolith sections of older age classes were correlated with water temperature cycles. The peak in the occurrence of opaque material deposition on the growing margin of otolith sections (September and October) occurred 2-3 months after the coldest mean monthly water temperature in the lakes (July: $\sim 4.0^{\circ}\text{C}$) and also the winter spawning period of *G. auratus* (Hardie *et al.*, in press).

Therefore, the periodicity of opaque material formation in *G. auratus* is likely to be linked to periods of slow somatic and otolith growth associated with low water temperatures. Phases of reproductive investment may also influence the growth of otoliths in mature fish. Other studies have found similar relationships between opaque material formation in galaxiid otoliths and spawning season (Morgan, 2003), and both water temperature and spawning season (Pen & Potter, 1991a, b; Pen *et al.*, 1991).

Length-at-age data for juvenile and adult fish showed clear growth patterns in both populations. The two distinct growth phases may be associated with this species' pelagic-to-epibenthic ontogenetic habitat shift (between 40 to 70 mm FL; Chapter 8) and sexual maturation, with most males maturing in the first year and females in their second (Hardie *et al.*, in press). We were unable to fit standard or two-phase von Bertalanffy growth models to the combined juvenile and adult data successfully; however, standard von Bertalanffy growth models did fit (somewhat poorly) adult data. The two-phase model was attempted as it was thought to be able to account for the transition in growth rate, given it has been successfully fitted to length-at-age data of longer-lived marine species which undergo similar growth patterns (Hearn & Polacheck, 2003; Tracey & Lyle, 2005). However, combinations of the two distinct phases of growth, lack of data around the transition period, and near-linear growth of adults (which are relatively short-lived) were problematic in the fitting of both models. We suggest that for similar reasons in some previous galaxiid aging studies, von Bertalanffy growth models have also not approximated true growth patterns well (Pen & Potter, 1991b; Pen *et al.*, 1993; Morgan, 2003).

Otolith weight correlates well with age in some species (Boehlert, 1985; Pawson, 1990), and its use may help refine the cost-effectiveness of procedures to estimate length-at-age

distributions (Francis & Campana, 2004; Francis *et al.*, 2005). It was a good indicator of age for *G. auratus*, but unlike functions such as the von Bertalanffy growth model, the OW-age linear relationship does not include information about changes in growth rates during the developmental stages of a species' life history. However, OW may be useful where quantifying intra-annual changes in growth is unimportant. Clearly, there is a need to further investigate growth modelling of species such as galaxiids which are likely to be relatively short-lived, have distinct juvenile and adult growth phases, and don't approach an asymptotic length as adults.

The growth patterns and longevity of galaxiids vary from annual, small-sized (maximum of ~40 mm FL) species such as *Galaxiella pusilla* (Humphries, 1986) to the largest species, *G. argenteus* (commonly >300 mm FL), which has been speculated to live for >20 years (McDowall, 2000). Similarly to other galaxiid species (Pollard, 1971; Humphries, 1986, 1989; Bonnett, 1990; Pen & Potter, 1991b; Rowe *et al.*, 2002a), female *G. auratus* grow larger and live longer than males, while both sexes have greater longevity than the 30 months previously speculated for this species (Fulton, 1990). The growth patterns of *G. auratus* are comparable to riverine *G. truttaceus* in south-western Australia (Morgan, 2003) which also appeared to have two disjointed growth phases with transitions at the end of the first year (45-60 mm in total length). Whilst von Bertalanffy curves fitted the adult data, some parameters were poorly estimated and biologically improbable. For example, L_{∞} for males in Lake Crescent was over five times larger than the maximum recorded length for males during this study and had a very large standard error. However, the predicted lengths of juveniles at the end of their first year (range: 66.5 to 71.9 mm FL, for males in Lake Sorell and Lake Crescent respectively) and predicted age of the largest fish collected

(235 mm FL = 15.86 years) were realistic. This provides some confidence in the growth curves for describing the growth of adults.

Von Bertalanffy growth curves for adults differed between the lakes with both sexes in Lake Sorell growing faster than those in Lake Crescent. There was no substantial difference between the water temperature regimes of the lakes; therefore, the contrasting growth patterns of the populations are due to other factors. The quantity of suitable food resources for various development stages is critical to fish growth (Wootton, 1998), and limited availability refuge habitats is known to change social behaviour and also decrease somatic growth (Fischer, 2000; Fischer & Öhl, 2005). During the four years prior to and during the study, both lakes had very low water levels. This caused a reduction in the availability of littoral and in-lake complex habitats (e.g. rocky substrates) in Lake Crescent, which provide refuge and food resources for adult *G. auratus*, whereas large areas of these habitats were still present in Lake Sorell (Hardie, 2003; Hardie *et al.*, in press; Chapter 8). The standing crop of phytoplankton in Lake Crescent is 10 times greater than that in Lake Sorell (Cheng and Tyler 1976b; A. Uytendaal, unpubl. data) and during this study, the *G. auratus* population in Lake Crescent was 7.7 times greater in relative abundance than the Lake Sorell population (Hardie *et al.*, 2005). Therefore, the greater food resources (i.e. zooplankton) for pelagic juveniles in Lake Crescent may result in increased survival through to maturity, but the growth of adults may be reduced by density-dependent constraints associated with limited areas that provide refuge and suitable food resources (i.e. macroinvertebrates).

Population structure

Our results are consistent with those of other studies which indicate that lacustrine populations of *Galaxias* spp. generally have greater proportions of larger older fish than

riverine populations (McDowall, 1968; Pollard, 1971; Humphries, 1989; Bonnett, 1990; Pen & Potter, 1991b; Morgan, 2003), with the exception of some larger-sized and longer-lived riverine species such as *G. argenteus* (David *et al.*, 2002) and *G. fasciatus* (West *et al.*, 2005). The age structure of the *G. auratus* population in Lake Crescent is similar to that of lacustrine populations of *G. truttaceus* of the TCP (Humphries, 1989) with many fish >2 years of age; however, the structure of the Lake Sorell population is more comparable to riverine *G. truttaceus* populations in Tasmania with few fish >2 years old.

The fish communities in lakes Crescent and Sorell typify many lakes of the TCP, with populations of galaxiids (including some endemic species) co-existing with introduced salmonids. Salmonids negatively impact on the distribution and abundance of galaxiids in several regions (Crown *et al.*, 1992). *Salmo trutta* and *O. mykiss* are known to prey heavily on *G. auratus* in both Lake Crescent and Lake Sorell (Stuart-Smith *et al.*, 2004); however, the more abundant salmonids in Lake Sorell are likely to influence the size and age structure of the *G. auratus* population through increased predation on small to medium-sized fish and direct competition with larger fish.

Conservation and management

Conservation and recreational fisheries management programs need to take into account the longevity of this threatened species and the age structure of its populations when interpreting *G. auratus* catch data and designing fisheries management strategies for lakes Crescent and Sorell. While *G. auratus* is longer-lived than many other galaxiid species, the short life-span of most fish, especially males, mean significant declines in abundance could occur in <3 years in response to events such as recruitment failures or sudden increases in the abundance of salmonids. Differences in the age structure of the *G. auratus* populations in lakes Crescent

and Sorell, which are exposed to different levels of competition and predation from salmonids, further supports the body of literature that illustrates the negative impacts of salmonids on galaxiids. This study highlights the need to quantify the growth patterns and age structure of populations of other threatened Tasmanian endemic galaxiids to enable the development of effective conservation-based fisheries management strategies.

6. Reproductive biology of the threatened golden galaxias, *Galaxias auratus* Johnston 1883 (Galaxiidae) and the influence of lake hydrology*

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6.1. Abstract

Golden galaxias *Galaxias auratus* (31-235 mm fork length L_F) were collected monthly from littoral habitats in lakes Crescent and Sorell, Tasmania, Australia, between July 2000 and December 2002. Spawning habitats were identified and monitored in both lakes, and these areas were surveyed in Lake Crescent. Trends in gonado-somatic indices and reproductive stages of development indicated that gonad development in both sexes begins in mid-summer and peaks in late autumn – early winter. Males mature at smaller sizes (fifty per cent at 52 mm L_F) than females (fifty per cent at 76 mm L_F), larger individuals are predominately females (ninety five per cent of fish ≥ 138 mm L_F), and overall male to female ratios are female-biased (c. 1:2). Spawning occurs late autumn – early spring (water temperatures = 1.4-9.7°C) with peaks in spawning activity in winter (mean water temperatures $< 5^\circ\text{C}$). Demersal adhesive eggs (c. 1.5 mm diameter) were found on cobble substrates (c. 20-250 mm diameter) in littoral areas (c. 0.2-0.6 m deep) and fecundity of fish 71-181 mm L_F ranged from 619 to 14478 eggs. The rate of change in water level over the 20 days prior to monthly sampling was important in explaining the occurrence of spent fish and this accounted for temporal differences in spawning between the populations. Lake hydrology influences the reproductive cycle of *G. auratus* by possibly providing a stimulus for spawning, and it controls the

availability of spawning habitat in Lake Crescent. Seasonal hydrological cycles (i.e. rises during late autumn – winter) and a minimum water level of 802.20 m AHD in Lake Crescent during autumn (above which littoral areas of cobble substrate are inundated) are critical to *G. auratus* populations.

Key words: Tasmanian Central Plateau; highland lakes; non-diadromous; spawning; water levels; threatened species

6.2. Introduction

Identifying vulnerable aspects of the life histories of threatened freshwater fishes is important to their conservation (Angermeier, 1995), and knowledge of the reproductive biology of these species is a critical component of their management. Coupled with biological research is the need to determine the ecological requirements of fishes, including how they use specific habitats. Effective management strategies recognise that species not only depend on the existence of suitable habitat, but on its availability in the right time and place (Naiman & Latterell, 2005).

Water level fluctuations can influence the abundance, distribution and life cycles of lacustrine fishes (Gasith & Gafny, 1990; Winfield, 2004a). The reproductive success of some littoral-spawning species is positively correlated with water level increases prior to, and during spawning (Maceina & Stimpert, 1998; Sammons *et al.*, 1999; Sammons *et al.*, 2001; Ozen & Noble, 2005). Whilst the governing mechanisms are often not understood, in some systems greater reproductive success has been linked to the inundation of suitable spawning habitats (Beam, 1983; Miranda *et al.*, 1984; Gafny *et al.*, 1992; Rowe *et al.*, 2002b). These results

suggest variation in the availability of spawning habitats in lentic waters can strongly influence fish production.

Fishes in the Galaxiidae are found on several land masses in the Southern Hemisphere, where they generally occur in cool temperate regions (McDowall & Fulton, 1996). Galaxiids dominate the native freshwater fish fauna of the Tasmanian Central Plateau (TCP), southern Australia, because of the occurrence of seven endemic lacustrine species (Hardie *et al.*, 2006b). Whilst the family exhibits two life history strategies (diadromous and non-diadromous), all endemic species of the TCP are non-diadromous (McDowall & Fulton, 1996). However, knowledge of their life cycles is limited and all seven species are currently considered threatened due to their highly restricted distributions (five species have ≤ 2 natural populations), impacts from introduced fishes, and alterations to water level regimes (Hardie *et al.*, 2006b).

The golden galaxias, *Galaxias auratus* Johnston, is the largest of the endemic TCP species, growing to ~280 mm in length (Johnston, 1883). Only four populations of *G. auratus* exist: two natural populations in the interconnected lakes Crescent and Sorell and two translocated populations in small off-stream agricultural water storages in the Clyde River catchment (Hardie, 2003; Hardie *et al.*, 2004). The reproductive biology of *G. auratus* has not previously been studied in detail. It has been suggested that spawning occurs during spring on rocky shores (Fulton, 1990; McDowall & Fulton, 1996; Allen *et al.*, 2002). However, the occurrence of fertilised eggs on fyke nets between May and October (Hardie *et al.*, 2005), suggests a longer and more winter-based spawning period.

If this species is a winter-spawner, it would be unusual amongst lacustrine galaxiids. To date, lacustrine galaxiid populations in Patagonian Argentina (Barriga *et al.*, 2002), New Zealand (Rowe & Chisnall, 1996a; Rowe *et al.*, 2002a) and south-eastern Australia (Pollard, 1971; Fulton, 1982; Humphries, 1989) have been found to spawn in spring-summer. Spawning habitats include submerged vegetation in littoral areas (Humphries, 1989) or inflowing tributaries (Pollard, 1971), and rocky shores (Fulton, 1982). Hydrologic conditions such as high flows in inflowing streams (Pollard, 1971), and also increases in water temperature (Humphries, 1989) are thought to be important spawning cues for some lacustrine species. The only other comprehensive accounts of galaxiid reproduction in Australia are mostly for riverine species (Humphries, 1986; O'Connor & Koehn, 1991; Pen & Potter, 1991a; Pen *et al.*, 1991; Pen *et al.*, 1993; Shirley & Raadik, 1997; O'Connor & Koehn, 1998; Morgan, 2003) and data for Australian lacustrine populations are more limited (Pollard, 1971, 1972a; Fulton, 1982; Humphries, 1989; Chapman *et al.*, 2006).

The unique TCP galaxiid fauna is currently exposed to impacts from altered hydrology caused by hydroelectric power schemes, and agricultural and municipal water allocations. These impacts are likely to be further exacerbated by the effects of climatic change (e.g. more frequent and intense *El Niño*-induced droughts (Kershaw *et al.*, 2003)) in the future.

Therefore, links between the reproductive cycles of threatened lacustrine galaxiids, such as *G. auratus*, and lake hydrology need to be documented so that water management strategies for the region can incorporate their requirements. Accordingly, this study examined the reproductive biology of *G. auratus* in lakes Crescent and Sorell on the TCP and investigated the influence of lake hydrology on the reproductive cycle of this species. The aims were to: (1) examine the sex ratio and size of maturation for males and females in both populations, (2) describe gonadal cycles by examining the gonado-somatic index (I_G) and development of

gonads over a 30 month period and quantify fecundity, (3) locate and survey spawning habitats and monitor spawning activity, and (4) define spawning periods and investigate the timing of spawning events in relation to temperature and water level fluctuations.

6.3. Materials and methods

Study site

The interconnected Lake Crescent (42°10'S; 147°10'E) and Lake Sorell (42°6'S; 147°10'E) lie 1 km apart and are situated at *c.* 800 m a.s.l. at the head of the Clyde River catchment in the south-east of the TCP (Fig. 1). Both lakes are quite large in comparison to other Tasmanian lakes (surface areas of 23 and 52 km², respectively) and relatively shallow (mean depths of 2.3 and 3.1 m at full supply, respectively) and originate from natural lakes that have had their levels raised on several occasions since the 1830s for water storage purposes. The Crescent-Sorell system has six ephemeral tributaries, the largest being Mountain Creek (mean annual discharge = 11767 ML; Uytendaal (2003)) which flows in at the northern end of Lake Sorell, and a single outflow, the Clyde River, which flows out of Lake Crescent to the south-west. The hydrology of the system is largely controlled by relatively low rainfall (annual mean = 699 mm), small catchment area (206 km²) and high evaporation rates during summer (mean = 4.5 mm day⁻¹). Water releases for downstream users (annual mean = 10 000 ML), which are managed by sluice gates in the Interlaken Canal and at the Clyde River outflow, also alter water level fluctuations (DPIWE, 2004). Historically, fluctuations have been seasonal (minimums occur March-April and maximums in October-November) and typically range from *c.* 0.2 to 0.9 m in both lakes (Inland Fisheries Service, Tasmania, unpubl. data).

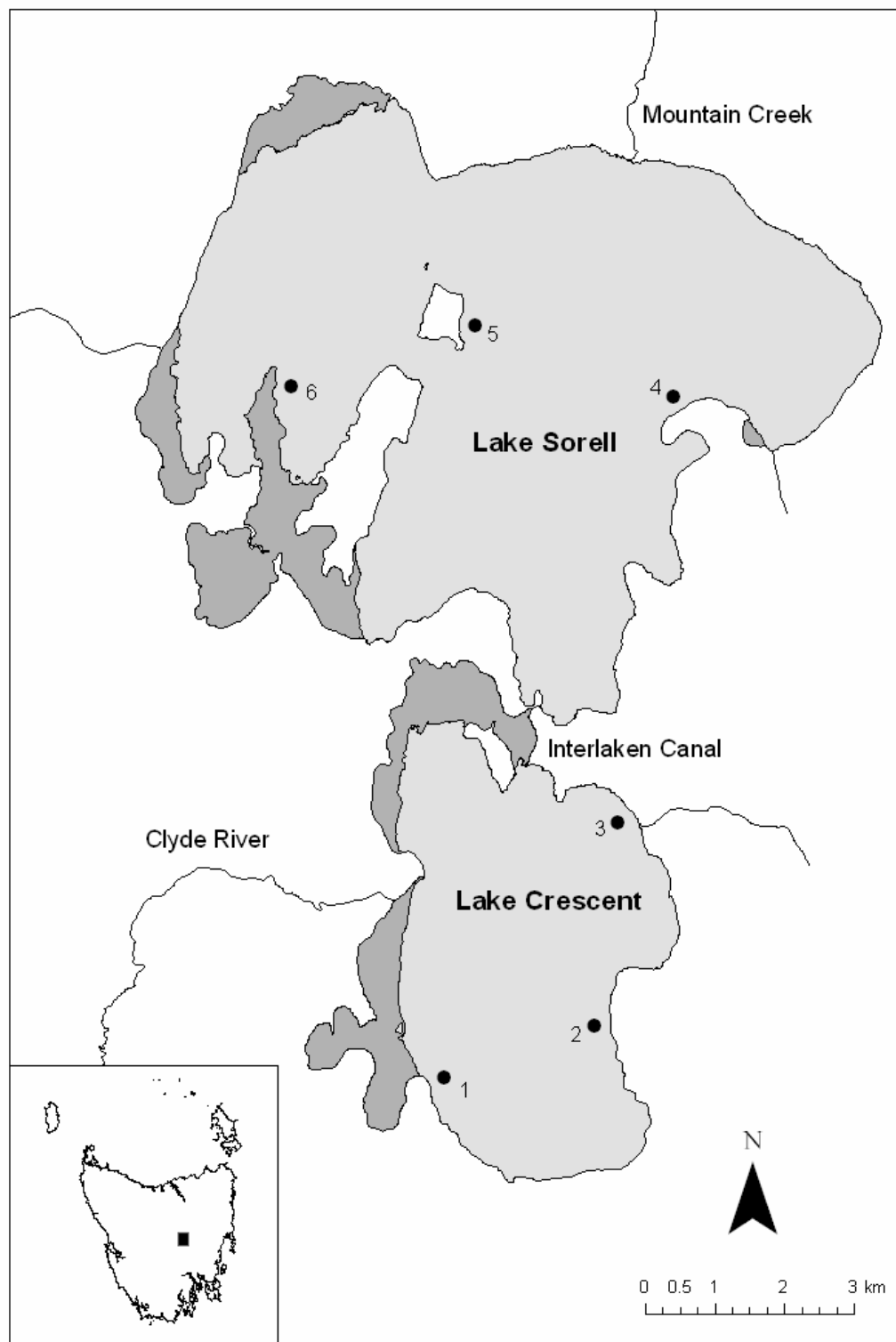


Fig. 1. Location of sampling sites (●) for *Galaxias auratus* in lakes Crescent and Sorell, Tasmania, Australia. Lake perimeters are at full supply levels. Wetland areas are depicted by dark shading.

Extensive littoral wetlands (Fig. 1), accounting for 17% of the surface area of Lake Crescent and 8% of Lake Sorell at full supply levels (Heffer, 2003), connect to the main bodies of the lakes at high water levels (Fig. 2). However, during this study these wetlands were not inundated (Fig. 3) and littoral habitats consisted of defined regions of three substrate types (fine sediment, rock and sand) and in-lake habitats by either fine sediment or rock. Historically, both lakes have also contained in-lake macrophyte beds, but none were present during this study because of low water levels and consequent increased exposure of bed sediments to wave action and elevated turbidity (Uytendaal, 2003, 2006).

The fish assemblage in both lakes consists of two native species, the endemic *G. auratus* and indigenous short-finned eel *Anguilla australis* Richardson, and three exotic species, brown trout *Salmo trutta* L., rainbow trout *Oncorhynchus mykiss* (Walbaum) and common carp *Cyprinus carpio* L. *Galaxias auratus* are ~8 times more abundant in Lake Crescent than in Lake Sorell, possibly due to differences in salmonid predation rates and lake productivity (Hardie *et al.*, 2005).

Environmental variables

Mean daily in-lake water temperatures between January 2000 and December 2002 were measured (depth 0.5 m) using Optic StowAway Temperature loggers (Model WTA). Mean daily water temperature in both lakes during this period was also modelled with the Dynamic Reservoir Simulation Model (DYRESM) (Antenucci, 2001) using meteorological data supplied by Hydro Tasmania, Tasmanian Bureau of Meteorology and the Tasmanian Inland Fisheries Service. Modelled temperatures closely approximated observed values ($R^2 \geq 0.95$) and were used to fill in occasional gaps in observed data caused by equipment failure. Daily

water level data for both lakes during the study was supplied by the Tasmanian Inland Fisheries Service and photoperiod data by Geoscience Australia (Geoscience Australia, 2006).

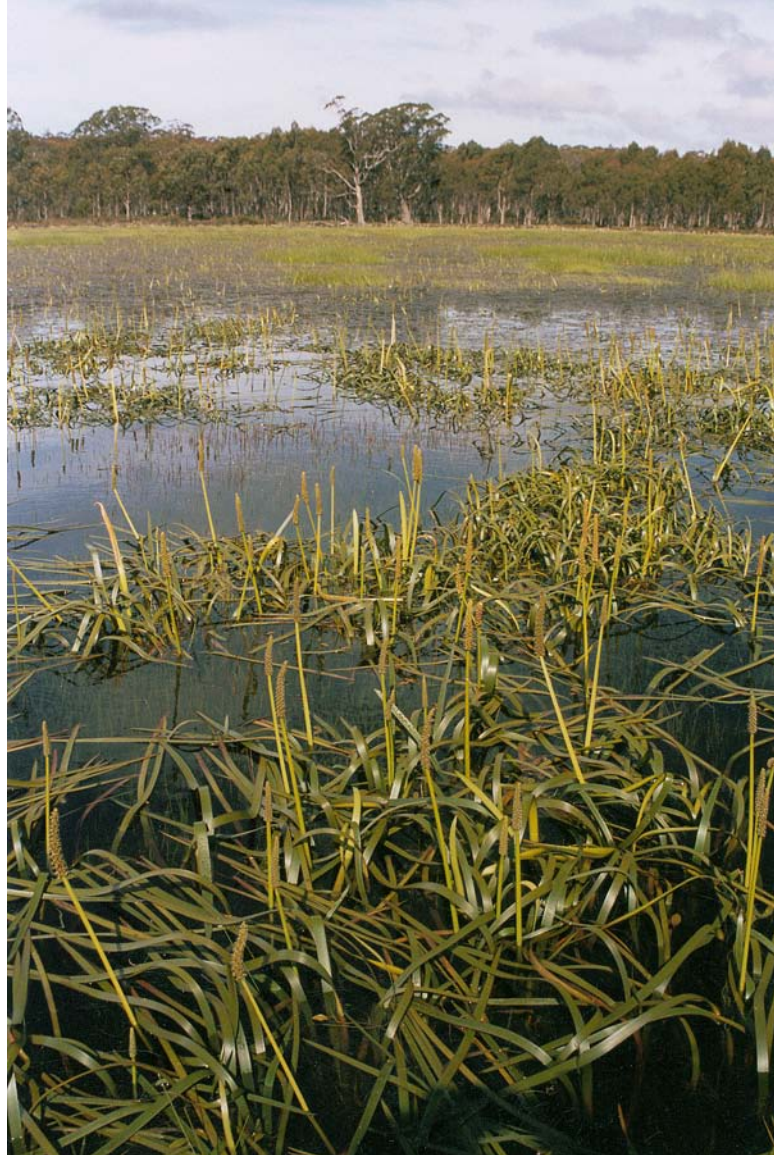


Fig. 2. Typical vegetated habitat of a shallow temporary wetland in the Interlaken region (Hazelwoods Lagoon, 2001).



Fig. 3. Dewatered wetland habitat on the southern shore of Lake Crescent (Bullies Marsh) during 2001.

Fish sampling and biological analysis

Galaxias auratus were sampled monthly from littoral habitats in lakes Crescent and Sorell between July 2000 and December 2002. On each sampling occasion, fish were collected at three sites in each lake (Fig. 1) which each represented one of the dominant littoral habitats (fine sediment, sand and rock substrates) in both lakes. On the first three sampling occasions, fish were sampled by electrofishing 100 m of shoreline at each site with a backpack electrofisher (Smith and Root Inc, 12-B POW). On the remaining sampling occasions, fish were sampled using four tandem-set fyke nets (2 mm stretched mesh) at each site (i.e. 12 nets per lake). All nets had an 84 × 70 mm aluminium screen in the entrance to avoid the capture of platypus (*Ornithorhynchus anatinus*), water birds and larger fish species (e.g. salmonids). Nets were set overnight (mean soak time = 18 hours) in shallow margins (depths ≤1.2 m). A subsample of *G. auratus* from each lake (Lake Crescent: $n = 43-77$; Lake Sorell: $n = 10-70$) was

euthanased in a lethal anaesthetic solution (Aqui-s™) and preserved in 70% ethanol for further analysis. The remaining captured fish were allowed to recover from anaesthesia and released at the site of capture.

Each *G. auratus* (Lake Crescent: $n = 1624$, Lake Sorell: $n = 1354$) was measured, fork length (L_F , mm), and weighed (M , g). Gonads were removed and weighed (M_g , mg) and the gonadosomatic index (I_G , %) was calculated ($I_G = 100 M_g M^{-1}$). Sex was determined by macroscopic and microscopic examination of gonads and gonadal stages of development of ovaries and testes (Table 1) were recorded. The fecundity (F) of stage 4 female fish in the June and July 2001 samples from both lakes ($n = 87$) was examined. The number of eggs in a 200 mg sub-sample of ovary from each fish was counted microscopically and the total number present in ovaries was extrapolated. The mean (\pm S.E.) sub-sample proportion of M_g was 10.9 ± 0.8 %. For eight fish, F analysis was replicated ($n = 5$) to determine the consistency of the ovaries and, hence, the precision of the technique (S.E. of each replication was <16 eggs).

Table 1. Volume and appearance of *Galaxias auratus* ovaries and testes at five stages of development*

Stage	Volume [†] (%)	Macroscopic/microscopic [‡] appearance
1. Immature/ Resting mature	<25	Lobes of testes and ovaries thin and transparent and not meeting each other at the ventral end of visceral cavity. Oocytes only visible upon rapture of tunica albuginea. Oocytes transparent and small to indistinct.
2. Immature developing/ mature re-developing (early)	26-50	Lobes of testes and ovaries close to meeting each other at the ventral end of visceral cavity. Testes white in colour with vermicular appearance. Oocytes distinct, and variable in size, some white, some yellow.
3. Immature developing/ mature re-developing (late)	51-75	Lobes of testes and ovaries meeting each other at the ventral end of visceral cavity. Testes white in colour with vermicular appearance. All oocytes yellow-orange in colour but small in size.
4. Ripe	>76	Body wall slightly – very distended with testes and ovaries filling the visceral cavity. Testes white in colour. Oocytes uniform in size and yellow-orange in colour. Sexes maybe determined externally with smooth v. nodular patterns visible on the body wall of males and females respectively.
5. Spent	Variable	Lobes of testes have slack appearance with flaccid tunica albuginea. Testes may have grey colour. An obvious decrease in the amount of ripe oocytes present in visceral cavity. Remaining oocytes may be disfigured and more pale in colour than viable oocytes. This stage also includes fish with very small newly formed resting stage gonads with residual oocytes and testes still present.

*Stages of development are modified from Fulton (1982).

[†]Estimated volume of visceral cavity occupied by gonads.

[‡]Viewed at x25 magnification.

Spawning site survey and rocky substrate mapping

Regular searches for spawning sites were performed in lakes Crescent and Sorell during the 2000 and 2001 spawning periods (winter – early spring). Littoral areas were searched by visual examination of submerged habitat such as rocks and woody debris, and kick-net sweeps using hand nets (250 μm mesh) were performed over different substrate types (Fig. 4). Where eggs were found, water depth was measured and the composition of the habitat was recorded. Subsequently, spawning activity was monitored *c.* fortnightly at known spawning sites in each lake (sites 2 and 4; Fig. 1) during the 2001 and 2002 spawning periods (late April – late September). The relative abundance of eggs at each site (eggs sweep⁻¹) was assessed by 30 kick-net sweeps (hand net, 250 μm mesh) of 30 s duration. These results estimated the density of eggs incubating at the site and, hence, indicated recent spawning activity.

Littoral areas of rocky substrate in Lake Crescent were surveyed during March 2002, when the water level was near an historical low (*c.* 802.445 m Australian Height Datum (AHD)), and significant proportions of these areas were dewatered. The extent of the regions of rocky substrate was recorded along transects (intervals of 100 m) at 90° to the shore using a global positioning system (GPS). Water depth (± 10 mm) at the rock-sediment interface was recorded and daily water levels were used as reference levels for submerged sites. These survey data were used to construct a map of the rocky shore habitat in Lake Crescent (using MapInfo®) and determine the influence of lake levels on its availability.



Fig. 4. Kick-net sweep to monitor spawning activity of *Galaxias auratus* at site 4 in Lake Sorell, July 2002 (see Fig. 1 for location) (sampler: Brett Mawbey).

Statistical analyses

Length frequency distributions of sexes in each population were compared using Kolmogorov-Smirnov two-sample tests and the unity of sex ratios was analysed using Pearson χ^2 proportions tests. Sizes at which the proportion of females accounted for 50% (P_{50}) and 95% (P_{95}) of sampled fish, and at which 50% (L_{F50}) and 95% (L_{F95}) of both males and females reached sexual maturity were estimated using binary logistic regression (Roa *et al.*, 1999). For maturity analyses, fish with gonads at stages 3 to 5 were considered sexually mature and only fish collected during the 2001 and 2002 spawning seasons (April-July) were included. Additionally, unsexed fish <80 mm L_F were regarded as immature and used to construct regression curves for both sexes.

To investigate the roles of the environmental variables in providing stimuli and suitable conditions for spawning during the 2000, 2001 and 2002 breeding seasons, multi-model procedures as described by Burnham and Anderson (2002) were used on a set of *a priori* regression models. This approach is superior to conventional variable selection procedures in multiple regression because information-theoretic measures (based on Akaike's Information Criterion (AIC)) rather than *P*-values are used to assess the fit of each model to the data. This measure is then used to rank all the models and, usually, a suite of models may be judged to be equally competitive. Inferences are then based on this suite of best-fitting models rather than selecting a single 'best fit' model and estimates of parameters can be averaged across all models. These procedures yield results that are less biased and more robust than the single 'best fitted' model selected by conventional approaches to multiple regression.

The successful use of these procedures requires a small set of carefully considered *a priori* models. In this study, the response variable was the proportion of fish in a sample that was spent (*spent* (%)), and all models included the mean I_G of females in the previous sample (I_{Gf} (%)) as an explanatory variable since spawning can only occur if a substantial number of females in the population have substantial ovarian mass. The full list of candidate models (M_1 - M_6) is presented in Table 2 (see Results), and their rationale is as follows. Temperature is often cited as a cue for spawning of teleosts (Jobling, 1995). Changes in lake level were potentially important since sustained, rapid rises in water level would likely inundate spawning habitat (rocky littoral areas, see Results) and keep it clear of fine sediments, whereas modest rises resulted in slow inundation of spawning habitat, which was usually accompanied significant sedimentation of otherwise suitable spawning substrata; falls in lake level usually resulted in deposition of sediments or dewatering of spawning habitat. For both temperature (*temp20* (°C)), and lake level (*level20* (mm)), the rate of change over the 20 d

prior to fish sampling (estimated as linear regression over time) was used since it was suspected that the fish would more likely respond to these variables integrated over time rather than a short-term event (i.e. rises in water levels are gradual in these lakes). Additionally, since the shortest time span between consecutive sample dates was 21 d, 20 d was used for these trends. There may be other habitat features in each lake that could generate differences between the lakes in the time of spawning, so the lake effect (*lake*) was included as a factor. Finally, the bathymetry and habitat maps of the two lakes suggested that the availability of spawning habitat would be more sensitive to water level in Lake Crescent than in Lake Sorell. Therefore, the most complicated model (M_6) that could be justified for this sized data set (termed the ‘global model’ by Burnham and Anderson (2002)) also included the interaction between lake level and lake (*lake:level20*).

None of the explanatory variables were strongly correlated with any of the others (all $|r| < 0.43$) and inspection of the diagnostic plots of M_6 suggested an arcsine-square-root transformation of the response variable was needed to improve the normality of residuals. Several other explanatory variables were considered, but were not included in the model set because they did not vary from year to year or between the lakes (e.g. photoperiod and lake \times temperature interaction). This is not to assert that these variables might never prove important to these fish; however, it is desirable not to over parameterise the global model given the size of the data set (Burnham & Anderson, 2002).

To rank the models, a variant of the AIC corrected for small sample sizes (AIC_c) was used. The model with the smallest AIC_c (AIC_{min}) is the closest fit with the data, and thus to rank each of the i models, AIC_c differences ($\Delta_i c$) are computed as $\Delta_i c = AIC_{c\ i} - AIC_{c\ min}$. Akaike weights (w_i) were then computed according to:

$$w_i = \frac{\exp(-0.5\Delta_i)}{\sum \exp(-0.5\Delta_r)}$$

which quantifies the weight of evidence given to each model. Because the w_i sum to 1 across the model set, models with substantially smaller w_i are poor fits to the data. It is unusual in ecological data for a single model to be clearly better than all the others (i.e. $w_i > 0.9$), and several models may have similar w_i , thus be equally likely to fit the data. To judge the relative importance of lake level and temperature, the w_i were summed across all models (w_+) that contained each of these two variables (Burnham & Anderson, 2002). The variable with the greater w_+ is the more important. The values of the individual parameter estimates for each explanatory variable were averaged across the entire model set for reporting, although the different units of these variables imply that most interest is in their signs rather than relative magnitudes. All statistical analyses were carried out in R v.2.2.0 (R Development Core Team, 2005) and the significance level for hypothesis tests was $P = 0.05$.

6.4. Results

Environmental variables

Abnormal climatic conditions just prior to, and early in the study, with an extended *El Niño*-induced drought between 1997 and 2001, caused below average and unseasonal rainfall in the catchment. As a result, lakes Crescent and Sorell experienced extremely low levels (maxima < 0.6 m below full supply levels) during this study, including record minima in 2000 in Sorell (802.620 m AHD) and in 2001 in Crescent (801.770 m AHD). Water level fluctuations followed similar trends in both lakes and there was considerable inter-annual variation with seasonal rises ranging from 0.200 to 0.970 m and falls from 0.280 to 0.640 m. Minimum and maximum levels in both lakes occurred during March-May and October-December, respectively. Whilst fluctuations in both lakes were similar, levels in Lake Sorell usually

began to rise slightly earlier than those in Lake Crescent in response to late autumn – winter rainfall due to direct seasonal input from the primary tributary, Mountain Creek. Fluctuations in Lake Sorell were also typically more rapid because of this input, and regulation of flows between the lakes and discharge from Lake Crescent.

Water temperatures in both lakes were similar between 2000 and 2002 with only a marginally significant difference between the monthly means of each lake (paired *t*-test; d.f. = 35, $P = 0.045$). However, the mean difference in the average monthly temperatures of the lakes was small ($<0.2^{\circ}\text{C}$) and not biologically meaningful. Temperature cycles had strong seasonal fluctuations with mean (\pm S.D.) monthly temperatures being $<5^{\circ}\text{C}$ during winter (minimum = $4.2 \pm 0.9^{\circ}\text{C}$ during June) and $>14^{\circ}\text{C}$ during summer (maximum = $16.4^{\circ}\text{C} \pm 1.7^{\circ}\text{C}$ during February). Day length in the study region varied from 9.08 to 15.27 h in June and December respectively.

Sexual dimorphism and sex ratio

A total of 2978 *G. auratus* were examined and fish from both populations covered large size ranges (Lake Crescent = 38-235 mm L_F ; Lake Sorell = 31-179 mm L_F). Sexual composition of pooled lake samples was dominated by females (60%) followed by males (31%) and unsexed (mostly immature) fish (9%). Female size-range differed significantly from that of males in Lake Crescent (Kolmogorov-Smirnov test; $n = 1488$, $P < 0.001$) and Lake Sorell (Kolmogorov-Smirnov test; $n = 1208$, $P < 0.001$) and in both lakes, mean (\pm S.D.) female L_F (Lake Crescent = 95 ± 21 mm; Lake Sorell = 92 ± 22 mm) was greater than males (Lake Crescent = 80 ± 17 mm; Lake Sorell = 77 ± 14 mm). The largest female measured was 235 mm $L_{F \text{ max}}$ and largest male was 179 mm $L_{F \text{ max}}$.

Overall male to female ratios were female-biased and differed significantly from unity in Lake Crescent (1:2.0) (Pearson χ^2 proportions test; d.f. = 1, n = 1488, P < 0.001) and Lake Sorell (1:1.8) (Pearson χ^2 proportions test; d.f. = 1, n = 1208, P < 0.001). In pooled population data, the proportion of females increased in progressive length classes (Fig. 5a), with P_{50} (\pm S.E.) = 71 ± 1 mm L_F and P_{95} = 138 ± 3 mm L_F .

Sexual maturation

Over the 30 month study period, 47% of fish sampled had gonads at stage 3 (late phase of development) or beyond. The smallest stage 4 (ripe) male measured 51 mm L_F and smallest female 60 mm L_F . The L_{F50} (\pm S.E.) of pooled population data for males was 52 ± 2 mm and females 76 ± 1 mm, while the L_{F95} for males was 75 ± 3 mm and females 87 ± 1 mm (Fig. 5b). Individuals appeared to remain sexually active at large sizes, with the largest stage 4 male measuring 171 mm L_F (100% of $L_{F \max}$) and largest female 181 mm L_F (77% of $L_{F \max}$). However, the mean L_F (\pm S.D.) of stage 4 fish (males: 78 ± 15 mm; females: 103 ± 19 mm) were <50% of their respective $L_{F \max}$.

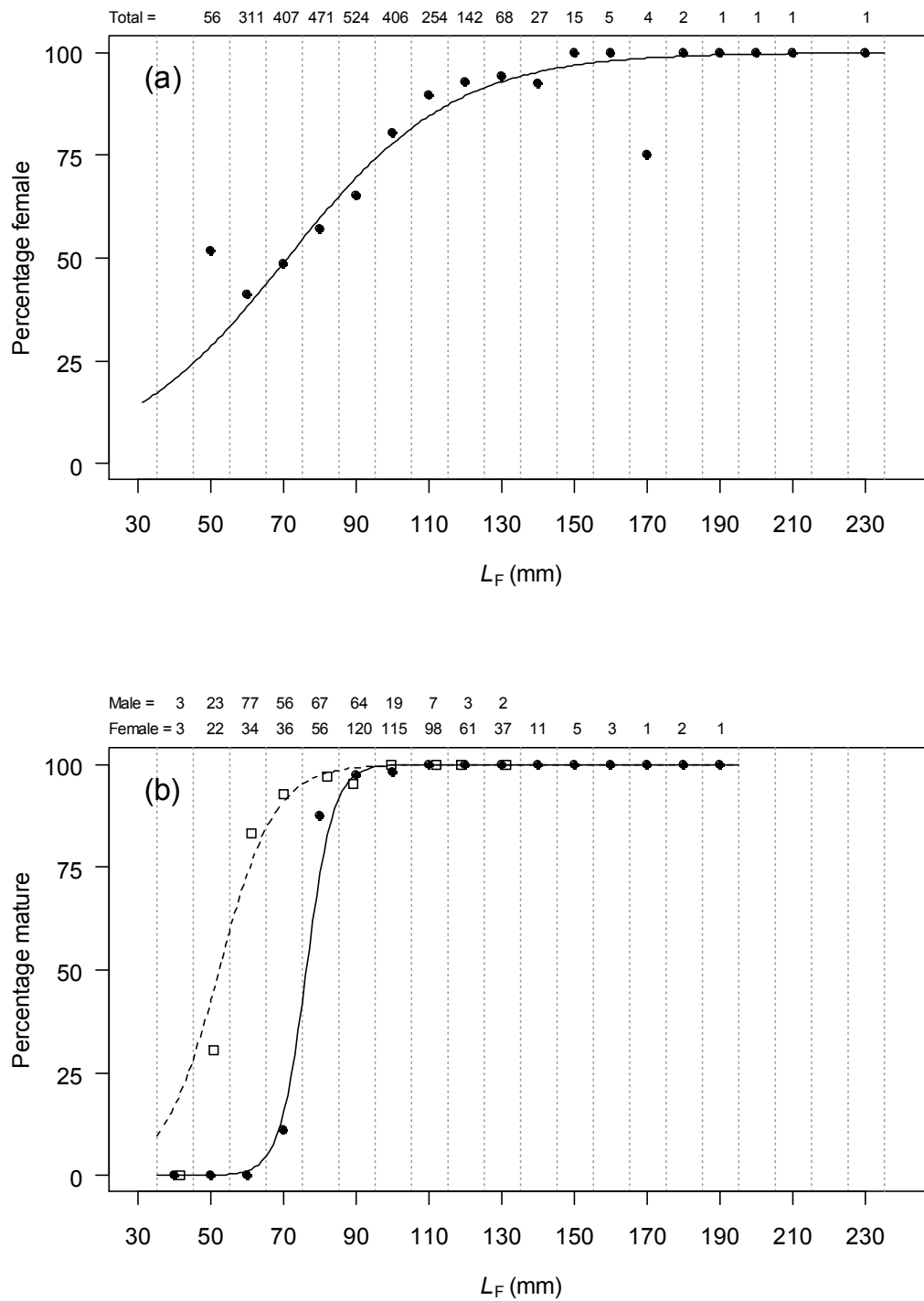


Fig. 5. Percentage of (a) females and (b) mature males (--) and females (–) as a function of L_F of *Galaxias auratus*. Maturity analyses are based on fish collected from lakes Crescent and Sorell between April and July of 2001 and 2002, whilst sexual proportions are from fish collected between July 2000 and December 2002. The proportion of sexually mature fish or females within each sequential 10 mm length class is plotted using logistic regression. Samples sizes for length increments are shown.

Gonadal cycle and fecundity

Annual cyclic patterns in male and female mean monthly I_G values were similar in both populations, although the timing and extent of periods when maximum I_G values were recorded varied (Fig. 6). Mean I_G of males and females were generally <3% between October and December (except females in Lake Sorell during October 2000). In both populations, mean I_G of both sexes increased from January onwards while water temperatures and photoperiod were decreasing, and males developed earlier than females in both populations. In Lake Crescent (Fig. 6b), during 2001, peak I_G values for males (>16%) occurred from March to July and from May to July for females (>15%). During 2002, the peak periods (>13%) for both sexes respectively began in the same months, but extended into August. Conversely, in Lake Sorell (Fig. 6a) the periods where peak I_G values occurred were shorter in duration. Males in Lake Sorell maintained I_G values >14% between February and May during 2001 and I_G values >16% between April and July during 2002. Sorell females had I_G values >12% between April and June during 2001 and an abrupt peak I_G of 16.1% during April 2002. The maximum and minimum individual I_G values were 0.06% and 37.8%, and 0.09% and 33.2% for males and females, respectively.

Trends in I_G were complemented by synchronous progression of gonadal stages of development in pooled sex data in both populations (Fig. 7). Minimal development occurred between October and December, with >85% of fish having stage 1 (resting) gonads (with the exception of fish in Lake Sorell during October 2000). The prominence (5-47%) of transitional developing stages 2 and 3 was brief (January-March) and the majority (49-100%) of fish were stage 4 (ripe) in April-June. Stage 5 (spent) fish appeared as early as April in both lakes, but were most common (>22%) during July-September of 2001 and 2002.

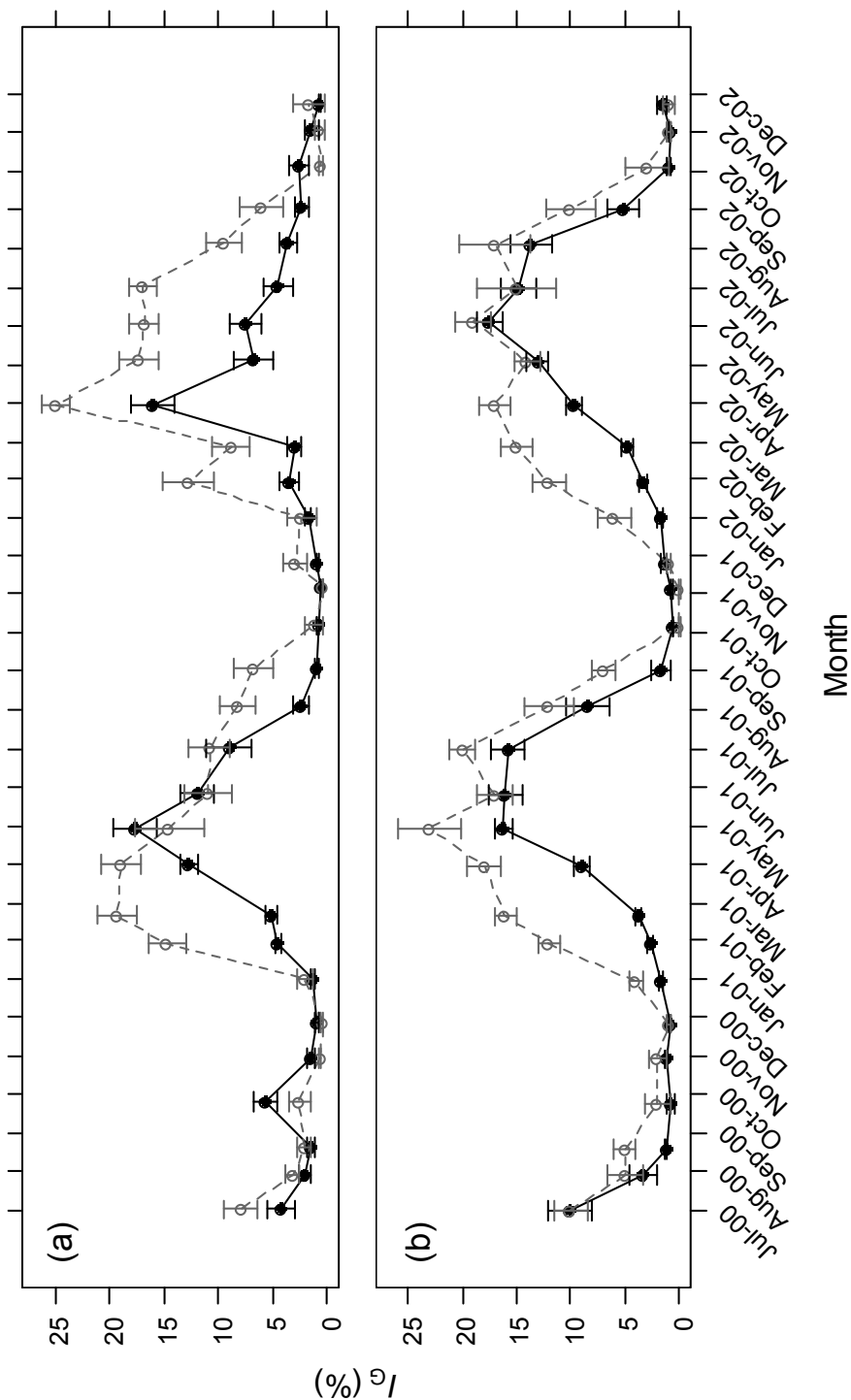


Fig. 6. Mean (± 1 S.E.) monthly gonado-somatic indices (I_G) of male (\circ) and female (\bullet) *Galaxias auratus* in (a) Lake Sorell and (b) Lake Crescent between July 2000 and December 2002.

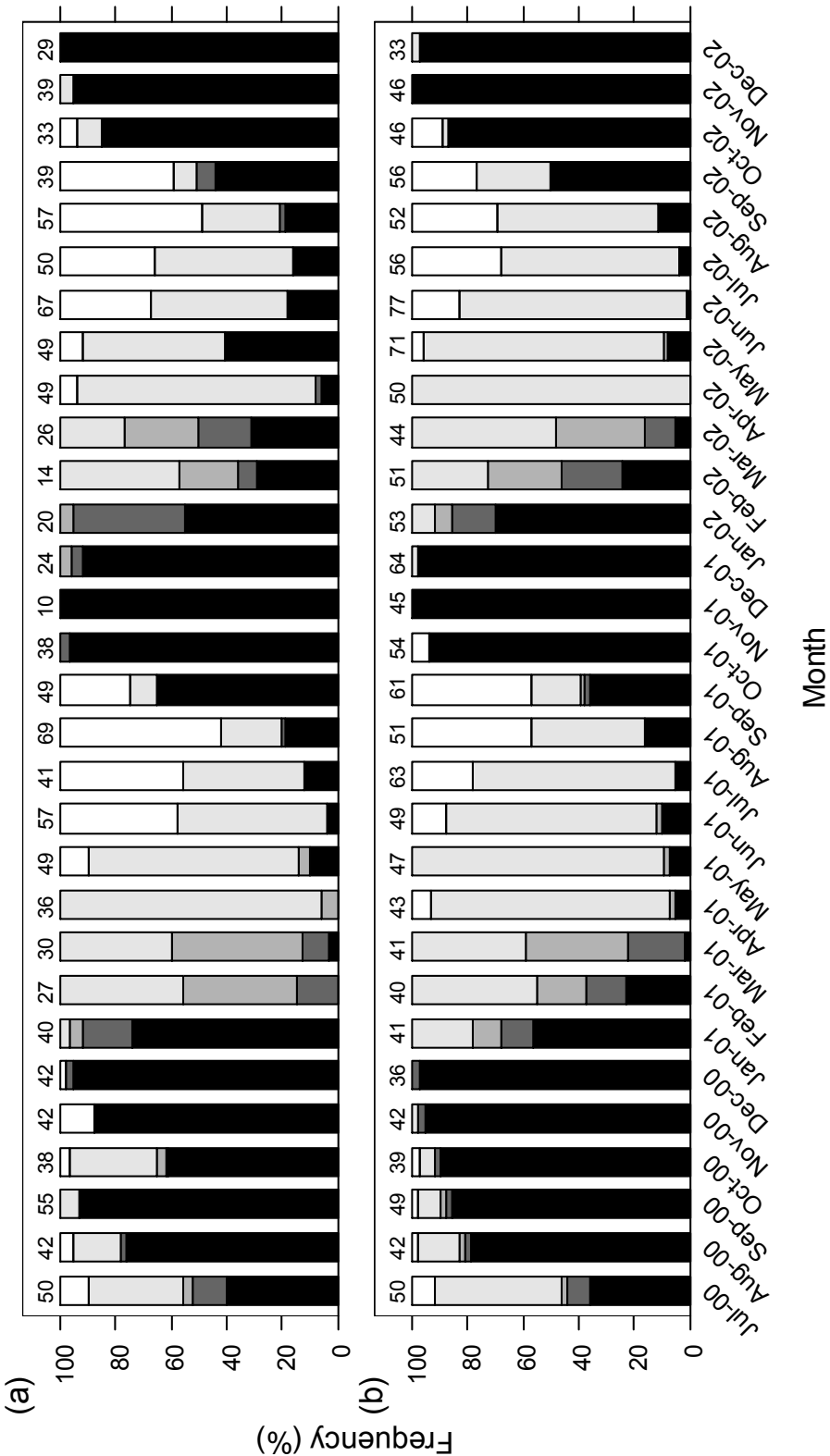


Fig. 7. Frequency of pooled male and female *Galaxias auratus* in (a) Lake Sorell and (b) Lake Crescent at each gonadal stage of development (■, 1; ■, 2; ■, 3; ■, 4; □, 5), July 2000 – December 2002. Sample sizes are shown for each month.

The fecundity (F) of fish ($n = 87$) ranging in size from 71 to 181 mm L_F varied from 619 to 14478. The relationship between F and L_F was: $F=0.0018 L_F^3$ ($R^2=0.72$) (Appendix 5).

Spawning

Habitat

In both lakes, spawning sites were found in littoral areas (0.2-0.6 m deep and usually <10 m from the shore) of cobble substrate (*c.* 20-250 mm diameter) mostly during winter – early spring in 2000-2002. Other dominant littoral habitats searched included areas of sand and sediment substrates and submerged woody debris, but no eggs were found in habitats other than those dominated by rocky substrates. On a single occasion, 3 eggs were found adhered to a single stem of the aquatic macrophyte common water milfoil *Myriophyllum* sp., which was growing in an area of rocky substrate.

Fertilised eggs (*c.* 1.5 mm in diameter) were transparent and adhesive, and developing embryos were clearly visible in advanced eggs. Eggs were scattered on the top and sides of rocks, positioned individually and occasionally in small clusters of 2-5. The locations of spawning sites moved progressively further in-shore as water levels rose, meaning the depth of spawning sites remained reasonably constant and eggs were continually deposited on new areas of substrate. Sedimentation of suitable spawning substratum and some spawning sites was evident following periods of strong winds, particularly in Lake Crescent. Algal growth on rocks at spawning sites increased after the incubation of most eggs had been completed (*i.e.* early October onwards).

The availability of suitable spawning habitat differed between the lakes. In Lake Crescent, littoral areas of rocky substrate were only found at four defined sites (Fig. 8) and at all locations this habitat was inundated at water levels >802.20 m AHD (Fig. 9). Conversely, in several regions in Lake Sorell (whilst not quantitatively surveyed) contained vast littoral areas of rocky substrate which generally extended well into the lake's basin.

Timing and environmental conditions

Peaks and subsequent decreases in mean I_G values (Fig. 6), frequencies of stage 5 fish (Fig. 10), and the density of eggs at spawning sites in 2001 and 2002 indicate *G. auratus* spawn over protracted periods between late April and early September. Females appeared to determine the time of spawning, with males developing gonad mass much earlier than females and maintaining it for longer (Fig. 6). Fish in Lake Sorell generally spawned earlier than those in Lake Crescent. During both 2001 and 2002, decreases in mean I_G values (Fig. 6) and increases in proportions of spent fish (Fig. 10) occurred during May-August in Lake Sorell and June-September in Lake Crescent. Maximum densities (range = 20-67 eggs sweep⁻¹) of eggs at spawning sites in 2001 and 2002 were observed in late August – early September in Lake Crescent and early July in Lake Sorell. At these times, average daily mean water temperatures were 4.8°C (range = 1.4-9.7°C) and water levels were rising (Fig. 10), and day length was <11.8 h.

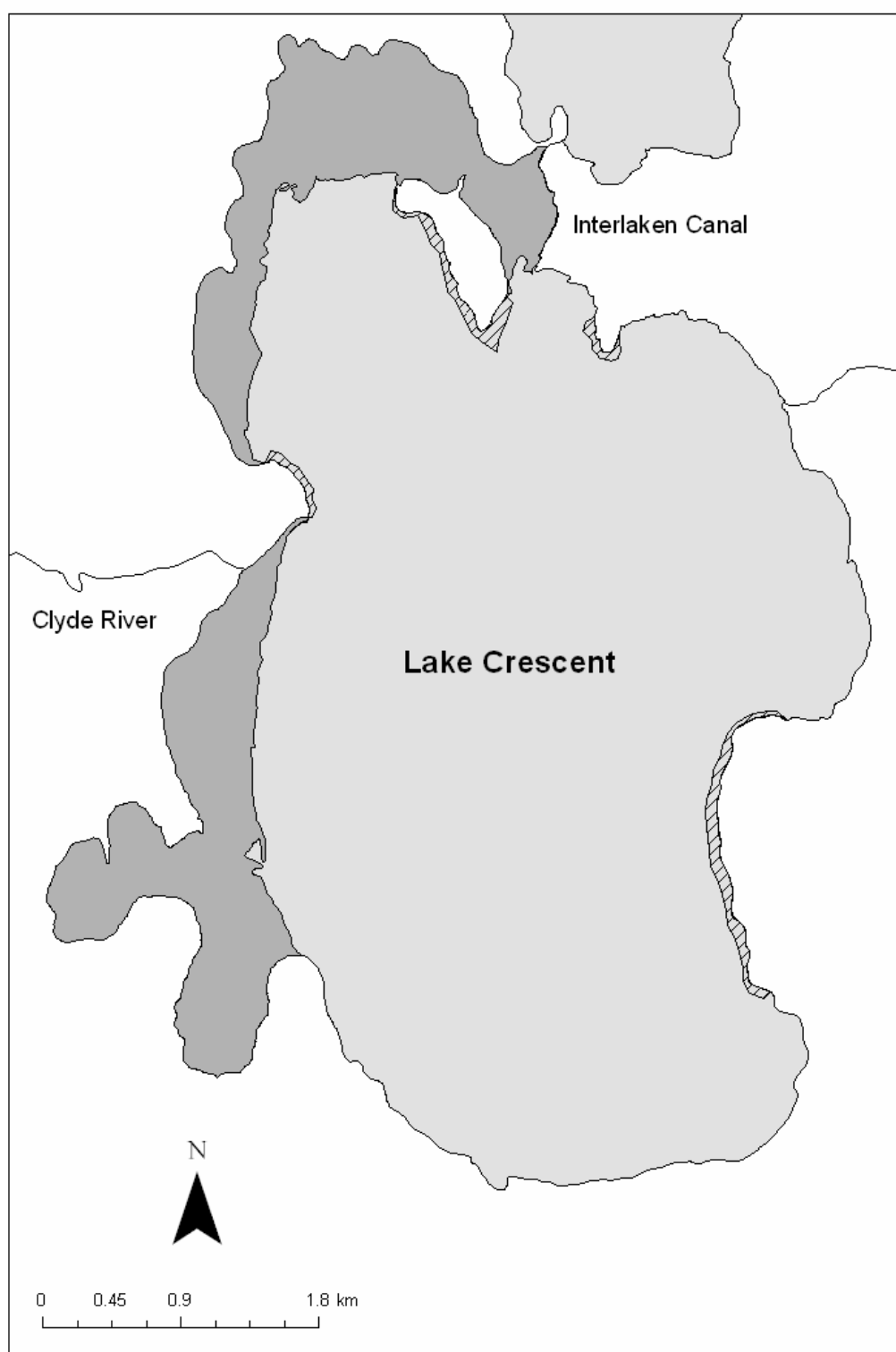


Fig. 8. Littoral areas of rocky substrate (regions filled with diagonal lines) in Lake Crescent surveyed during March 2002. Lake perimeters are at full supply levels. Wetlands areas are depicted by dark shading.



Fig. 9. Dewatered littoral area of rocky substrate on a northern shore in Lake Crescent during 2001. This area would provide suitable spawning habitat for *Galaxias auratus* when inundated.

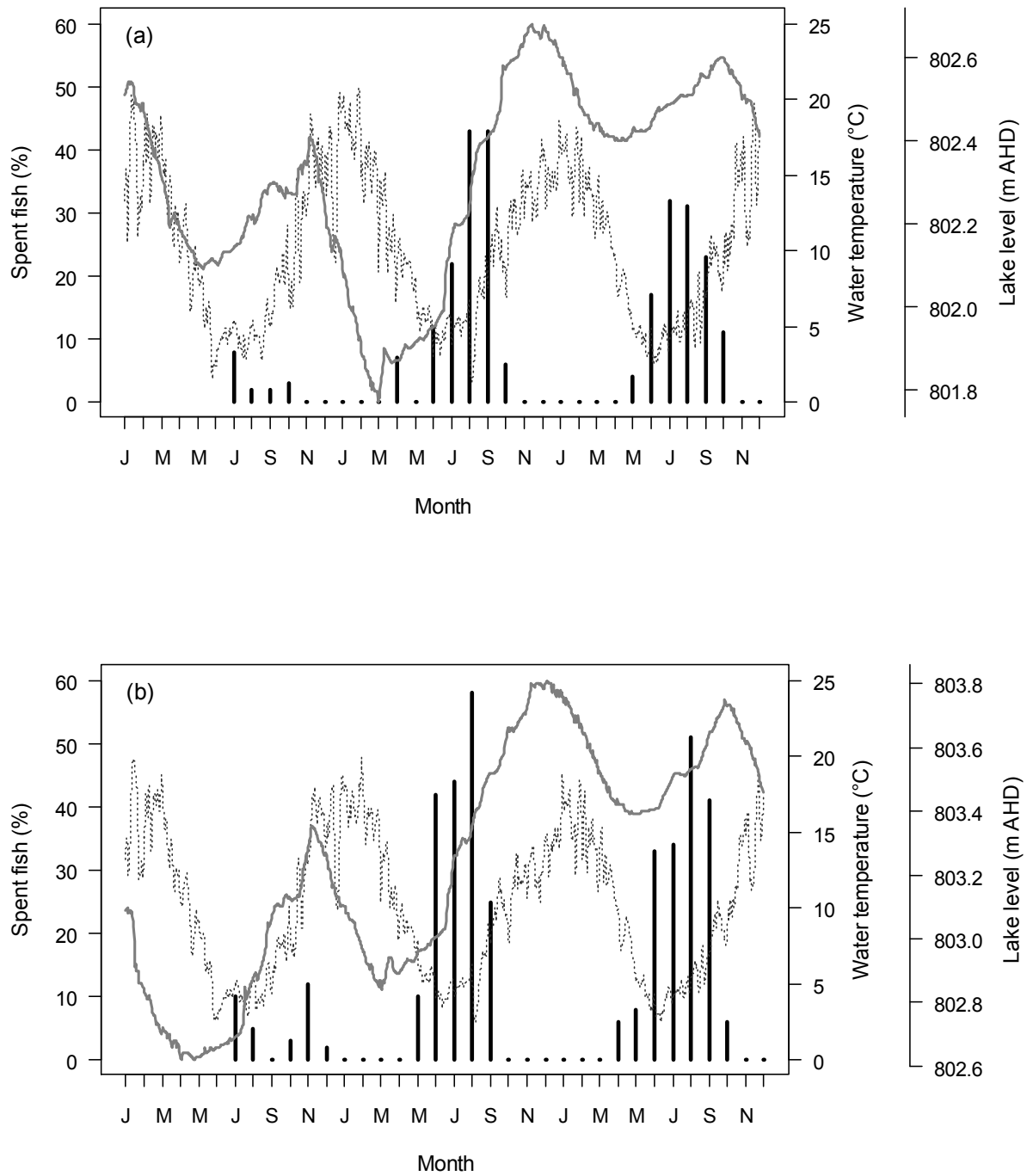


Fig. 10. Per cent of spent (stage 5) *Galaxias auratus* (■) and daily mean water temperatures (—) and water levels (···) in (a) Lake Crescent and (b) Lake Sorell between January 2000 and December 2002. Water levels are measured in metres of elevation according to the Australian Height Datum (m AHD).

The proportion of spent fish on a sampling occasion was best explained ($w_+ = 0.984$) by models (M_3 , M_4 , M_5 and M_6), which included the rate of change in water levels over the previous 20 days (*level20*) (Table 2). The best performing model (M_3) included only *level20*; however, all other models including this variable also had a Δ_{ic} of <2 , therefore, their performance was approximately equal (Burnham & Anderson, 2002). The sign of the averaged estimate for this parameter was positive (value of 0.0177) showing that the proportion of spent fish is positively related to the rate of lake level change. Although the rate of change in temperature (*temp20*) was also positively related to the proportion of fish spawned (average parameter estimate = 0.1140), it was less important ($w_+ = 0.644$) than *level20*. Overall, Lake Sorell had slightly greater proportions of spent fish across all sampling occasions than Lake Crescent (average parameter estimate = 0.0383). The inclusion of the *lake:level20* interaction in the subset of competitive models (Table 2) suggested differential responses between the populations to changes in lake level. This is consistent with the known differences in available habitats between the two lakes: Sorell has an abundance of suitable rocky spawning areas even at low levels, whereas these habitats are more limited in Lake Crescent and not available until levels rise above 802.20 m AHD.

Table 2. Multiple linear regression results of models of environmental conditions used to predict the timing of *Galaxias auratus* spawning in lakes Crescent and Sorell during 2000-2002 based on the occurrence of spent fish. Models are ranked in order of best fit according to Akaike Information Criterion (AICc), Δ_{iC} is the difference in AICc and w_i is the AICc weight.

Rank	Model	Model*	AICc	Δ_{iC}	w_i
1	M ₃	$spent \sim I_{df} + level20$	-30.53	0.000	0.349
2	M ₅	$spent \sim I_{df} + level20 + temp20 + lake$	-30.17	0.362	0.292
3	M ₄	$spent \sim I_{df} + level20 + temp20$	-29.17	1.360	0.177
4	M ₆	$spent \sim I_{df} + level20 + temp20 + lake + lake:level20$	-29.04	1.489	0.166
5	M ₂	$spent \sim I_{df} + temp20$	-23.23	7.300	0.009
6	M ₁	$spent \sim I_{df}$	-22.61	7.923	0.007

*Modelling variables include the occurrence of spent fish ($spent$ (%)), mean I_G of females in previous sample (I_{df} (%)), change in daily water level over the previous 20 days ($level20$ (mm)), change in mean daily water temperature over the previous 20 days ($temp20$ (°C)), differences between lakes ($lake$), and the lake \times water level interaction ($lake:level20$).

6.5. Discussion

Sexual dimorphism, sex ratio and maturation

In this study, *G. auratus* females (v. males) matured later and attained greater sizes, and overall they appeared to be more abundant. Based upon age and growth data from a related study (Chapter 5), length-at-maturity estimates (males v. females: $L_{F50} = 52$ v. 76 mm, $L_{F95} = 75$ v. 87 mm) indicate some males mature in their first year and the remainder in their second, whereas females mature in their second year. Age-at-maturity varies amongst galaxiid species, from short-lived *Galaxiella* spp. with predominantly annual life cycles (Humphries, 1986; Pen *et al.*, 1991; Pen *et al.*, 1993), to longer-lived (commonly >3 years) species such as banded kokopu *Galaxias fasciatus* Gray with males and females maturing at 2 and 4 years respectively (Hopkins, 1979b). In both a lacustrine population of the closely related spotted galaxias *Galaxias truttaceus* (Valenciennes) on the TCP (Humphries, 1989) and a diadromous population in south-western Australia (Morgan, 2003), some males matured in their first year and the remainder of the populations in their second.

Larger *G. auratus* were predominately females in lakes Crescent and Sorell (i.e. $P_{95} = 138$ mm L_F) and sex ratios of sampled fish from both populations were strongly female-biased (c. 2:1). Several other researchers have found larger fish tend to be females in lacustrine (Pollard, 1971; Rowe *et al.*, 2002a), non-diadromous riverine (Bonnett, 1990; O'Connor & Koehn, 1991; Pen & Potter, 1991b), and some diadromous (McDowall, 1968; Hopkins, 1979a; Morgan, 2003) populations of *Galaxias* spp. Female-biased sex ratios have also been reported for dwarf inanga *Galaxias gracilis* McDowall (Rowe & Chisnall, 1996a) and lacustrine populations of common galaxias *Galaxias maculatus* (Jenyns) (Pollard, 1971; Chapman *et al.*, 2006). Sex ratio estimates can be biased by methods of capture, time of sampling, and sub-sampling of fish from catches. However, given the large size-range of

G. auratus collected using fyke nets (31-235 mm L_F) and relatively large monthly sample sizes (mean = 50 fish), the overall female-biased sex ratios in each population are likely to be reasonable estimates of proportionality. These findings suggest greater longevity or growth in females is due to either physiological differences or mechanisms which increase mortality in males at smaller sizes (e.g. aggressive spawning behaviour). The testes of teleosts frequently represent a much lower proportion of the body weight than ovaries (Wootton, 1998); however, male *G. auratus* had larger gonads (v. body weight) than females which they maintained for longer periods. The greater reproductive investment of males may decrease their longevity (or growth) by reducing feeding rates during gonad development and spawning. Impaired locomotion at these times due to large-sized testes (i.e. I_G values >30%) would also increase their vulnerability to predation by introduced salmonids.

Gonadal cycle and spawning

Previously, *G. auratus* were thought to spawn during spring (Fulton, 1990; McDowall & Fulton, 1996; Allen *et al.*, 2002). However, this study shows that whilst the peak investment period in the gonadal cycle of *G. auratus* is similar to that of lacustrine populations of *G. truttaceus* (Humphries, 1989) and *G. maculatus* (Pollard, 1971) in south-eastern Australia, spawning occurs mostly during winter. Both sexes of *G. auratus* begin to develop gonad mass when day length is near its maximum and the rate of development is rapid between January and May (mean I_G values increase from <7% to >13% in both sexes), when day length and water temperature decline markedly suggesting these are important stimuli. Development during this time would allow fish to utilise the relatively short periods of warmer water temperatures (i.e. >10°C) in these highland lakes and their likely increases in food resources. Similarly to lacustrine *G. truttaceus* of the TCP (Humphries, 1989), male *G. auratus* develop gonad mass and reach ripe condition earlier than females (some running-ripe in late February)

indicating that females dictate when spawning occurs. By April, many males and females are ripe and some spawning activity was recorded in both populations, but most fish spawn 2-4 months later during winter. Whilst this delay would give first year males further time to mature, allowing them to contribute to spawning, it is likely to be associated with a wait for suitable environmental conditions.

Whilst *G. auratus* in these neighbouring lakes had extended winter-based spawning periods, peaks in spawning activity occurred earlier in Lake Sorell than in Lake Crescent during both 2001 and 2002. Multiple regression modelling indicated that rises in water levels over the 20 days prior to monthly sampling was the most important factor in predicting the occurrence of spent fish in both lakes. These temporal variations in spawning are likely to be associated with the differing hydrological regimes of the lakes (e.g. levels rise earlier in Lake Sorell). Changes in water temperature did not appear to be an important spawning cue with spawning periods encompassing a large range of temperatures (1.4-9.7°C); however, further finer-scale temporal sampling would be needed to evaluate its importance.

The winter spawning of *G. auratus* is unusual for a non-diadromous galaxiid. Selective pressures in freshwater teleost populations usually ensure spawning occurs at times that provide subsequent optimal conditions (i.e. abundant food resources and favourable temperatures) for larvae during their first growing season (Wootton, 1998). The two life history strategies of galaxiid fishes illustrate this pattern; generally diadromous galaxiid populations spawn in late autumn – early winter, whilst non-diadromous populations in lotic and lentic waters spawn in late winter – early summer (Humphries, 1989; Pen & Potter, 1991a). These strategies allow larvae and juveniles to encounter conditions that are advantageous to their growth and survival in their respective estuarine/marine and freshwater

nursery habitats. Other lacustrine galaxiid populations on the TCP follow the usual pattern of non-diadromous galaxiids with *G. truttaceus* (Humphries, 1989) and Great Lake paragalaxias *Paragalaxias eleotroides* McDowall and Fulton (Fulton, 1982) spawning in spring and Shannon paragalaxias *Paragalaxias dissimilis* (Regan) in summer (Fulton, 1982). Whilst *G. auratus* larvae occur in the water column during winter, the peak hatching period is early-mid spring (Chapter 8). Therefore, *G. auratus* appear to respond to different spawning stimuli (i.e. rising water levels) to those of other galaxiids on the TCP, which lengthens incubation of their eggs. However, their overall life cycle follows the usual pattern of a non-diadromous lacustrine galaxiid.

Fecundity and spawning site

Egg size and fecundity of galaxiids vary both between and within species which follow either diadromous or non-diadromous life history strategies (see Pen & Potter(1991a)). Generally, fecundity is proportional to body size (Humphries, 1989) with smaller-sized species (commonly <80 mm L_F) having fewer eggs (commonly <1000), whilst larger-sized galaxiids (particularly those which are diadromous or have close diadromous ancestry) are more fecund (c. 1000-20000 eggs). The results of this study indicate that the fecundity (71 to 181 mm L_F fish had from 619 to 14478 eggs) and egg size (c. 1.5 mm diameter) of *G. auratus* are consistent with values for diadromous and lacustrine stocks of the closely related *G. truttaceus* in Tasmania (Humphries, 1989) and also show similarities to other larger-sized species such as *G. fasciatus* (Hopkins, 1979b) and climbing galaxias *Galaxias brevipinnis* Günther (O'Connor & Koehn, 1998).

Although the negative impacts of introduced salmonids on galaxiids have been well documented (Crowl *et al.*, 1992; McDowall, 2003), there is still limited information about the

mechanisms associated with these interactions. Interestingly, some other galaxiids in Tasmania that have much lower fecundity have also had their distributions severely reduced by introduced salmonids (e.g. Swan galaxias *Galaxias fontanus* Fulton (Crook & Sanger, 1998a) and Clarence galaxias *Galaxias johnstoni* Scott (Crook & Sanger, 1998b)). Therefore, the relatively high fecundity of *G. auratus* may have contributed to this species coexistence with introduced salmonids for over a century despite heavy predation (Stuart-Smith *et al.*, 2004). Based on this information, studying the biology of other threatened galaxiids may also provide insight into the resilience of their populations to impacts from introduced fishes.

Deposition of eggs in shallow marginal habitats is common amongst galaxiids which occupy both lotic (O'Connor & Koehn, 1991; Allibone & Townsend, 1997; O'Connor & Koehn, 1998; Moore *et al.*, 1999; Charteris *et al.*, 2003) and lentic (Pollard, 1971; Fulton, 1982; Humphries, 1989) environments. *Galaxias auratus* spawned demersal adhesive eggs over cobble substrates in littoral areas. Whilst spawning behaviour was not observed, the dispersed arrangement of eggs at spawning sites indicates that spawning takes place mid-water and eggs settle onto the substrate rather than being laid in defined nests. The occurrence of eggs on a stem of aquatic vegetation on a single occasion in Lake Sorell and on submerged vegetation in an off-stream agricultural water storage containing a refuge population of this species (S. A. Hardie, unpubl. obs.), suggest that *G. auratus* will use aquatic vegetation when available. Furthermore, the occurrence of spawned eggs on fyke nets (Hardie *et al.*, 2005) also shows flexibility in spawning site selection and perhaps the primarily requirement for surfaces for which eggs can adhere. Other *Galaxias* spp. also spawn on rocks (O'Connor & Koehn, 1991; Allibone & Townsend, 1997; Moore *et al.*, 1999) and submerged vegetation (Pollard, 1971; Humphries, 1989; Charteris *et al.*, 2003).

Influence of lake hydrology

Hydrological variables play important roles in the reproduction of some galaxiids. For example, high flows in rivers (Pen & Potter, 1991a; O'Connor & Koehn, 1998; Charteris *et al.*, 2003) or inflowing tributaries of lakes (Pollard, 1971) have been associated with the time of spawning. This study suggests that lake hydrology strongly influences the reproductive cycle of *G. auratus* by possibly providing a stimulus for spawning and controlling the availability of spawning habitat. Due to relatively low water levels, the quantity of suitable spawning habitat for *G. auratus* was much less in Lake Crescent than in Lake Sorell and its availability there was significantly influenced by water level fluctuations. Some teleost species delay spawning until suitable spawning substrate is available (Lam, 1983; Jobling, 1995). This appeared to occur in Lake Crescent during 2001, when spawning occurred 2-3 months later than in Lake Sorell, during which time almost no spawning habitat was submerged (i.e. levels were <802.200 m AHD). The timing and magnitude of rises in water level may be important for successful recruitment of this species as delays could cause the uncoupling of larval emergence and availability of suitable food resources (Cushing, 1990), resulting in slower growth in later spawned fish (Ludsin & DeVries, 1997; Pine *et al.*, 2000). Based on this information, the production of the Lake Crescent population is likely to be significantly constrained by limited availability of spawning habitat during periods of relatively low water levels.

Similar to other *Galaxias* spp. which have permanently inundated spawning sites (O'Connor & Koehn, 1991; Allibone & Townsend, 1997), *G. auratus* did not appear to show parental care of eggs (in fact cannibalism of eggs occurred (Hardie, 2003)); therefore, egg survival depends on environmental conditions. The relatively shallow (20-60 cm) spawning sites of *G. auratus* mean their eggs are prone to mortality via disturbance from wind-induced wave

action (Johnson, 1961; Rupp, 1965) including translocation to unsuitable habitats (Ventling-Schwank & Livingstone, 1994), and desiccation due to decreases in water levels during incubation (Rupp, 1965). Clean rocky substrates may also be important for spawning and the survival of eggs. Loads of particulate (mostly organic) material in the water of these lakes are inversely related to the water levels (Uytendaal, 2003, 2006). During this study, sedimentation at spawning sites was observed following strong wind events, particularly in Lake Crescent during 2001 when levels were very low (S. A. Hardie, unpubl. obs.). Egg mortality was not assessed, but sedimentation is known to cause mortality of teleost eggs (Bruton, 1985). Algal growth on rocks in littoral areas late in the spawning season (early October onwards) could also reduce egg survival (Gafny *et al.*, 1992) forcing *G. auratus* to spawn in winter before algal growth occurs. When available, aquatic vegetation in adjacent wetlands to lakes Crescent and Sorell may provide more secure spawning habitat (i.e. not influenced by sedimentation), and connection of these large areas would significantly increase the amount of spawning habitat in both lakes.

This study suggests that water levels strongly influence the spawning of *G. auratus*; however, further explicit testing of the relative importance of proximal environmental cues via controlled experiments is required to define the roles of temperature and those driven by lake level. Management options for these and other shallow lakes on the TCP are limited to manipulation of water level regimes, and this study provides evidence that seasonal hydrological cycles (i.e. rises during late autumn – winter) and a base ‘environmental level’ of 802.20 m AHD in Lake Crescent during autumn are critical to *G. auratus* populations. Compared with interactions with exotic species (e.g. predation by salmonids), stressors associated with lake hydrology have previously been overlooked as threats to *G. auratus*. Future changes in lacustrine hydrological regimes of the TCP caused by climatic trends,

episodic droughts and increased hydroelectric and agricultural demands are likely to pose serious threats to galaxiids in this region. Detailed bathymetries coupled with mapping littoral habitats have been useful in linking fish biology and hydrological changes elsewhere (see Rowe *et al.*, (2002b)). This is a prominent information gap for the TCP that urgently needs to be filled before more sophisticated risk analyses can be attempted for this region's galaxiid fauna.

7. Spawning-related fungal infection of golden galaxias, *Galaxias auratus* (Galaxiidae)*

*Published (with minor adjustments) as Hardie S.A., Pyecroft, S.B., Barmuta L.A. & White R.W.G. (2007) in *Journal of Fish Biology* 70, 1287-1294.

7.1. Abstract

Golden galaxias *Galaxias auratus* with dermal fungal lesions occurred periodically in monthly samples collected over 2.5 years from lakes Crescent and Sorell, Tasmania, Australia. Affected fish were most abundant late in spawning seasons (July-September) and their occurrence was effectively modelled by the progression of gonadal stages of development. Histopathological and microbiological examination of samples of affected fish found skin lesions to be characteristic of fungal dermatitis and a *Saprolegnia* sp. was identified as the causative agent. The periodicity of infection is likely to be linked to spawning-related factors (e.g. aggressive social interactions) and it is probable that saprolegniasis is associated with seasonal mortality. However, further work is required to determine mechanisms which influence fungal colonisation and the mortality rates of infected *G. auratus*.

Key words: Galaxiidae; dermatitis; *Saprolegnia*; reproduction; mortality

7.2. Introduction

Fishes of the family Galaxiidae are largely restricted to fresh water and estuarine habitats in cool temperate regions of several land masses in the Southern Hemisphere (Allen *et al.*, 2002), and are particularly prominent in the native piscine fauna of southern Australia and

New Zealand (McDowall & Fulton, 1996; McDowall, 2000). Whilst various aspects of the biology and ecology of many galaxiid species have been examined, sources of mortality have received little attention, and there is limited information about diseases and parasites that affect galaxiids (Blackburn, 1950; Pollard, 1974; Smith & Hickman, 1983; Viozzi *et al.*, 2000; Morgan, 2003). The life-span of galaxiid fishes varies (1 to >20 years), but most are relatively short-lived (commonly <6 years), and females often achieve greater ages than males (Pollard, 1971; Bonnett, 1990; Pen & Potter, 1991b). The latter suggests that mechanisms which result in mortality may be sex-specific.

Physico-chemical, physiological and social stressors can predispose freshwater teleosts to infectious diseases (Austin & Austin, 1987) and spawning is known to be a critical period when the risk of infection increases (Wootton, 1998). Some short-lived migratory galaxiid species are thought to incur mass post-spawning mortality (Blackburn, 1950; McDowall, 1968); furthermore, spawning-related mortality of diadromous (O'Connor & Koehn, 1998) and non-diadromous (Pollard, 1971) populations has previously been reported.

The non-diadromous golden galaxias *Galaxias auratus* Johnston is endemic to lakes Crescent (42°10' S; 147°10' E) and Sorell (42°6' S; 147°10' E) on the Central Plateau of Tasmania, Australia. These highland lakes (*c.* 800 m a.s.l.) are relatively shallow (mean depths <3.5 m) and turbid (mean of *c.* 130 NTU during study). *Galaxias auratus* is considered to be 'threatened' (Hardie *et al.*, 2004) and has recently been the subject of an extensive life history study (Hardie, 2003; Hardie *et al.*, in press; Chapters 5 and 8) during which monthly samples were collected from littoral habitats in both lakes between July 2000 and December 2002. During this work, individuals with focal dermatitis periodically occurred in both populations.

This study aimed to identify the possible aetiological agent(s) and investigate the timing of its occurrence in relation to the reproductive cycle of *G. auratus*.

7.3. Materials and methods

Random sub-samples of *G. auratus* ($n = 10-77$) taken from monthly catches collected using electrofishing (July-September 2000) and fyke netting (October 2000 – December 2002) were euthanased (Aqui-s™) and preserved (70% ethanol). Each fish (Lake Crescent: $n = 1624$, Lake Sorell: $n = 1354$) was measured, fork length (L_F , mm), and sex was determined by macroscopic and microscopic examination of gonads. Gonadal stages of development (D_S) of ovaries and testes (ordinal scale from 1 = resting to 6 = spent adults) were assigned where gonads were visible, and the presence and location of dermatitis lesions was also recorded. Log-linear modelling, with generalized linear models (GLM) (poisson distribution, log-link), was performed in R version 2.2.0 (R Development Core Team, 2005) to analyse the resulting three-way table of frequencies defined by the presence or absence of skin lesions crossed with lake and D_S (Venables & Ripley, 2002). Sub-samples of infected fish were examined from months where they were most abundant in each lake during all three years (Lake Crescent: $n = 8$, Lake Sorell: $n = 32$) using standard histopathological methods. A further sample of affected fish (Lake Crescent: $n = 8$, Lake Sorell: $n = 4$), collected by fyke netting in August 2005, was also analysed using histopathological and microbiological methods in order to characterise more fully the lesions and allow identification of aetiological agent(s).

7.4. Results

The skin lesions on all 12 fish collected in 2005 were characteristic of a fungal dermatitis and the fungus was identified as a *Saprolegnia* sp. (Saprolegniaceae). Pathology was characterised by fungal hyphal mats (Fig. 1a) overlying areas of scale loss and epithelial erosion. Fungal

hyphae penetrated deeper layers of the dermis and tracked down fascial planes of skeletal muscles (Fig. 1b, c). Host inflammatory response to the fungal hyphae was minimal and marked myonecrosis (Fig. 1b) was observed in some areas. Fungal elements morphologically similar to *Saprolegnia* were also associated with skin lesions from fish sampled at a similar time in previous years. Dermal lesions were found on 70 individuals in the 2000 to 2002 temporal samples (Lake Crescent: $n = 12$, Lake Sorell: $n = 58$), including 20 males, 49 females and one unsexed fish. The size of infected fish ranged from 56 to 149 mm L_F , and infected males were smaller (mean \pm S.E. = 87 ± 4 mm L_F) than females (105 ± 3 mm L_F). Lesions were found at a range of body locations on individuals which did not differ between sexes, other than 6 females having lesions on tail fins whereas none were found in this area on males. Infections were most common on the abdomen (23% of infection sites), lateral flanks (22%), anal fin and vent (18%), pectoral fin (15%), and pelvic and dorsal fins (both <10%). The extent of lesions varied from 20-60% amongst affected fish and, therefore, in many cases was likely to have caused, or been associated with, mortality in due course (Bruno & Wood, 1999). Overall, affected fish occurred (inclusive) from May to October in samples from Lake Sorell and July to September in Lake Crescent (Fig. 2), but were most abundant in July and August in both populations when >70% of affected fish in each lake were recorded. In addition to the fish in sub-samples which presented with dermatitis, many other affected fish were observed in catches at these times. On several occasions during spawning seasons, small numbers (<10 fish) of deceased individuals (mostly spent fish) with extensive dermatitis lesions were also found in the littoral wash at known spawning sites (rocky shores) in both lakes (Fig. 3). These observations, coupled with the extensive nature of many lesions on live fish, indicate that their occurrence was not associated with skin abrasion during capture, but to other predisposing factors.

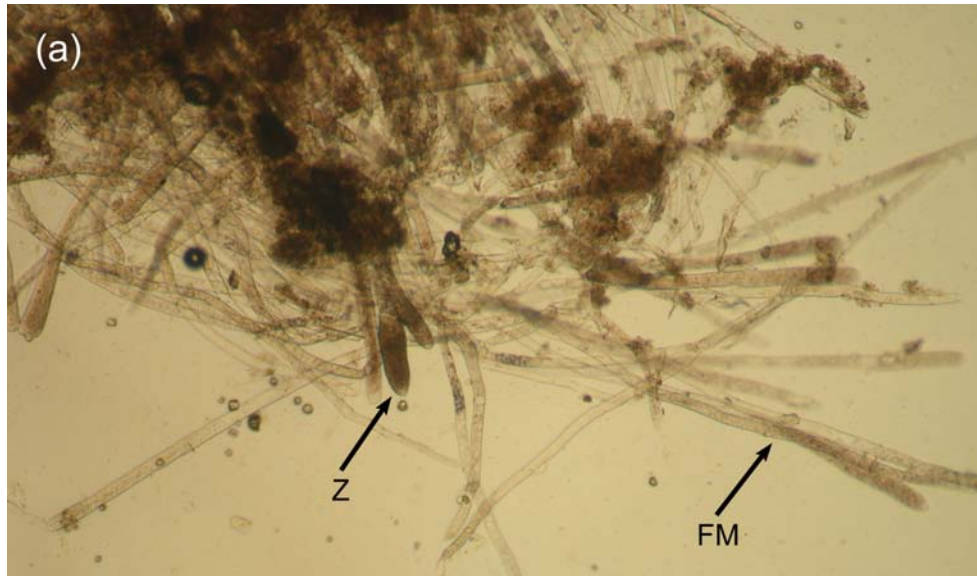


Fig. 1a. Wet preparation of *Saprolegnia* fungal mat and organic debris from the skin of *Galaxias auratus*. Zoosporangia (Z) and non septate fungal mycelium (FM) are indicated ($\times 100$).

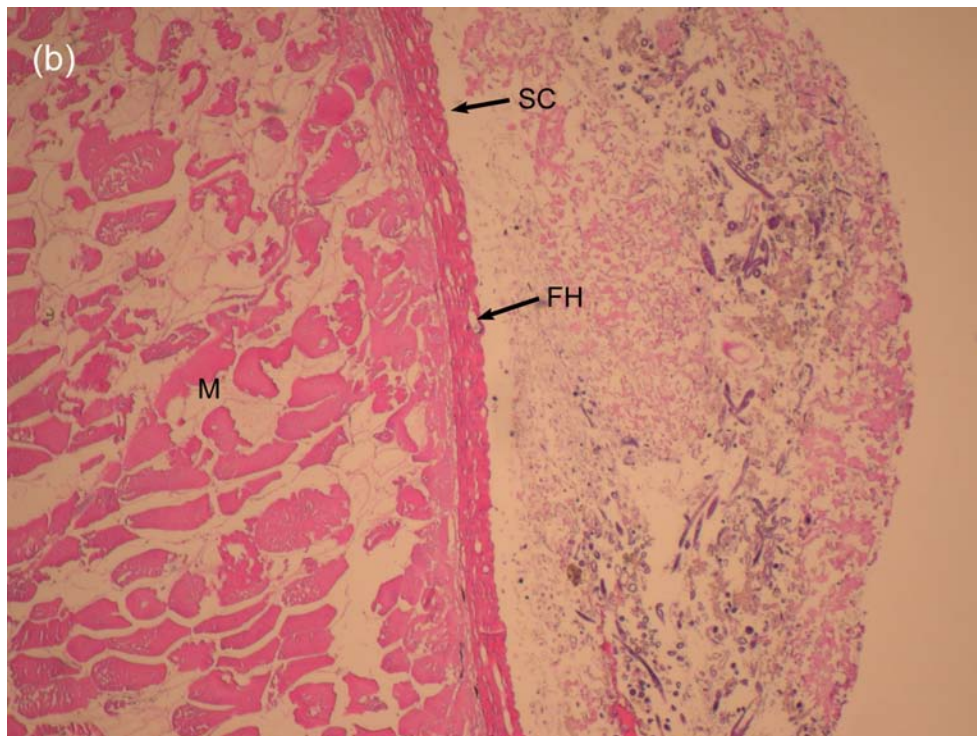


Fig. 1b. Section of skin lesion from *Galaxias auratus* showing epithelial loss with an overlying fungal mat. Fungal hyphae (FH) are penetrating the stratum compactum (SC) of the dermis and marked myonecrosis is present within the underlying skeletal muscle (M). Hematoxylin and eosin ($\times 40$).

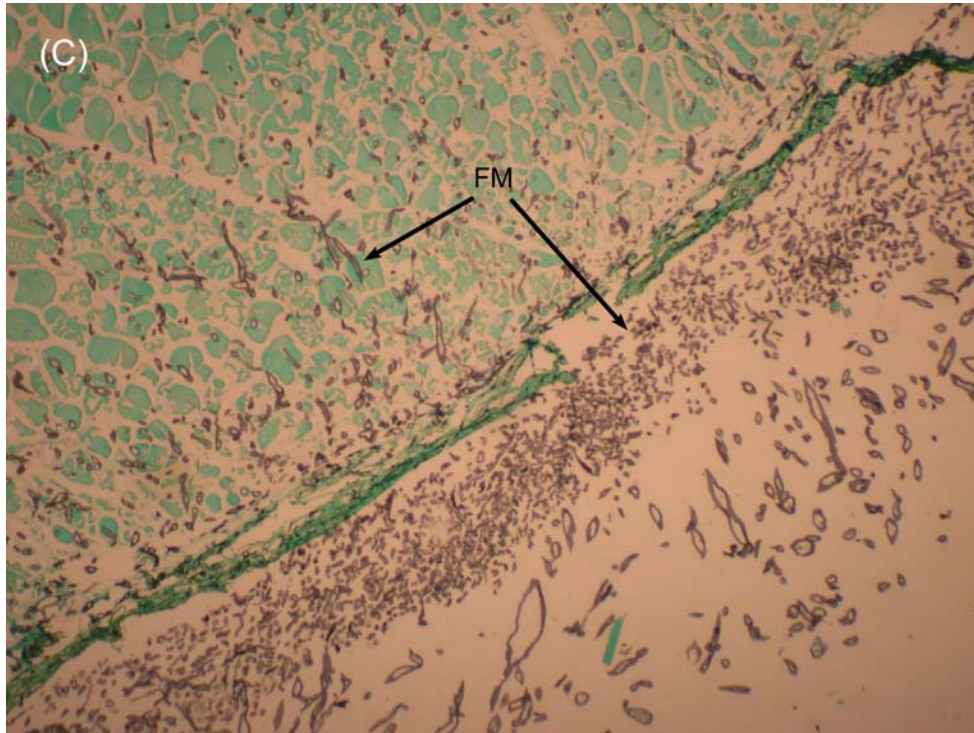


Fig. 1c. Fungal mycelium (FM) penetrating the skin and muscle tissue of *Galaxias auratus*. Hyphae are highlighted (black) with Grocott's Methenamine silver ($\times 40$).

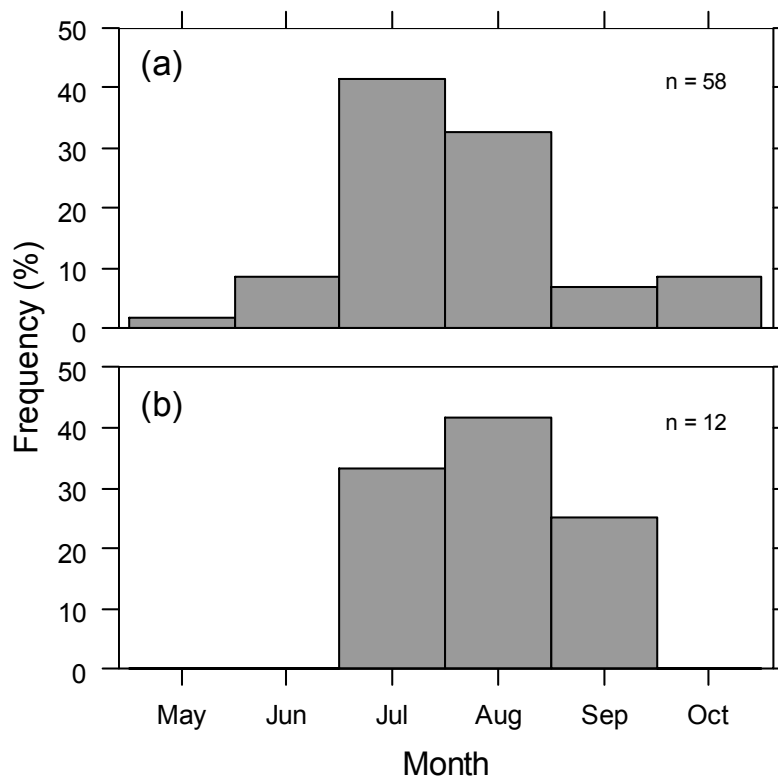


Fig. 2. Frequency of occurrence of *Galaxias auratus* with focal dermatitis in monthly samples from (a) Lake Sorell and (b) Lake Crescent (July 2000 to December 2002).



Fig. 3. Deceased, spent *Galaxias auratus* with extensive dermatitis lesions found in the littoral wash at a spawning site on the western shore of Lake Crescent during August 2005.

GLM analyses showed that both D_S and lake were significantly associated with the occurrence of dermatitis ($D_S \times \text{Dermatitis}$, $P < 0.0001$; $\text{Lake} \times \text{Dermatitis}$, $P < 0.0001$) and the 3-way interaction was the only term not needed in the model ($P = 0.24$, and thus the final model was not over dispersed since the residual deviance was similar to the residual d.f.). The pattern of probabilities of finding fish with dermatitis (Table 1) show that fish were most frequently affected in the later stages of development ($D_S = 4$ to 6), with a peak at $D_S = 5$ (spent fish); affected fish were also more likely to occur in Lake Sorell than Lake Crescent.

Table 1. Fitted probabilities of finding *Galaxias auratus* with focal dermatitis for each gonadal stage of development in lakes Crescent and Sorell. Observed probabilities were identical (to the nearest 0.001) to fitted probabilities.

	D_S^*					
	1	2	3	4	5	6
Lake Crescent	0.010	<0.001	<0.001	0.003	0.027	0.008
Lake Sorell	0.011	0.044	<0.001	0.036	0.121	0.082
<i>n</i>	7	2	0	15	29	17

*Gonad stages of development (D_S) are: 1 = immature resting, 2 = early developing, 3 = late developing, 4 = ripe, 5 = spent, and 6 = mature resting.

7.5. Discussion

Saprolegnia spp. are common pathogenic fungi which infect living and dead freshwater teleosts and their eggs globally (Bruno & Wood, 1999; Hussein & Hatai, 2002), including several species under aquaculture in Australia (Lategan *et al.*, 2004a, b; S. B. Pyecroft, unpubl. obs.). Susceptibility to saprolegniasis is increased by physical stress and damage to the epidermis which may result from aggressive social interactions or overcrowding (Bruno & Wood, 1999). For example, Richards & Pickering (1978) speculated that physical damage to the epidermis associated with spawning behaviour (e.g. redd digging and aggressive social interactions) and changes in the structure of the epidermis (i.e. reduction in mucous cells) of spawning brown trout *Salmo trutta* L. may increase vulnerability to *Saprolegnia* infection. As in many species, salmonids are known to have reduced condition during spawning (Jamet, 1995) which could also contribute to increased rates of infection. In Australia during winter, Saprolegniasis has been found to increase in silver perch *Bidyanus bidyanus* (Mitchell) (Lategan *et al.*, 2004a), short-finned eel *Anguilla australis* Richardson (Lategan *et al.*, 2004b) in aquaculture situations and introduced salmonids in hatcheries and wild populations (S. B.

Pyecroft, unpubl. obs.), suggesting stressors related to cold water temperatures may increase the likelihood of infection. Poor water quality including high organic loadings can also increase the chances of infection (Bruno & Wood, 1999).

It is possible, therefore, that several underlying factors may be associated with the occurrence of fungal dermatitis in *G. auratus* populations and the observed relationship between the chance of infection and the reproductive cycle of this species. At the time of this study, Lake Crescent and Lake Sorell both had low water levels (Hardie *et al.*, in press; Chapter 8); hence, suspended organic loads were constantly very high due to wind-induced suspension of sediment and organic material (Uytendaal, 2006). This could have increased the likelihood of infection, but does not account for its seasonal occurrence. Infection was more prevalent in July and August (austral winter) when water temperatures (monthly means \pm S.D.) are very low in these lakes (July = $4.0 \pm 0.9^\circ\text{C}$, August = $4.8 \pm 1.1^\circ\text{C}$). Whilst the reproductive physiology and spawning behaviour of *G. auratus* is largely unknown, several spawning-related factors are likely to increase the chances of infection. These include overcrowding (i.e. spawning aggregations (Hardie *et al.*, 2005)), physical damage from aggressive social interactions (i.e. fin nipping) or spawning activity (i.e. abrasion on rocks), and physiological changes (i.e. poor condition, which has been found to occur in lacustrine common galaxias *Galaxias maculatus* (Jenyns) in Australia (Pollard, 1972b)). Collectively, these factors account for the observed increased rates of infection in fish that were fully developed and waiting for spawning cues ($D_S = 4$), recently spent ($D_S = 5$), and those that spawned earlier in the season and were recovering and beginning to re-develop gonads ($D_S = 6$) during sub-optimal conditions (i.e. winter).

Mortality at spawning sites has previously been observed for diadromous climbing galaxias *Galaxias brevipinnis* Günther (O'Connor & Koehn, 1998) and non-diadromous *G. maculatus* (Pollard, 1971) in Australia. Both studies recorded live and deceased spent fish with focal dermatitis at spawning sites which Pollard (1971) suggested to be as a result of *Saprolegnia* infection. These lowland populations spawned in April-May and August-October respectively, hence, water temperatures were significantly warmer (*c.* >9°C) than those in lakes Crescent and Sorell during the time *G. auratus* presenting fungal dermatitis were recorded. This suggests that spawning activity rather than low temperature is the main cause of fungal dermatitis in galaxiids; however, further research using controlled treatments would be required to evaluate this hypothesis.

It is uncertain whether there is sexual bias in the susceptibility of *G. auratus* to infection. Whilst there were more than double the number of affected females than males (49 to 20), the overall male to female ratio (including immature developing and mature fish) in these populations is also strongly female-biased (*c.* 1:2; Hardie *et al.*, in press). However, male *G. auratus* have a shorter life-span than females (Chapter 5) so males maybe more susceptible. As *Saprolegnia* infections proceed, lethargy and loss of equilibrium follow (Bruno & Wood, 1999); therefore, mortality of *G. auratus* with saprolegniasis is likely to be further exacerbated by an increased risk of piscine (e.g. introduced salmonids) and avian predation.

Saprolegnia sp. are also known to infect eggs of Canterbury galaxias *Galaxias vulgaris* Stokell (Cadwallader, 1976) and *G. maculatus* (Benzie, 1968) in aquaria. A fungus with similar morphological appearance to that of the *Saprolegnia* sp. recorded on adult *G. auratus* was also found on dead eggs at spawning sites during the study. This was particularly

prevalent at sites in Lake Crescent that had been smothered by sediment during strong wind events (S. A. Hardie, unpubl. obs.). Therefore, *Saprolegnia* infections may also be associated with the mortality of *G. auratus* eggs.

Strategies to manage water levels which alleviate the prolonged occurrence of low levels in lakes Crescent and Sorell, and hence reduce organic loads, may lessen the chances of *Saprolegnia* infection in *G. auratus*. However, the results of this study indicate that spawning-related stressors increase the rate of infection. It is probable that saprolegniasis is associated with seasonal mortality in *G. auratus* populations in lakes Crescent and Sorell; however, further work is required to determine mechanisms which influence fungal colonisation and the mortality rates of infected fish.

8. Recruitment dynamics of a non-diadromous lacustrine galaxiid fish: the roles of water level fluctuations and habitat availability*

*To be submitted to *Freshwater Biology* as an *Applied Issues* paper following minor adjustments.

8.1. Abstract

1. This study examined the early stages of recruitment in golden galaxias (*Galaxias auratus*) populations in Lake Crescent and Lake Sorell, central Tasmania, Australia over five years. These highland lakes have similar hydrology, and physical and chemical attributes, but in contrast to Lake Sorell, Lake Crescent has limited areas of spawning habitat (rocky littoral substrate) for *G. auratus* and access to these habitats is water level dependent. This enabled the roles of water level fluctuations and habitat availability in recruitment to be investigated.
2. Periods of larval emergence varied annually within and between the populations of the two lakes, despite consistent and highly seasonal water temperatures. Monthly patterns in larval emergence and abundance were associated with the timing of inundation of spawning habitats during late autumn – spring.
3. In Lake Crescent, seasonal abundances of larvae were strongly related to the magnitude of water level rises during spawning and egg incubation (i.e. May-September), whereas there was no such relationship in Lake Sorell.
4. Despite the occurrence of larvae in pelagic habitats during winter, they did not begin to grow until spring when water temperatures rose above ~10°C. This suggests coupling of water level and water temperature regimes is important for the survival and growth of larvae. Low water levels also reduced the availability of littoral nursery habitats for juveniles.

5. Water level fluctuations play a key role in the recruitment of *G. auratus* which relies on access to complex littoral habitats for spawning. Because many lacustrine galaxiids use littoral habitats during their early life stages, alterations to water levels and seasonal hydrological regimes may impact on their recruitment by restricting access to these habitats at critical times.

Key words: Galaxiidae; lakes; fish larvae; hydrology; littoral habitats

8.2. Introduction

Hydrological variables, such as water level fluctuations, are crucial to the life history of many lacustrine fishes, especially those that use the littoral during their life cycle. This habitat is strongly affected by hydrological variability (Gasith & Gafny, 1990; Winfield, 2004a).

Similar to the more widely studied influence of flows on riverine fishes (Freeman *et al.*, 2001; Humphries *et al.*, 2002; Lobón-Cerviá & Rincón, 2004), water level fluctuations may also affect fish in lentic waters by controlling spawning success (Gafny *et al.*, 1992; Sammons *et al.*, 1999) and recruitment (Sammons & Bettoli, 2000; Havens *et al.*, 2005). Anthropogenic manipulation of water levels in impounded natural lentic systems can alter the timing, magnitude, duration and periodicity of fluctuations. For these reasons, knowledge of relationships between water level regimes and the recruitment dynamics of lacustrine fishes is important to the management of fish assemblages in waters with altered hydrology.

Many lacustrine fishes use structurally complex habitats (e.g. aquatic vegetation) for spawning, incubation of eggs, and nursery environments for young-of-the-year (YOY) fish (Chambers & Trippel, 1997a; Wootton, 1998). However, minor changes in water levels in shallow lakes can alter habitat quality. For example, the abundance and species richness of

aquatic vegetation (Boulton & Brock, 1999) and stability of conditions over littoral substrates (Imboden, 2004) can be altered by hydrological manipulation. Furthermore, abnormally low levels may prevent access to littoral zones and increase sedimentation rates (Scheffer, 2004) and, thus, reduce the suitability of remaining habitats for spawning (Bruton, 1985). Sudden changes in hydrology can also alter interaction in food webs and the trophic status of lakes (Scheffer *et al.*, 1993) causing YOY to encounter adverse conditions for growth and survival.

Several abiotic (e.g. water temperature) and biotic (e.g. competitive interactions) factors affect year-class strength (YCS) (Helminen *et al.*, 1997; Ludsin & DeVries, 1997; Fulford *et al.*, 2006), and the timing of recruitment events can be more dependent on endogenous rhythms than ambient conditions (Munro *et al.*, 1990; Jobling, 1995). However, the search for environmental predictors of recruitment strength has received much attention from researchers (Myers, 1998). The life history and demographic parameters of a species may determine whether its abundances are likely to be related to environmental variables (Chambers & Trippel, 1997b). Indeed, species occurring in discrete geographical regions that do not have flexible reproductive strategies are more likely to exhibit environment-recruitment correlations due to their specialised use of habitats and reliance on specific conditions.

Despite being complicated by factors such as density-dependent processes (Cowan *et al.*, 2000) and the availability of food resources (Cushing, 1990), high larval abundance may produce high numbers of recruits (Mooij, 1996; Karjalainen *et al.*, 2000). Furthermore, relatively small fluctuations in the survival of larvae can result in large inter-annual variations in YCS. The dynamics of YCS has often been studied in species that are exploited commercially (Chambers & Trippel, 1997a); however, it is also important to the conservation

of threatened species as it may provide a early measure of population viability. Quantifiable relationships between hydrological variables and recruitment for several lacustrine species are known to occur (Miranda *et al.*, 1984; Maceina & Stimpert, 1998). However, whilst mechanisms causing these relationships, such as the availability of suitable spawning (Ozen & Noble, 2005) or nursery (Sammons & Bettoli, 2000; Havens *et al.*, 2005) habitats, are sometimes suggested, they are seldom robustly tested in natural waters. More subtle factors including physiological cues provided by hydrological variables may also be important for reproduction (Lam, 1983), but are difficult to investigate. Clearly, if fish rely upon cues or conditions associated with hydrological cycles, then alterations to water level regimes may affect recruitment and control YCS, but the governing mechanism(s) need to be identified if these relationships are to be incorporated into lake-specific management strategies.

The Galaxiidae are relatively small-sized (adults usually <300 mm long) fishes which occur in a diverse range of freshwater habitats in the Southern Hemisphere (McDowall & Fulton, 1996). The family is particularly prevalent in inland waters in southern Australia and New Zealand (McDowall & Frankenberg, 1981), where several non-diadromous forms occur, many in discrete lentic systems. Lacustrine galaxiids typically spawn in the austral spring – early summer, their larvae are free-swimming and pelagic, and juveniles recruit into adult epibenthic habitats before the following winter (see McDowall & Fulton, 1996; McDowall, 2000). Galaxiid species dominate the native freshwater fish fauna of the Tasmania Central Plateau, southern Australia, owing to the occurrence of seven endemic lacustrine species, all of which have restricted distributions and are considered to be threatened (Hardie *et al.*, 2006b). The hydrological regimes of most of the larger waters in this region, including those occupied by six endemic galaxiid species, have been altered by schemes to supply water for hydroelectric power generation, and agricultural and municipal purposes. These

manipulations potentially threaten the region's galaxiid fauna (Hardie *et al.*, 2006b), and, currently, local water level management strategies do not consider the needs of indigenous fishes.

Whilst aspects of the ontogeny of some lacustrine galaxiids have been examined (Cussac *et al.*, 1992; Rowe & Chisnall, 1996a), recruitment processes have not previously been studied over more than two seasons. For the purposes of this chapter, the term 'recruitment' refers to the abundance of the earliest age at which the YOY cohort can be determined (Myers, 1998); for this species it is the pelagic larvae-juvenile stage. In this study, we compare the inter-annual recruitment dynamics (timing of emergence, abundance of larvae, growth of YOY and timing of habitat shifts) of golden galaxias (*Galaxias auratus* Johnston) populations in Lake Crescent and Lake Sorell, central Tasmania over a five-year period. *Galaxias auratus* is endemic to these interconnected highland lakes and has a non-diadromous life history with spawning occurring during late autumn – winter in littoral habitats (Hardie *et al.*, 2004). Our objective was to define the ecology of the early life stages of *G. auratus* and investigate the roles of habitat availability and water level fluctuations in recruitment. Our hypotheses were that: (1) larval abundance will be greater during years of higher water levels in Lake Crescent due to increased availability of spawning habitat, and (2) the timing and magnitude of seasonal rises in water levels may also influence the occurrence and abundance of larvae. Based on our findings, we assess threats to other lacustrine galaxiids and discuss management issues.

8.3. Materials and methods

Study site and hydrology

The interconnected Lake Crescent (42°10'S, 147°10'E) and Lake Sorell (42°6'S, 147°10'E) lie 1 km apart and are situated at ~800 m a.s.l. in the south-east of the Tasmanian Central Plateau, Australia (Fig. 1). Both lakes are impounded natural waters that have had their levels raised on several occasions since the 1830s for water storage purposes. Lakes Crescent and Sorell are quite large (surface areas of 23 and 52 km², respectively), relatively shallow (mean depths <3.5 m at full supply levels), and turbid (mean of ~100 NTU in both lakes during this study; Uytendaal, 2003). The lakes are very similar physically and chemically and are located in an area of uniform geology, climate, soils and vegetation (Cheng & Tyler, 1973).

Physicochemical properties of the water columns of both lakes are typically well homogenised due to their shallowness and prevailing wind action. The trophic status of these lakes differs, with Lake Sorell being mesotrophic and Lake Crescent moderately eutrophic (Cheng & Tyler, 1976a), and Lake Crescent has a phytoplankton standing crop biomass 10 times that of Lake Sorell (Cheng & Tyler, 1976b). The fish assemblages of both lakes include two native species, the endemic *G. auratus* and indigenous short-finned eel (*Anguilla australis* Richardson), and three exotic species, brown trout (*Salmo trutta* L.), rainbow trout (*Oncorhynchus mykiss* Walbaum) and common carp (*Cyprinus carpio* L.). *Galaxias auratus* are ~8 times more abundant in Lake Crescent than in Lake Sorell possibly due to differences in salmonid predation rates and lake productivity (Hardie *et al.*, 2005).

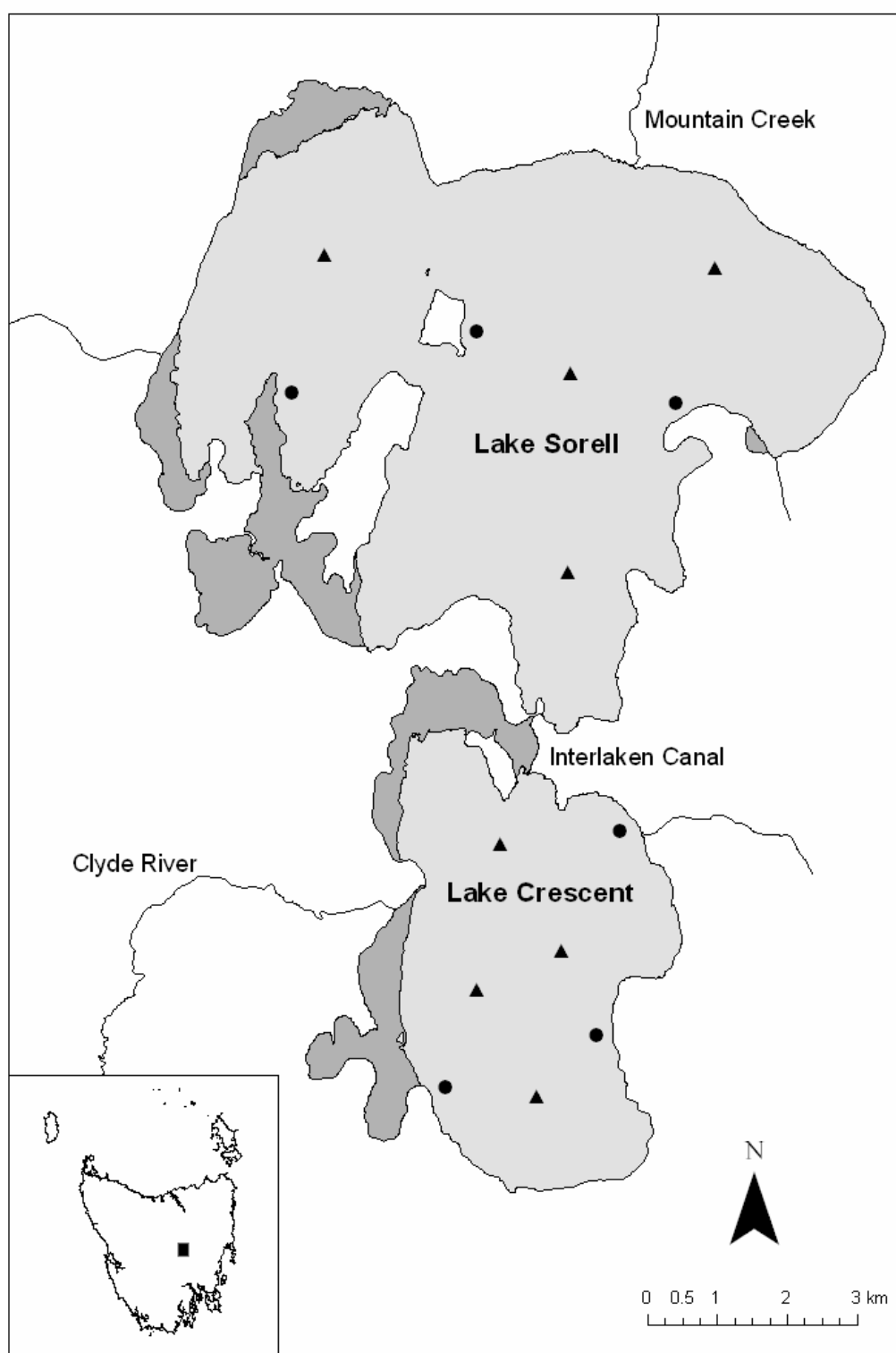


Fig. 1. Location of littoral (●) and pelagic (▲) sampling sites for *Galaxias auratus* in lakes Crescent and Sorell, Tasmania, Australia. Lake perimeters are at full supply levels. Wetland areas are depicted by dark shading.

The Crescent-Sorell system has a relatively small catchment area (206 km²) and six ephemeral tributaries (Fig. 1.). The largest tributary, Mountain Creek (mean annual discharge = 11 767 ML; Uytendaal, 2003), flows in at the northern end of Lake Sorell, whilst the single outflow, the Clyde River, flows out of Lake Crescent to the south-west. The hydrology of this system is primarily controlled by climatic variables including relatively low rainfall (annual mean = 699 mm), and high evaporation rates during summer (mean 4.5 mm day⁻¹), whilst water releases for downstream users (annual mean = 10 000 ML) also alter water levels (DPIWE, 2004). Historically, water level fluctuations have occurred seasonally (minimums occur March-April and maximums in October-November) and typically range from *c.* 0.2 to 0.9 m in both lakes (Fig. 2). Since 1970, both lakes have maintained relatively high levels, except during *El Niño*-induced droughts in 1982-1984 and 1997-1999 (Fig. 2).

The availability of the spawning habitats (e.g. littoral areas of rocky substrate or aquatic vegetation) for *G. auratus* differs substantially between the lakes (Hardie *et al.*, in press). In Lake Crescent, there are four defined rocky shorelines and at all locations this habitat is inundated at water levels >802.2 m Australian Height Datum (AHD). Conversely, Lake Sorell contains vast littoral areas of rocky substrate which extend well into the lake's basin in many regions. Extensive adjacent wetland areas (Fig. 1), accounting for 17% of the surface area of Lake Crescent and 8% of Lake Sorell at full supply levels (Heffer, 2003), connect to the main bodies of these lakes at high water levels (Lake Crescent ≥802.6 m AHD and Lake Sorell ≥803.7 m AHD; Fig. 2).

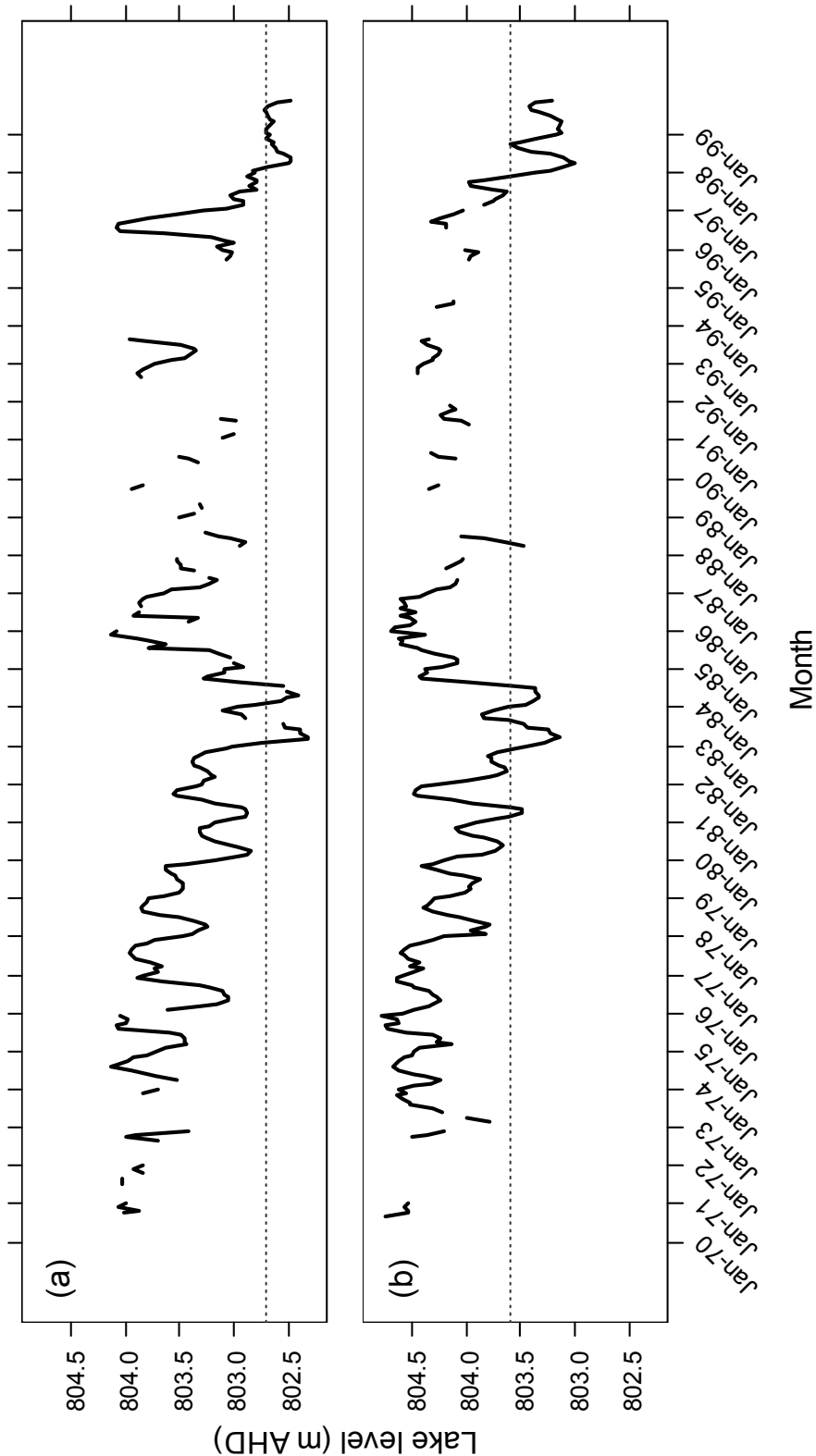


Fig. 2. Water levels in (a) Lake Crescent and (b) Lake Sorell between 1970 and 1999. Water levels are measured in metres of elevation according to the Australian Height Datum (m AHD). Levels at which littoral wetlands connect to the lakes are shown by horizontal dashed lines. Water level data were provided by the Inland Fisheries Service, Tasmania.

In the past, both lakes have also contained in-lake macrophytes, but no significant stands were present during this study owing to low water levels and consequent increased exposure of bed sediments to wave action and elevated turbidity (Uytendaal, 2006). Therefore, in-lake benthic areas consisted primarily of fine sediment in Lake Crescent and a mixture of fine sediment and rock in Lake Sorell during this study.

Water temperature

Mean daily water temperature was measured in lakes Crescent and Sorell between January 2000 and December 2002 (depth 0.5 m) using Optic StowAway Temperature loggers (Model WTA). Mean daily water temperature in both lakes during this period was also modelled using Dynamic Reservoir Simulation Model (DYRESM) (Antenucci, 2001) with meteorological data supplied by Hydro Tasmania, Bureau of Meteorology and the Tasmanian Inland Fisheries Service (all Hobart, Australia). Modelled data closely approximated observed temperatures in both lakes during this period ($R^2 \geq 0.95$) and the resultant models were used to estimate mean daily water temperatures in each lake between January 2002 and December 2004 with meteorological data for this period. The difference between the water temperature regimes of the lakes was examined using a paired *t*-test of mean monthly temperatures.

Pelagic fish sampling

Pelagic YOY *G. auratus* were sampled monthly from Lake Crescent and Lake Sorell between July 2000 and December 2003 to examine the duration of emergence periods and times at which peaks in hatching occur. YOY were also sampled in February 2004 to aid growth modelling of the 2003 cohort and between August and December 2004 to gain a measure of the peak abundance for that year. Fish were collected at four spatially distributed sites in each lake (Fig. 1) by towing a conical ichthyoplankton net (400 mm diameter, 1.25 m tail with

500 μm mesh) for 10 min approximately 15 m behind a boat at a speed ($\sim 2 \text{ m s}^{-1}$) that ensured the net was sampling the top 1 m of the water column. For the purposes of this paper, fish captured in pelagic habitats during the calendar year of their birth (i.e. $\leq 40 \text{ mm}$ fork length, FL) are referred to as ‘larvae’ and all other YOY (typically 41–70 mm FL) as ‘juveniles’ (at $\sim 40 \text{ mm}$ FL, the majority of first year *G. auratus* begin to develop adult pigmentation (Frijlink, 1999) and undergo an ontogenetic shift from pelagic to littoral habitats (see Results), both of which are decisive events in the early development of teleosts (Balon, 1984)). All material collected in ichthyoplankton samples was euthanased in an anaesthetic solution (Aqui-s™) and preserved in 70% ethanol. *Galaxias auratus* larvae were identified following unpublished keys (Frijlink, 1999), the number of larvae and juveniles in each sample were counted (fish tow^{-1}), and the first 100 larvae and all juveniles in each sample were measured (mm FL).

Littoral fish sampling

Juvenile and adult *G. auratus* were sampled monthly from littoral habitats (depths $\leq 1.2 \text{ m}$) in Lake Crescent and Lake Sorell between October 2000 and December 2002. Fish were collected at three sites in each lake (Fig. 1) which represented one of the dominant littoral habitats (fine sediment, sand and rock) in both lakes at the time of the study. Fish were sampled overnight (mean soak time of 18 hours) using four fyke nets (two $3 \text{ m} \times 0.6 \text{ m}$ wing and two $5 \text{ m} \times 0.6 \text{ m}$ wing nets with 600 mm D-shaped entrance and 2 mm stretched mesh) at each site (i.e. 12 nets per lake). All nets had an $84 \times 70 \text{ mm}$ aluminium screen in the entrance to avoid the capture of platypus (*Ornithorhynchus anatinus*), water birds and larger fish species (e.g. salmonids). Catches from littoral habitats were anaesthetised in an anaesthetic solution (Aqui-s™) and a random sub-sample of 50 fish from each site were measured (mm FL). Fish were then revived in fresh water and released at the site of capture.

Data analysis

YOY *G. auratus* from pelagic habitats were assigned to cohorts based on the year of their birth, which was clearly defined by length-classes in temporal catches. The timing of emergence was analysed by examining the variability in the occurrence of newly hatched larvae (≤ 8 mm FL) in pelagic samples using Kolmogorov-Smirnov two-sample tests with pooled monthly length data for each lake. Data for 2004 were excluded from these analyses because sampling did not begin until August (i.e. the initial occurrence of larvae in each lake was probably missed in this year). Larval catch data for sites sampled between August and December of each year were $\log_{10}(x + 1)$ transformed to homogenise variances and normalise residuals, and repeated-measures ANOVAs were used to test for differences in mean abundances between lakes and years. In these analyses, lakes were treated as between subject factors, years as the within-subject factor, and sample sites were the subjects. Regression analysis was used to assess relations between lake hydrology (i.e. magnitude of rise between May and September when *G. auratus* spawn and egg incubation occurs) and mean abundance of larvae between August and December in both lakes across years.

Growth of YOY in the 2000-2003 cohorts in each population, where fish were captured in tow net samples beyond their first December, was examined by fitting logistic growth models to mean monthly lengths for each cohort. The logistic growth equation is:

$$L_t = L_{\infty} \left[1 + e^{\left(-K[t+t_0] \right)} \right]^{-1}$$

where L_t is the length at age t (days), L_{∞} is the asymptotic length, K is the age-specific growth rate, and t_0 is a shift parameter to allow the extrapolated length at age 0 to be nonzero (Kaufmann, 1981; McCallum, 2000). Growth models for each cohort and lake were compared using likelihood-ratio tests (Kimura, 1980). The periods when growth began to occur in cohorts in each lake were also examined by one-way ANOVAs with year and month as fixed

factors. Fine-meshed fyke nets captured a large size-range of *G. auratus* in littoral areas (31-235 mm FL); therefore, the monthly sampling regime between October 2000 and December 2002 enabled the timing of the pelagic-littoral habitat shift of YOY fish to be investigated. The timing and duration of the habitat shifts in each lake were assessed by length-frequency histograms of monthly data from pelagic and littoral habitats during this period. All analyses were carried out in R version 2.2.0 (R Development Core Team, 2005), and growth modelling included functions in the package nlme (linear and nonlinear mixed effects models) (Pinheiro *et al.*, 2005). The significance level for hypothesis tests was $P = 0.05$.

8.4. Results

Hydrology, habitat availability and water temperature

In comparison to historical water levels (1970-1999; Fig. 2), both Lake Crescent and Lake Sorell experienced very low levels during this five-year study, including record minima in 2000 in Sorell (802.620 m AHD) and in 2001 in Crescent (801.770 m AHD) (Fig. 3). This was largely due to abnormal climatic conditions just prior to and early in the study with an extended *El Niño*-induced drought between 1997 and 2001 causing below average and unseasonal rainfall in the catchment. During the study, annual base levels in lakes Crescent and Sorell rose progressively and water level fluctuations followed similar trends in both lakes (Fig. 3) which were comparable to historical seasonal patterns (Fig. 2). There was considerable inter-annual variation in fluctuations with seasonal rises ranging from 0.200 to 1.050 m and falls from 0.280 to 0.650 m. Minimum and maximum levels generally occurred during March-May and October-December, respectively; however, the first substantial seasonal rise (0.1 m) occurred as early as 30 January in 2004 in Lake Sorell, and as late as 18 August in Lake Crescent in 2002.

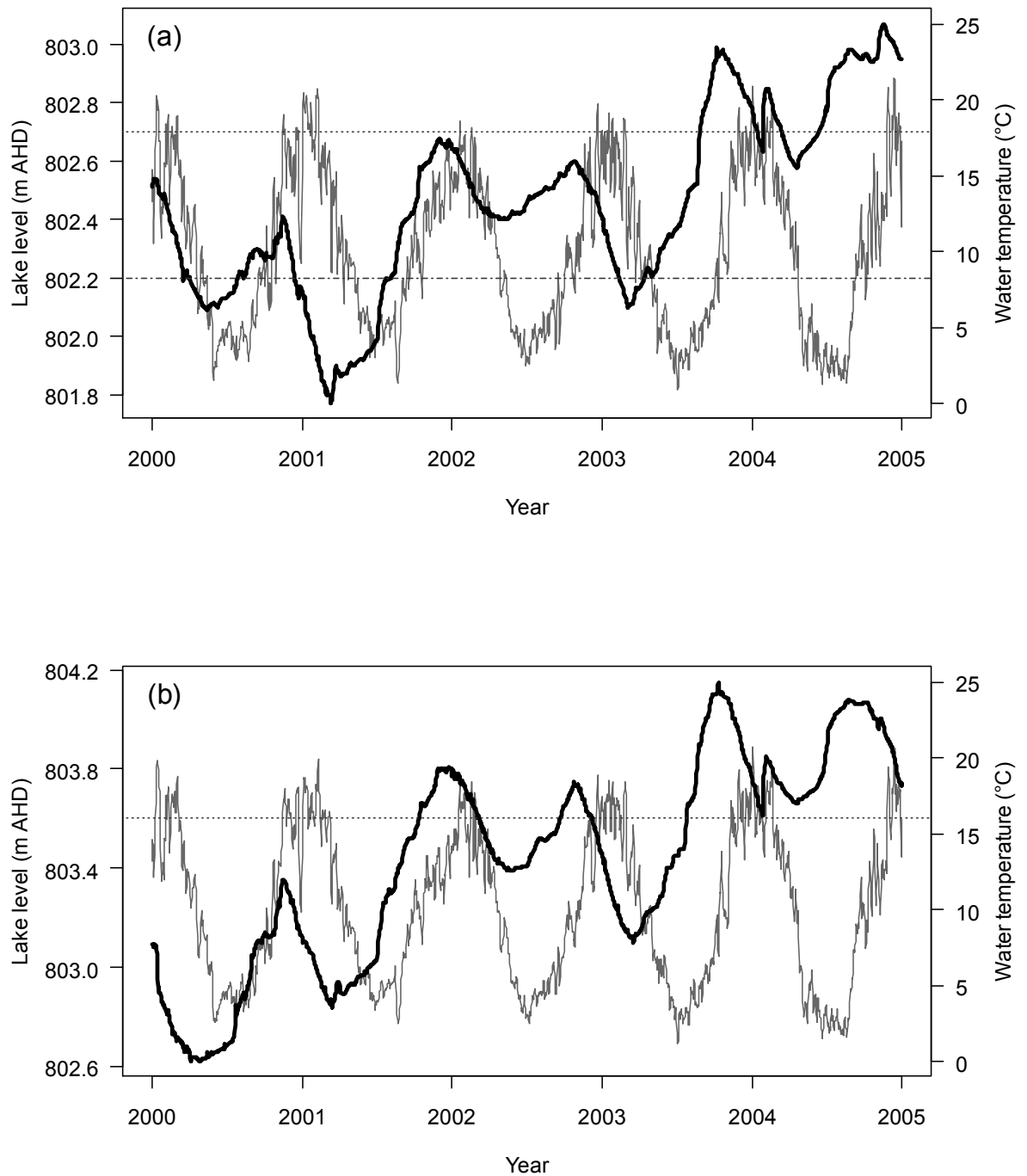


Fig. 3. Daily water levels and mean daily water temperatures in (a) Lake Crescent and (b) Lake Sorell between 2000 and 2005. Water levels are measured in metres of elevation according to the Australian Height Datum (m AHD). Levels at which littoral wetlands connect to the lakes and the level at which rocky shores begin to be inundated in Lake Crescent are shown by dashed, and dashed and dotted horizontal lines respectively.

Whilst fluctuations in both lakes were similar, levels in Lake Sorell (Fig. 3b) usually began to rise slightly earlier than those in Lake Crescent (Fig. 3a) in response to late autumn – winter rainfall due to direct seasonal input from the primary tributary, Mountain Creek. Fluctuations in Lake Sorell (Fig. 3b) (both rises and falls) were also typically more rapid because of this input, and regulation of flows between the lakes and discharge from Lake Crescent.

The historically low water levels that persisted throughout this study periodically altered access to littoral wetlands and rocky spawning habitat in Lake Crescent (Fig. 3). The wetlands associated with both lakes were not inundated sufficiently to allow *G. auratus* access (especially during spawning) despite marginal flooding (≤ 0.55 m) over threshold levels late in 2003 and 2004 in Lake Crescent, and late in 2001, 2002, 2003 and throughout 2004 in Lake Sorell (Fig. 3). Rocky shores were completely dewatered in Lake Crescent on three occasions; briefly (~61 days) in 2003, and for extended periods in 2000 (~121 days) and 2001 (~235 days) (Fig. 3a). Whilst the 2003 dewatering occurred prior to the spawning period of *G. auratus*, the availability of this habitat for spawning was limited by the low levels in 2000 and 2001 with inundation being delayed until late July – early August in each season respectively. Therefore, during this study, littoral habitats in Lake Crescent consisted mostly of fine sediment and sand with only limited areas of rock, whereas in Lake Sorell extensive areas of all three substrates were present, with rock dominating the littoral (>60% of shoreline).

Overall, water temperatures in lakes Crescent and Sorell were similar between 2000 and 2004 (Fig. 3) with no significant difference between monthly means (paired *t*-test for all months for 2000-2004: $t = 1.513$, d.f. = 59, $P = 0.136$). However, temperatures varied marginally (mean differences in the average monthly temperatures $< 0.5^{\circ}\text{C}$) between the lakes during 2001

(paired *t*-test for all months for 2001: $t = 2.481$, d.f. = 11, $P < 0.05$) and 2003 (paired *t*-test for all months for 2003: $t = 2.874$, d.f. = 11, $P < 0.05$), probably due to increased disparity in depths of each lake at those times (see Fig. 3). Temperature cycles in pooled lake data had strong seasonal variation with mean (\pm S.D.) monthly temperatures being $< 5^{\circ}\text{C}$ during winter (minimum of $3.6 \pm 1.1^{\circ}\text{C}$ during July) and $> 15^{\circ}\text{C}$ during summer (maximum of $16.5^{\circ}\text{C} \pm 2.1^{\circ}\text{C}$ during January).

Timing and duration of larval emergence

Galaxias auratus larvae were collected in pelagic habitats in Lake Crescent and Lake Sorell during all years of the study; however, there was considerable inter-annual variation in the occurrence of newly hatched larvae (< 8 mm FL) both within (Kolmogorov-Smirnov two-sample tests for Lake Crescent all years: $P < 0.001$, except 2002 vs 2003 $P = 0.4989$; for Lake Sorell all years: $P < 0.0001$) and between (Kolmogorov-Smirnov two-sample tests for years across lakes: $P < 0.0001$) the lakes (Fig. 4). In both lakes, during monthly sampling between July 2000 and December 2003 (with the exception of August 2000 in Lake Sorell), newly hatched larvae occurred as early as May and as late as December. Across seasons, the periods over which newly hatched larvae were collected were similar in both populations with new recruits appearing progressively earlier between 2000 and 2003. Peaks in the occurrence of newly hatched individuals tended to be earlier and less defined in Lake Sorell than Lake Crescent. In Lake Sorell $> 68\%$ of newly hatched larvae were captured during September in 2000 and 2004, whereas between 2001 and 2003 hatching did not appear to peak as dramatically and $> 50\%$ of newly hatched larvae occurred in July-September. Conversely, in Lake Crescent in four of the five years, $> 82\%$ of the newly hatched larvae were captured during September-October.

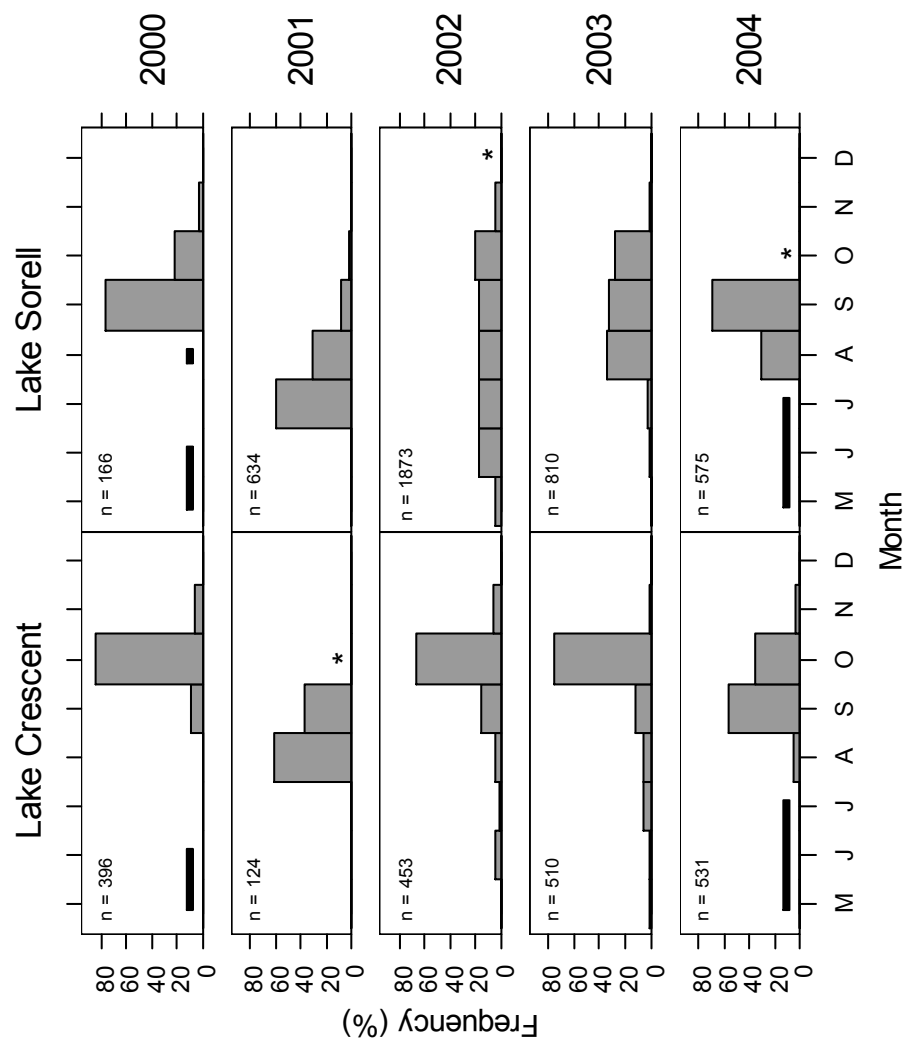


Fig. 4. Frequency of occurrence of newly hatched *Galaxias auratus* larvae (<8 mm FL) in pelagic samples for each spawning season (May-December) in lakes Crescent and Sorell between 2000 and 2004. Months not sampled (horizontal bars) and proportions <1% (*) are indicated.

Abundance of larvae

During emergence periods (May-December), the abundance of larvae in Lake Crescent generally increased from winter to spring with peaks in September or October (Fig. 5). Conversely, in Lake Sorell, larvae usually remained abundant from winter onwards, although slight peaks did occur during late winter – spring in 2002-2004 (Fig. 6). Maximum monthly mean (\pm S.E.) abundances of larvae occurred in November 2003 in Lake Crescent (1754 ± 564 fish tow⁻¹) and September 2004 in Lake Sorell (1364 ± 90 fish tow⁻¹). Many juveniles were also captured in pelagic habitats in Lake Crescent between May and December in each year (Fig. 5), although there were noticeably fewer fish during 2003 (mean for period = 0.28 fish tow⁻¹) than in other years (range of means = 3.6 - 8.4 fish tow⁻¹). In contrast, only small numbers of juveniles (≤ 2 fish per month) were sporadically collected in pelagic samples in Lake Sorell (Fig. 6).

Maximum mean abundances of larvae during August-December occurred in 2001 in Lake Crescent (92.1 ± 0.3 fish tow⁻¹) and 2004 in Lake Sorell (276.4 ± 0.3 fish tow⁻¹). There was a highly significant lake \times year interaction (ANOVA, $F_{4,24} = 20.319$, $P < 0.0001$) on mean larval abundance, indicating that larval abundances in the two populations were influenced by different temporal factors. This inter-annual variation in larval abundances had a strong relationship with lake hydrology in Lake Crescent; however, no such association was observed in Lake Sorell (Fig. 7). In Lake Crescent (Fig. 7b), the magnitude of the rise in water level (WL) between May and September, when spawning is known to occur, was strongly and positively related to the mean abundance of larvae (AL) in each spawning season (August-December) ($\log_{10} AL = 0.981 + 1.905 WL$, $R^2 = 0.97$). In Lake Sorell (Fig. 7a), during all years, except 2001, mean abundances of larvae were similar even though annual water level increases ranged from 0.145 to 0.695 m.

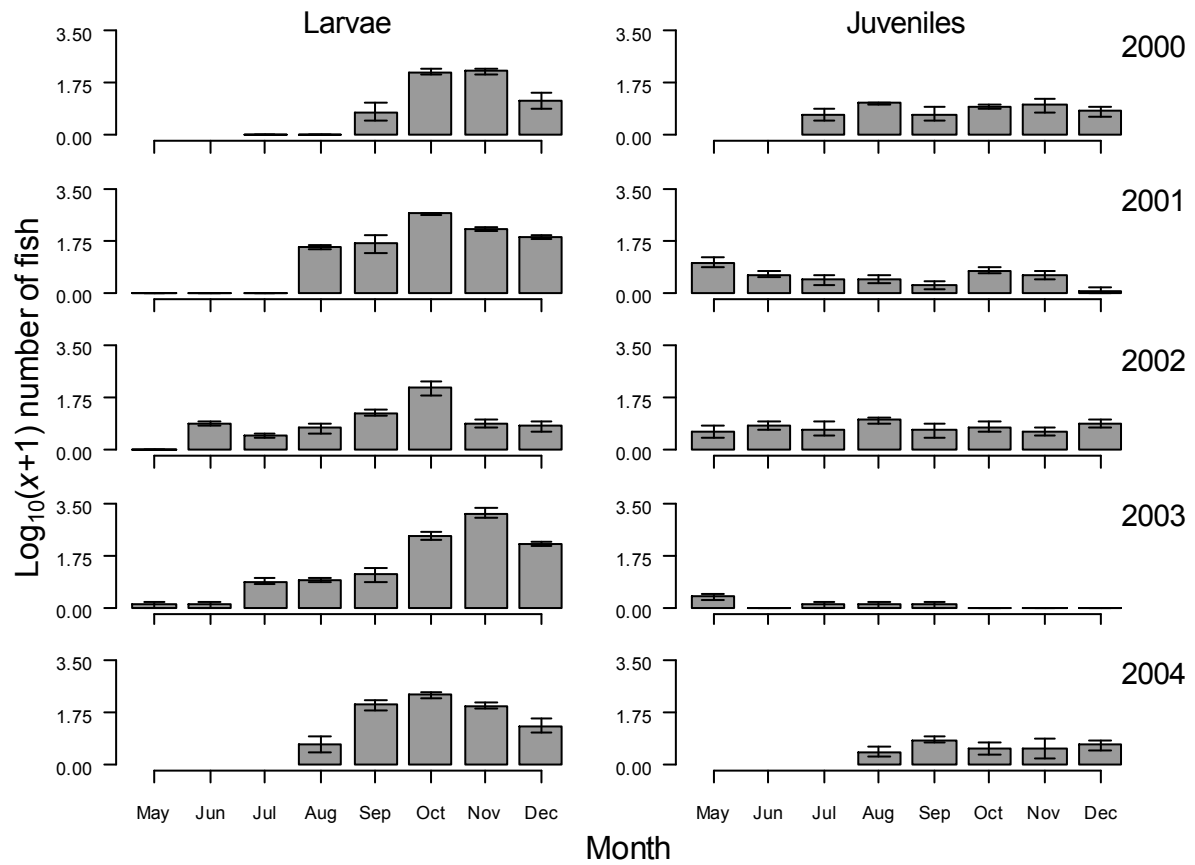


Fig. 5. Mean (±1 S.E.) catch of *Galaxias auratus* larvae (≤40 mm FL) and juveniles (>40 mm FL) in pelagic samples for each spawning season (May-December) in Lake Crescent between 2000 and 2004.

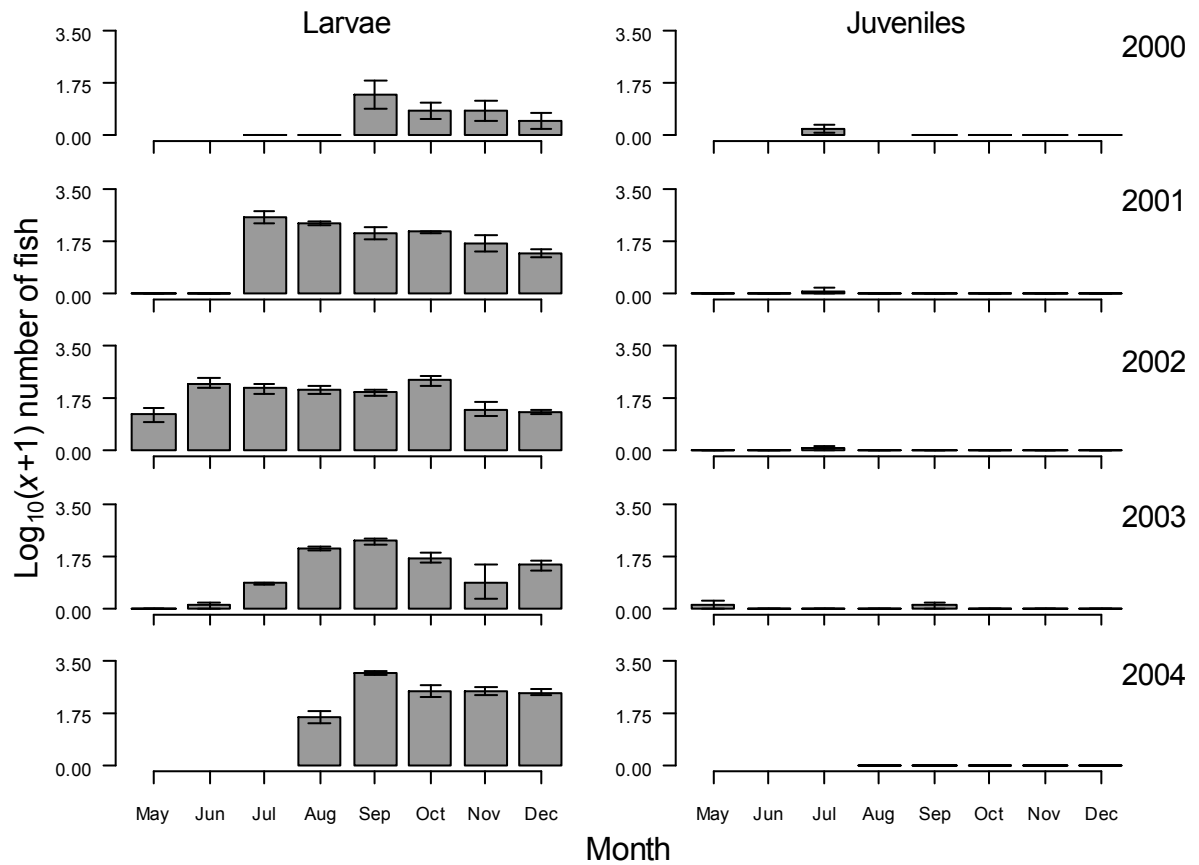


Fig. 6. Mean (± 1 S.E.) catch of *Galaxias auratus* larvae (≤ 40 mm FL) and juveniles (> 40 mm FL) in pelagic samples for each spawning season (May-December) in Lake Sorell between 2000 and 2004.

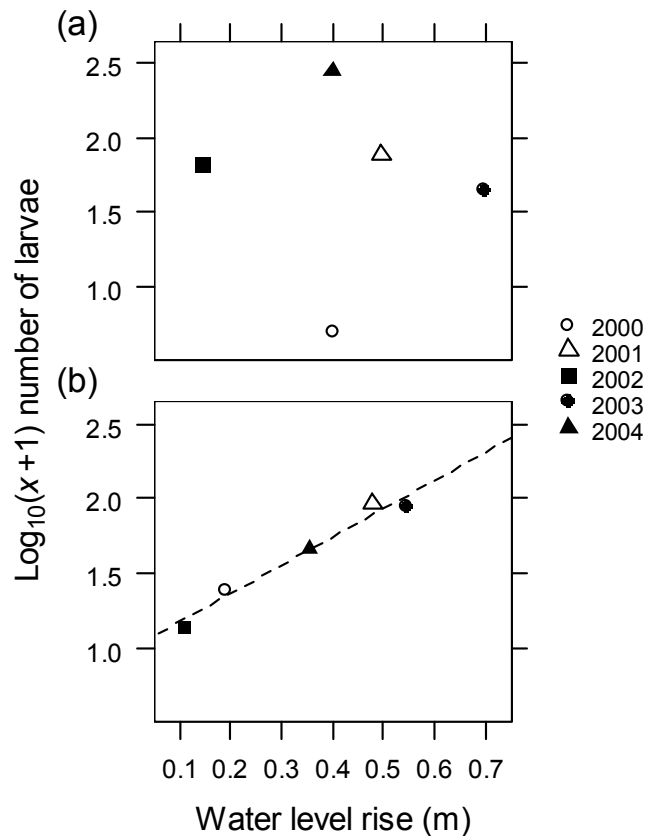


Fig. 7. Rise in water level between May and September and the mean catch of *Galaxias auratus* larvae (≤ 40 mm FL) in pelagic samples for each spawning season (August-December) in (a) Lake Crescent and (b) Lake Sorell between 2000 and 2004. A linear model of the relationship in Lake Crescent is indicated by a dashed line.

Growth of young-of-the-year fish

Growth curves for YOY fish based on mean monthly lengths of cohorts followed similar patterns in Lake Crescent and Lake Sorell (Fig. 8). Despite larvae being present during winter in both lakes, there was hardly any growth until spring. The initial growth period generally occurred in September or October, but varied between lakes and years. In Lake Crescent (Fig. 8a), the mean length for September was significantly greater ($P < 0.0001$) than that of August during 2001, whereas this occurred in October in comparison to September in 2000, 2002, 2003 and 2004 (all $P < 0.05$). In Lake Sorell (Fig. 8b), mean lengths were greater in September than in August in 2001 and 2003 (both $P < 0.05$), October than in September in

2000 and 2004 (both $P < 0.0001$), and in November than in October in 2002 ($P < 0.0001$).

Mean monthly temperatures of pooled lake and year data increased from 4.2, 7.3, 9.9 and 13.7°C during August, September, October and November respectively. In all years, growth was rapid over summer in both populations. In Lake Crescent, where YOY were sometimes captured in the water column beyond April (Fig. 8a), growth slowed during the following autumn and trajectories approached asymptotic lengths (55-60 mm FL) by winter.

Logistic growth models fitted well to monthly length data for the 2000-2002 cohorts in both populations; however, due to few fish being captured after December (Fig. 8), growth of the 2003 cohorts was poorly estimated (i.e. L_{∞} estimates were large compared to maximum lengths and had large standard errors ($>128\%$)) (Table 1). Growth of cohorts differed significantly both between ($P < 0.01$) and within ($P < 0.0001$) the populations (both when the 2003 cohort was and was not included in the analyses). In both lakes, excluding the 2003 cohort, asymptotic lengths at the end of the first growing season (L_{∞} estimates) were similar across years (range = 56.56-61.20 mm FL) despite considerable variation in growth rates (K) (Table 1). Growth rates were always greater in Lake Crescent than in Lake Sorell in corresponding years and were noticeably higher in 2000 and 2002 ($K \sim 0.8$) (Table 1).

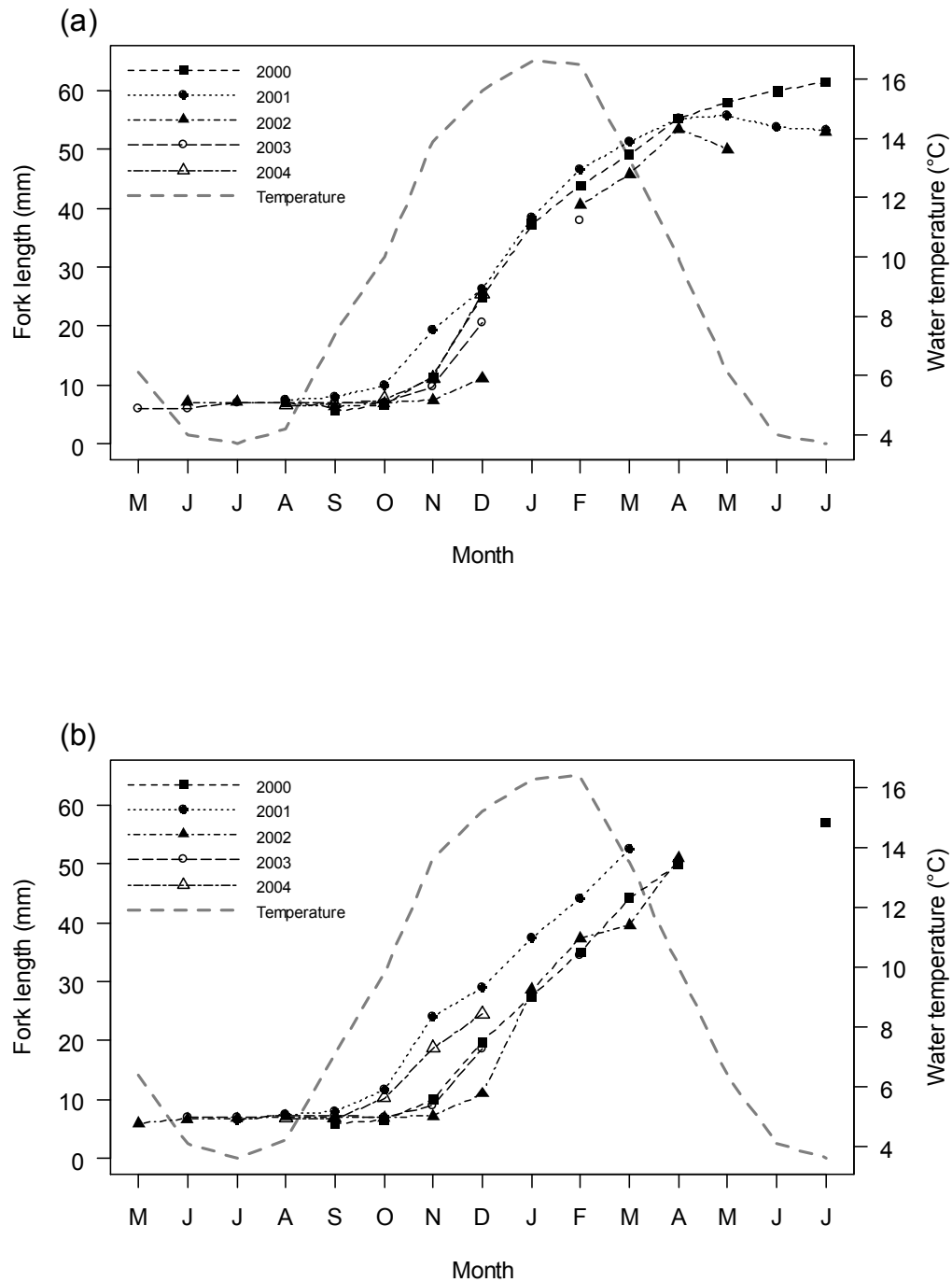


Fig. 8. Monthly mean lengths of young-of-the-year *Galaxias auratus* for each recruitment season (May to the following July) and mean water temperatures in (a) Lake Crescent and (b) Lake Sorell between 2000 and 2004. Logistic growth models for the 2000, 2001, 2002 and 2003 cohorts are shown in Table 1.

Table 1. Logistic growth model parameter estimates (± 1 S.E.) for *Galaxias auratus* cohorts in lakes Crescent and Sorell.

Lake	Cohort	<i>n</i>	L_{∞} (mm)	K (day ⁻¹)	t_0 (days)
Crescent	2000	1141	56.56 \pm 3.51	0.83 \pm 0.12	-4.36 \pm 0.25
	2001	1443	60.16 \pm 4.56	0.66 \pm 0.10	-4.23 \pm 0.34
	2002	461	60.31 \pm 6.36	0.79 \pm 0.14	-5.39 \pm 0.41
	2003	1522	86.15 \pm 110.36	0.52 \pm 0.26	-6.45 \pm 4.57
Sorell	2000	316	60.54 \pm 8.81	0.59 \pm 0.11	-5.37 \pm 0.63
	2001	1040	61.20 \pm 8.80	0.58 \pm 0.12	-4.14 \pm 0.63
	2002	1016	57.83 \pm 7.23	0.69 \pm 0.13	-5.39 \pm 0.49
	2003	888	188.55 \pm 1024.58	0.41 \pm 0.28	-9.64 \pm 18.55

Ontogenetic habitat shift

YOY fish in the 2000 (Fig. 9 and Fig. 10) and 2001 cohorts began to move from pelagic areas to adult habitats (littoral areas of fine sediment, sand and rock) in January (mean (\pm S.D.) sizes, Lake Crescent = 43.5 \pm 2.8 mm FL; Lake Sorell = 41.8 \pm 3.6 mm FL). The majority of YOY undertook this habitat shift between January and April in Lake Sorell with few individuals being captured in tow net samples after this time (Fig. 10a; see also Fig. 8b). The shift was generally more gradual in Lake Crescent with many fish remaining in the water column into the following spring – early summer (size ranges 40-80 mm FL) (Fig. 9a; see also Fig. 8a) even though these cohorts were also abundant in littoral samples at these times (see Fig. 5).

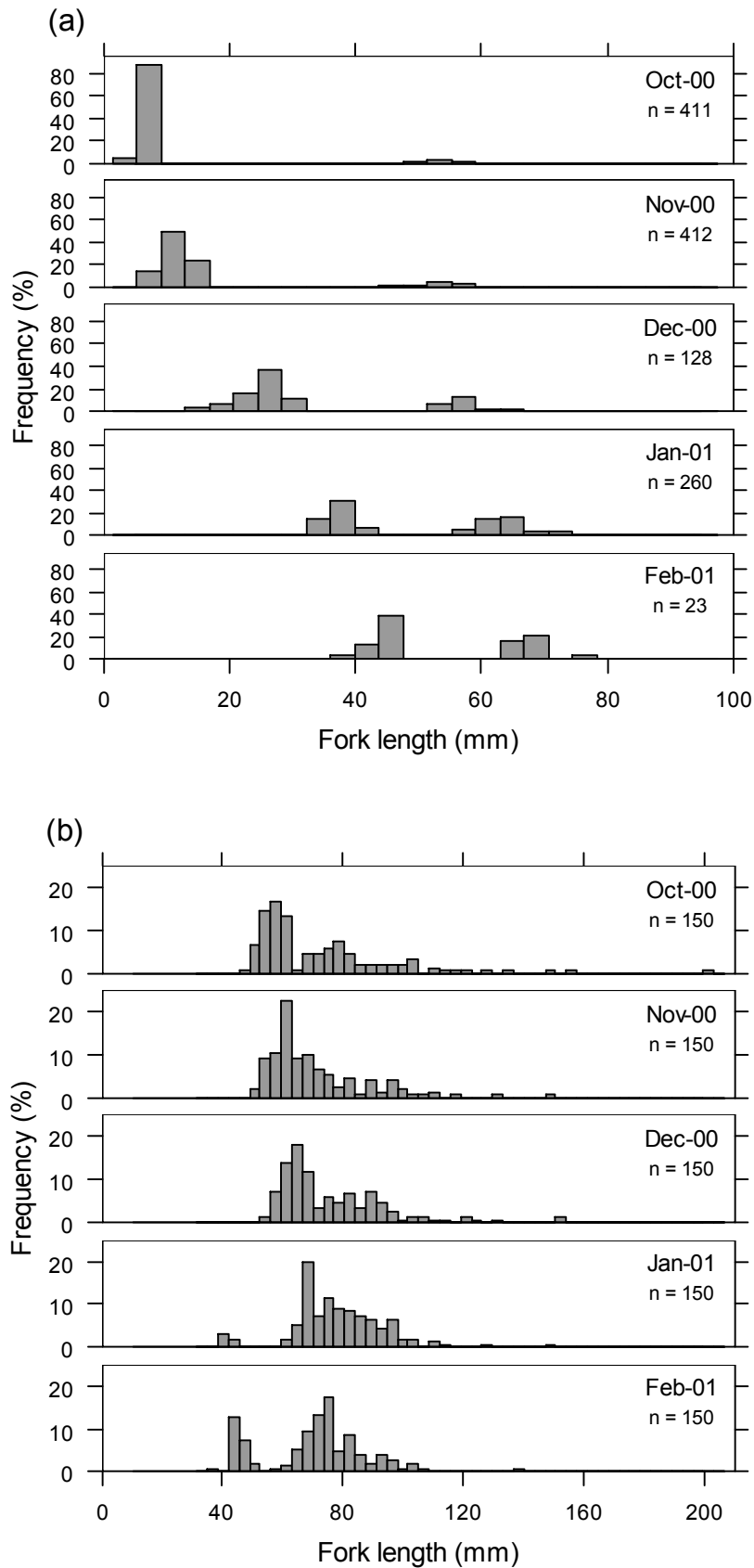


Fig. 9. Length frequencies of *Galaxias auratus* captured monthly from (a) pelagic and (b) littoral habitats in Lake Crescent between October 2000 and February 2001.

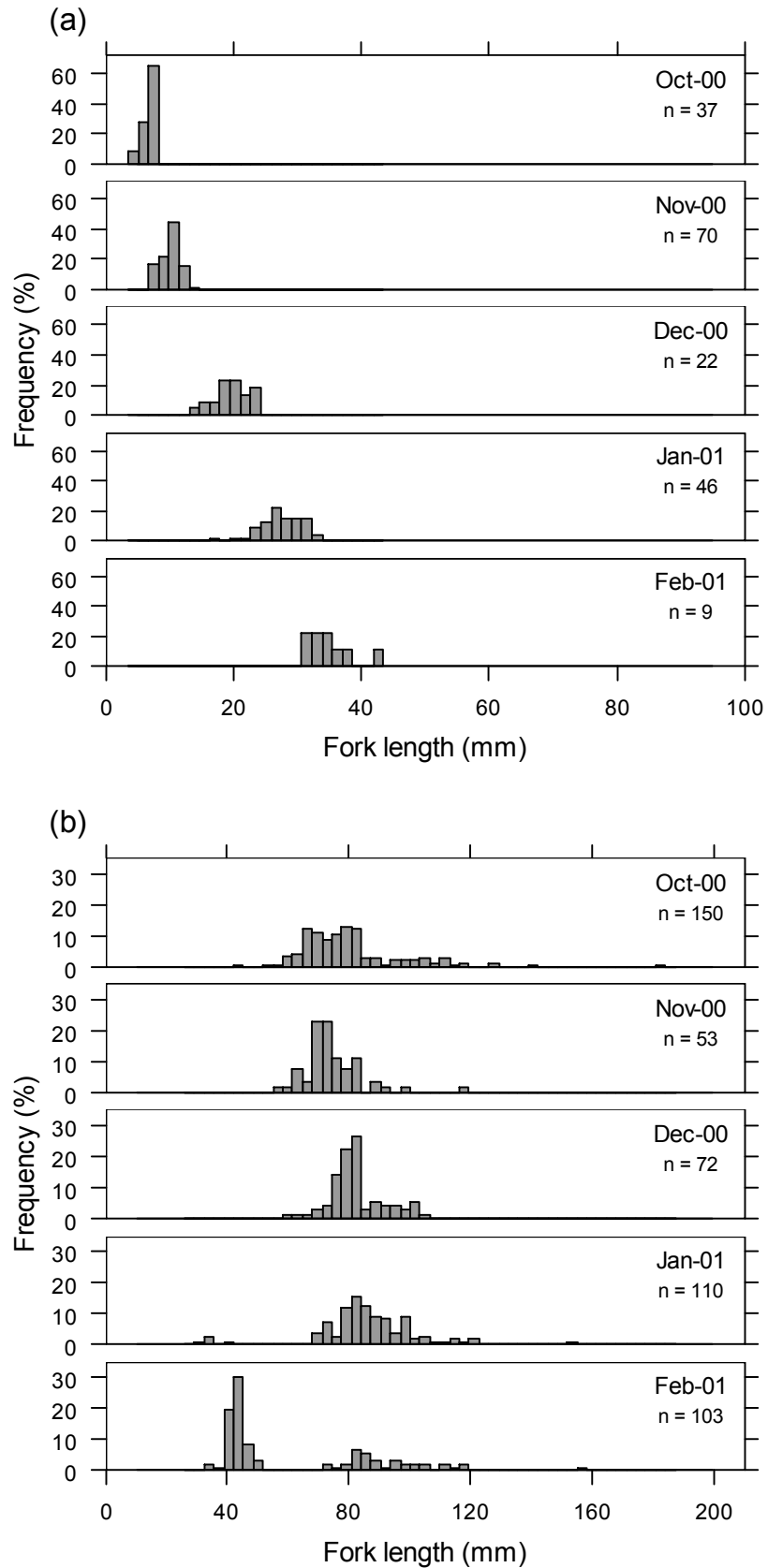


Fig. 10. Length frequencies of *Galaxias auratus* captured monthly from (a) pelagic and (b) littoral habitats in Lake Sorell between October 2000 and February 2001.

8.5. Discussion

Larval emergence

Fluctuations in water temperature often trigger spawning in temperate freshwater fishes (Wootton, 1998). However, other factors, such as the availability of suitable habitats, may also control reproduction (Gafny *et al.*, 1992) and the subsequent progression of early life stages (Sammons *et al.*, 1999). In this study, the timing and duration of emergence of *G. auratus* larvae varied considerably among years, both within and between populations, despite consistent and highly seasonal water temperatures. Newly hatched larvae (<8 mm FL) occurred as early as May in both lakes while water temperatures were ~6°C and decreasing, and as late as December (in Lake Sorell) when it had risen to ~15°C. Surprisingly, in both lakes, new larvae were present in pelagic habitats for 6-8 months during 2002 and 2003, and many were present during winter in 2001-2003 when water temperatures were generally <5°C and as low as ~1°C. This is unusual for a non-diadromous galaxiid as other species generally spawn during late winter – early summer (Humphries, 1989; Pen & Potter, 1991a) presumably to optimise the chances of larvae encountering favourable conditions (e.g. water temperatures and food resources). Indeed, limited data for other lacustrine galaxiids that inhabit highland (i.e. >500 m a.s.l.) lakes in south-eastern Australia (Fulton, 1982; Humphries, 1989), New Zealand (Rowe *et al.*, 2002a) and Patagonian Argentina (Barriga *et al.*, 2002) indicate their larval phases occur during spring or summer. Therefore, *G. auratus* have much earlier and more extended periods of emergence, which suggest earlier and more protracted spawning in response to factors other than ambient conditions. This is supported by the results of Hardie *et al.*, (in press).

Larvae were more prevalent during winter in Lake Sorell where ample spawning habitat was constantly available and water levels rose earlier than in Lake Crescent (i.e. initial rise of

0.1 m occurred 6-130 days earlier in each year). Therefore, the timing of increases in water levels during autumn – winter appeared to influence the emergence of larvae. Additionally, peaks in the occurrence of newly hatched larvae (>60%) happened earlier in both populations in 2001 than in all other years (excluding 2004 where no sampling was undertaken until August), when the earliest (late March) substantial rise (0.1 m) took place. Therefore, inter-annual variation in larval emergence in each lake was also linked to hydrology. Whilst these results reflect spawning times, increasing temperatures in late winter – spring are likely to complicate this relationship by reducing the incubation period of later spawned eggs, thus, synchronising the hatching of eggs spawned at different times (Karjalainen *et al.*, 2000). This is indicated by peaks in the occurrence of newly hatched larvae during spring when water temperature initially increased.

Abundance of larvae and hydrology

Temporal abundances of *G. auratus* larvae in lakes Crescent and Sorell also corresponded to differences in hydrology and habitat availability. Larvae tended to be more abundant in Lake Sorell than in Lake Crescent during winter, and their mean monthly abundances often peaked earlier than in Lake Crescent. This reflects differences in spawning in response to rises in water levels: in Lake Sorell fish spawned earlier because extensive areas of habitat were always available, whilst in Lake Crescent spawning was later and more protracted with progressive inundation of the limited habitat. Larval abundances did not correlate with increased base water levels in either lake; however, the mean seasonal abundances of larvae in Lake Crescent were strongly related to the magnitude of rises in water levels during spawning and egg incubation (i.e. May-September).

Other researchers have highlighted the influence of hydrology on the abundance of juvenile fish in lentic waters (Sammons & Bettoli, 2000; Maceina, 2003) and some have suggested that the governing mechanisms are related to changes in the availability of habitats for certain life stages (Hargeby *et al.*, 2005; Yamamoto *et al.*, 2006). Many littoral spawning species use complex habitats for breeding, and the condition or integrity of these areas is also critical to reproductive success (Bruton, 1985; Gafny *et al.*, 1992). *Galaxias auratus* scatter adhesive eggs, for which there is no parental care, in shallow (0.2-0.6 m deep) littoral areas (Hardie *et al.*, in press). In Lake Crescent, where loads of suspended particulate matter (mostly organic) vary considerably while water levels are low (i.e. range in concentrations of total suspended solids during study = 40-1000 mg L⁻¹; Uytendaal, 2006), most of the spawning habitat of this species is periodically exposed to wind-induced wave action and, hence, prone to sedimentation (S. A. Hardie, unpubl. obs.). Therefore, the observed relationship between hydrology and recruitment in Lake Crescent, may not only be related to the quantity of seasonal flooded spawning habitat, but its quality (e.g. degree of surface sedimentation).

Historically, both lakes Crescent and Sorell have contained submerged macrophytes during periods of higher water levels (~0.8 m greater than during this study). *Galaxias auratus* also use submerged aquatic vegetation as a spawning habitat when available (S. A. Hardie, unpubl. data). These areas are likely to provide additional spawning habitats that are not as susceptible as rocky shores to sedimentation. Thus, the observed hydrology-recruitment relationship in Lake Crescent may be weaker in wetter years as large areas of more favourable spawning habitats would be available.

Growth of YOY fish and habitat shift

Coincidence of the timing of larval production, favourable water temperatures and the availability and quantity of suitable food resources is crucial for recruitment (Cushing, 1990; King *et al.*, 2003). Despite the occurrence of *G. auratus* larvae in the pelagic zones of lakes Crescent and Sorell during winter, they did not begin to grow until spring when water temperatures rose above ~10°C. Therefore, the late autumn – early winter emergence of some larvae in both lakes appears disadvantageous to their growth and survival. The extended spawning periods in both populations did, however, ensure that some embryos emerged when conditions (i.e. both temperatures and food resources) were likely to be more favourable. Thus, even though larval emergence of this species was protracted, it was fish that hatched during late winter – spring which were likely to contribute to the YOY cohort.

Despite variations in abundance of YOY and the time at which their growth began, density-dependent factors did not appear to constrain growth in either lake. Summer growth of YOY cohorts did vary slightly (range in K for 2000-2002 cohorts = 0.58-0.83), but cohorts in both lakes reached similar lengths by the end of the first growing season (L_{∞} estimates for 2000-2002 cohorts = 56.56-61.20 mm FL). However, rapid growth by YOY between November and March is likely to be very important in these highland lakes. If emergence was delayed by unseasonal water level fluctuations, YOY could be smaller and in poorer condition at the end of their first growing season, thereby increasing mortality during the following winter (Byström *et al.*, 1998).

Similar to other lacustrine *Galaxias* species (Cussac *et al.*, 1992; Rowe & Chisnall, 1996a), *G. auratus* undergo littoral – pelagic – littoral ontogenetic habitat shifts during their life cycle. Such habitat shifts have been related to predation risk and resource limitation (Werner & Hall,

1988; Eklöv & Persson, 1996), and both are likely to be associated with this movement by *G. auratus*: galaxiids >40 mm FL are a significant diet item for salmonids in both lakes (Stuart-Smith *et al.*, 2004) and *G. auratus* shift from zooplankton to epibenthic invertebrate prey at this time (Hardie *et al.*, 2004). In both lakes, most juveniles (40-50 mm FL) began the shift to the littoral in January and, in Lake Sorell, basically all juveniles were in littoral areas by April. Conversely, in Lake Crescent where there was likely to be strong competition between adults and juveniles for the limited areas of rocky shore habitat, many individuals remained in the water column during the following winter and spring. Juvenile *G. auratus* have been found to use littoral macrophytes for shelter when available (Hardie *et al.*, 2006a), therefore, higher water levels which allow YOY to access the extensive littoral wetlands of these lakes are likely to increase recruitment by providing additional nursery habitats for juveniles.

Water levels and conservation of lacustrine galaxiids

Whilst galaxiids occur in an array of habitats, many inhabit lentic waters (McDowall, 2000; Allen *et al.*, 2002). Several species occur in relatively small populations that are geographically isolated due to either natural or anthropogenic barriers. In many cases, these restricted distributions and localised threats have led to their imperilled status (McDowall & Rowe, 1996; Koster, 2003; Hardie *et al.*, 2006b). Unlike riverine diadromous populations which can be replenished by immigration, non-diadromous lacustrine populations may have limited ability to facilitate internal recovery from major perturbations such as impoundment (Hamr, 1995).

Currently, little is known of the roles of hydrological variables in the life histories of lacustrine galaxiids. In this study, the availability of complex littoral habitats was controlled

by water level fluctuations and we found associations between hydrology, the timing of emergence and abundance of *G. auratus* larvae. Clearly, the observed relationship between larval abundances and water level fluctuations in Lake Crescent shows that hydrology can heavily influence the recruitment of species which rely on littoral habitats for spawning. Therefore, besides the need to address the lack of knowledge regarding the biology of many galaxiids (Hardie *et al.*, 2006b), future research should also focus on defining relationships between hydrological variables and life history strategies, and habitats that are limiting for different life stages (Rosenfeld & Hatfield, 2006). Because many galaxiid species use littoral habitats during early life stages (Fulton, 1982; Rowe & Chisnall, 1996a), hydrological variables may play key roles in their recruitment. Furthermore, we suggest that alterations to seasonal hydrological regimes pose significant threats to the galaxiid fauna of the TCP and lacustrine galaxiids in other regions.

9. General Discussion

Tasmanian galaxiids possess attributes that are typical of threatened freshwater fishes in many regions: small size, some are migratory, many have restricted distributions, and most have no commercial or recreational value (Angermeier, 1995; Economidis, 2002; Fagan *et al.*, 2002). Therefore, it is not surprising that the Tasmanian fauna is so imperilled (i.e. 69% of species are considered threatened; Chapter 2). Several Tasmanian species have recently been included in recovery plans (11 species in the most recent plan) (Crook & Sanger, 1997; Threatened Species Section, 2006); however, this does not necessarily safeguard the fauna from further decline. Currently, due to insufficient knowledge of life histories and the risks associated with perceived threats, robust species-specific recovery targets for Tasmanian species can not be set.

Clearly, if management of the Tasmanian galaxiid fauna is to ensure its long-term viability, three key areas need to be investigated: (1) the biology of species, (2), their ecology and environmental requirements, and (3) processes which influence their populations. This thesis has examined the conservation biology of *Galaxias auratus* Johnston, a previously unstudied Tasmanian lacustrine species, by focusing on these key areas, as well as methods to monitor its populations. The outcomes of this work have provided: (1) insight into suitable methodologies for monitoring lacustrine galaxiid populations, (2) determined vulnerable aspects of the biology of *G. auratus* and critical phases of their life cycle, and (3) highlighted the importance of hydrological variables and the availability of littoral habitats in the life history of this species. Based on these outcomes, issues relating to the conservation and of *G. auratus* and other lacustrine galaxiids are discussed, and directions for management of these species and further research are outlined.

9.1. Conservation of the golden galaxias and other lacustrine galaxiids

Population monitoring

Due to their small-size and preference for complex, littoral habitats (McDowall & Frankenberg, 1981), epibenthic lacustrine galaxiids are difficult to sample quantitatively, hence, estimates of population size are not typically used in their management. However, qualitative monitoring (i.e. using catch-per-unit-effort (CPUE) data) is often used to determine the status of populations (Rowe & Chisnall, 1997b; Rowe *et al.*, 2002a; Threatened Species Section, 2006). Without assessments of the efficiency of the chosen sampling gear(s) and the catchability of target species, the value of such work can be limited.

For instance, in Tasmania, despite not being formally evaluated, monitoring regimes for different galaxiid species across a range of habitats (e.g. shallow marshes and large lakes) rely primarily upon a single methodology: backpack electrofishing in wadable areas during daylight. Furthermore, the timing of annual sampling events is governed by climatic conditions and regimented field schedules, with monitoring occurring on a single occasion between spring and autumn. In some cases, these regimes have been unable to detect declines until populations have disappeared (e.g. *Paragalaxias mesotes* McDowall and Fulton population in Woods Lake during the mid-1990s (Threatened Species Section, 2006)). Therefore, other than providing some degree of presence/absence information, a cost-benefit analysis of these monitoring strategies is likely to be unfavourable to their continued use.

In Chapter 3, the comparison of three applicable sampling methods (fyke netting, electrofishing and snorkelling) during day-time and at night showed that the abundance and size-structure of catches of *G. auratus* can vary considerably depending on the regime used. Both adults and juveniles were nocturnally active with many adults moving into littoral areas

at night. Additionally, monthly catches of *G. auratus* in fyke nets in lakes Crescent and Sorell (Chapter 4) varied seasonally with substantial increases during their extended spawning period. These findings question the effectiveness of current population monitoring strategies in Tasmania, given the lack of information about the biology and activity patterns of the other species being ‘managed’. Clearly, the aims of monitoring programs need to be well defined. If they are intended to detect population trends (i.e. changes in abundance or size-structure), then fundamental requirements include knowledge of relevant life history attributes such as the biology and habitat use (Chapters 3 and 4), activity patterns (Chapter 3), and also catchability towards sampling gear (Chapter 3).

Sophisticated population viability analysis (PVA) models are increasingly being used to assess the status of threatened species (Brook *et al.*, 2000). They require substantial data for their parameterisation, but such data are often difficult to acquire for threatened species that occur in low abundance. However, habitat-abundance (i.e. CPUE data) models may provide a good starting point for conservation planning (Chu *et al.*, 2006; Rosenfeld & Hatfield, 2006). Therefore, I propose an alternative approach which focuses on habitat condition for monitoring the status of some galaxiid species. By quantifying factors that govern population trends (such as access to certain habitats or abundance of piscine predators) and routinely assessing them in conjunction with qualitative sampling of the target species, more precise estimates could be made. Rosenfeld and Hatfield (2006) proposed population limitation models for different life stages (eggs, juveniles and adults) of freshwater fish which demonstrate this approach and these models are applicable to *G. auratus* populations. During this study, access to rocky littoral areas at spawning time (Chapters 6 and 8) and the abundance of salmonid predators (Chapters 4 and 5), were the most influential factors on the abundance of *G. auratus* populations in lakes Crescent and Sorell. Access to spawning

habitats for *G. auratus* is controlled by water levels (particularly in Lake Crescent) and salmonid (*Salmo trutta* L. and *Oncorhynchus mykiss* Walbaum) abundances vary according to stocking rates and flows in tributaries during their spawning times. All of these factors are reasonably easy to monitor and, in fact, some of this information is already gathered by different agencies. Whilst different factors are likely to be important for other galaxiids (e.g. the extent of submerged charophyte algal beds for *Paragalaxias* spp.; Chapter 2), when applied to other species, this approach would help direct research towards important information gaps and provide more precise estimates of the status of populations in the short- and long-term. Additionally, if such models can be validated by standardised population surveys at times of different habitat conditions, the current regular frequency of qualitative monitoring could be refined to target periods where the viability of populations is uncertain. In the long-term, this would make better use of the limited financial and logistical resources that are available for managing threatened galaxiids.

Biology and critical life cycle phases

Sudden declines in freshwater fish populations usually occur due to interactions between biological limitations and anthropogenic impacts (Bruton, 1995; Richter *et al.*, 1997). Many galaxiid species have comparable life histories and habitat associations (e.g. *G. auratus* (Chapters 3, 5, 6 and 8) and lacustrine *Galaxias truttaceus* Valenciennes (Humphries, 1989; Morgan, 2003) and *Galaxias brevipinnis* Günther (Rowe *et al.*, 2002a)); therefore, similar biological parameters and life cycle requirements may contribute to the risk of endangerment of populations in different regions. Whilst *G. auratus* populations in lakes Crescent and Sorell were abundant at the time of this study, this research revealed several attributes of their biology and life history that make them vulnerable (Table 1).

Table 1. Biological and life history attributes of *Galaxias auratus* that are likely to cause their populations to be vulnerable to sudden declines.

Biological attribute	Threat
Small-size	Although some individuals reach >200 mm in length, most are <120 mm long and are, therefore, susceptible to predation and competitive exclusion from preferred habitats by introduced salmonids and other larger and more aggressive species (Chapter 5).
Short longevity	While some individuals live for up to 10 years, most fish reach ≤ 3 years of age and males generally live for ≤ 2 years (Chapter 5). Therefore, the resilience of populations to major perturbations (such as poor recruitment events) is likely to be poor.
Unbalanced sex ratios	Male to female sex ratios are strongly female biased (<i>c.</i> 1:2) (Chapter 6) and, therefore, reproduction may be limited if males, which are generally shorter-lived (≤ 2 years) than females (Chapter 5), become less abundant due to consecutive poor recruitment events.
Long incubation period of unguarded eggs	Eggs incubate in littoral areas that are shallow (<0.6 m) and turbulent (i.e. exposed to strong wave action) for approximately 30–45 days (Chapter 6). This means they are vulnerable to changes in the condition of these habitats (i.e. dewatering and sedimentation; Chapters 6 and 8) and predation by aquatic predators.
Reliance upon complex littoral habitats at several life stages	Littoral habitats are used for spawning and as nursery habitats for juveniles, and adults also periodically use these areas during diel movements for feeding and refuge (Chapters 3, 6 and 8). Because the availability and condition of these habitats is influenced by hydrology (Chapters 6 and 8), populations are susceptible to hydrological manipulations and climatic variability.
Succession of life cycle stages is influenced by hydrology	The timing of spawning and recruitment events is strongly influenced by hydrological variables. Seasonal water level fluctuations may provide stimuli for spawning (Chapters 6) and the magnitude of rises during spawning control the abundance of larvae and the strength of recruitment (Chapter 8). Hydrological manipulations may decouple natural seasonal water level and water temperatures regimes, thus, create adverse conditions for spawning and/or the development of juveniles.

Galaxias auratus is one of the largest lacustrine galaxiids (McDowall & Fulton, 1996; McDowall, 2000); however, the restricted age- and size-structure of their populations make them susceptible to short-term impacts (i.e. over 1-3 breeding seasons) (Table 1). Because males generally only live for <3 years and are 1-fold less abundant than females, the reproductive capacity of their populations could be severely reduced by events (e.g. recruitment failures) which further skew sex ratios. Additionally, overall population structure appeared to be constrained by competition with and predation by introduced salmonids (Chapter 5). Therefore, whilst *G. auratus* coexists with salmonids, these introduced predators may further reduce the capacity of their populations to recover from sudden perturbations.

Similar to several other lacustrine galaxiids (Fulton, 1982; Humphries, 1989; Rowe & Chisnall, 1996a), *G. auratus* use the littoral zone at several life stages; therefore, the quantity and quality of this habitat is critical to their populations (Table 1). In particular, the availability and condition of complex littoral habitats (i.e. littoral areas of rock substrata or macrophytes) in shallow areas (<1 m deep) is of utmost importance. These habitats are used for spawning and incubation of eggs (mostly 0.2-0.6 m deep), as nursery areas for juveniles in the later stages of development, and by adults for feeding and refuge. The bathymetry and disposition of spawning habitats in lakes Crescent and Sorell mean that the availability of spawning habitat is heavily influenced by hydrology, and the influence of hydrology differs between these two lakes (Chapters 6 and 8). Other processes such as eutrophication (Uytendaal, 2006), as well as wind-resuspension and transport of fine sediments, are likely to effect the condition (i.e. degree of sediment cover and interstitial complexity) of these habitats. Therefore, resource bottlenecks in littoral habitats at several of their life stages (i.e. spawning, recruitment, and adult life) may limit the size of *G. auratus* populations. Furthermore, because spawning and the incubation of their eggs are influenced by

hydrological conditions, their populations are vulnerable to hydrological manipulations and extremes in climatic variability. These findings suggest that both intrinsic biological parameters which limit the resilience of populations and extrinsic processes that negatively impact on life cycles should both be considered in risk assessments of galaxiid populations.

Hydrology and habitat availability

In natural lentic waters, seasonal hydrological cycles control the temporal and spatial availability of littoral habitats (Gasith & Gafny, 1990; Winfield, 2004a). However, in systems that are subject to anthropogenic hydrological manipulations (e.g. hydro-electric impoundments), seasonality in water level fluctuations may no longer exist or even be reversed, and complexity of littoral habitats may be altered. For the duration of this research, hydrology governed the availability of critical habitats (i.e. rocky shores and littoral wetlands) for *G. auratus* and had a profound influence on their life cycle, particularly in Lake Crescent (Chapters 6 and 8; Fig. 1). Given the impacts from hydrological manipulations to the life stages associated with reproduction and recruitment (Fig. 1), prolonged occurrence of these conditions have the potential to significantly reduce the abundance and, therefore, viability of *G. auratus* populations. Clearly, similar scenarios may have deleterious impacts on other lacustrine galaxiids and these threats could have contributed to the recent declines in some Tasmanian galaxiid populations, such as *P. mesotes* in hydrologically-altered Woods Lake (Threatened Species Section, 2006). However, to fully address this question we require further understanding of the life histories and basic habitat associations of other species. The findings of this thesis indicate that such work should focus on reproductive strategies and quantitative surveys of habitats which are likely to be critical for different life stages (Rowe *et al.*, 2002b), such as littoral areas of rocky substrate (spawning habitats; Chapter 6) or macrophytes (shelter for juveniles; Chapter 3).

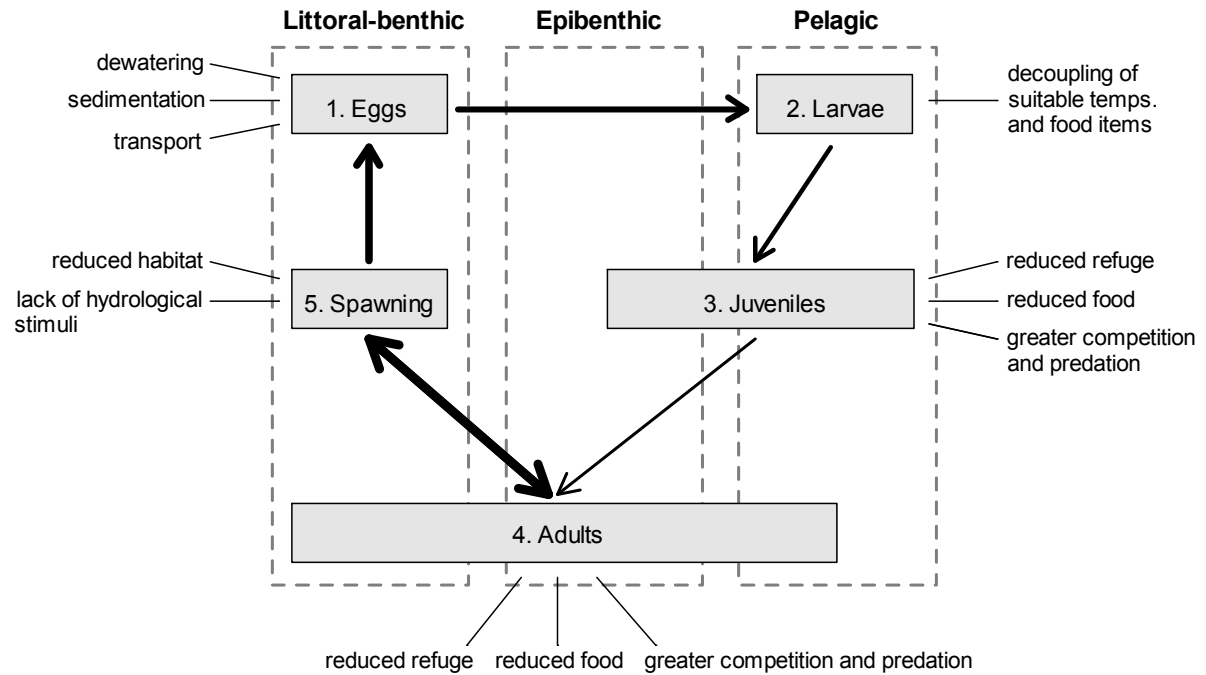


Fig. 1. Impacts of hydrological manipulations (primarily reduced water levels and marginal or unseasonal fluctuations) on five stages in the life cycle of *Galaxias auratus* based on the findings of this thesis (Chapters 3, 5, 6 and 8). The model assumes the hydrological alterations have occurred over ≤ 2 years and, hence, adults remain abundant. The relative strength of the pathways between each life stage is indicated by the thickness of the arrows. The three primarily habitats used during each stage are shown and the relative proportion of time each life stage occupies these habitats is represented by the position of the horizontal bars.

Habitats evolve through natural and anthropogenic processes (Naiman & Latterell, 2005) and the persistence of freshwater fish populations relies upon their ability to adapt to these changes. In Tasmanian lakes, the two most influential factors on the condition of physical habitats are: (1) impoundment and subsequent manipulations, and (2) climatic variability. For example, prior to the damming of their outflows over a century ago, Lake Crescent and Lake Sorell were shallow lagoons which contained abundant submerged macrophytes throughout their basins (LCWP, 1987). Furthermore, periodic El Nino-induced droughts strongly influence the hydrology and habitat landscapes of these lakes by altering typical seasonal

water level fluctuations and lowering water levels (Chapter 8). However, *G. auratus* populations have remained abundant and, therefore, must be more flexible in their life history than some other Tasmanian species which have undergone significant declines following major alterations of their habitats (e.g. *Galaxias pedderensis* Frankenberg in the recently impounded Lake Pedder (Hamr, 1995)). Thus, in the long-term, management of galaxiids with restricted distributions should consider their resilience to habitat alterations such as those that are associated with changes in hydrological conditions. The ability to use different habitats (depending on availability) at certain life stages, such as spawning, is likely to influence the viability of galaxiid populations in hydrologically altered systems.

9.2. Management strategies and further research

Golden galaxias

The two main threats to *G. auratus* populations in lakes Crescent and Sorell are: (1) altered water level regimes (Chapters 5, 6 and 8) and (2) salmonid competition/predation (Chapter 5; Stuart-Smith *et al.*, 2004) (for further information about other threats and management strategies for this species see Hardie, 2003). Whilst the hydrology of these shallow lakes is strongly influenced by climatic variability (such as the drought that occurred during this study; Chapter 8), there is scope to include the habitat requirements of *G. auratus* in the hydrological management of these lakes. Currently, sluice gates in the Interlaken Canal (which connects the two lakes) and the outflow of Lake Crescent are typically used to optimise the volume of water stored in the system and supply water at appropriate times to downstream users (irrigators and municipalities). This means there is often no discharge from Lake Sorell into Lake Crescent, nor out of Lake Crescent outside of the irrigation season (i.e. during May-November); hence, seasonal water level rises in Lake Crescent are not as great nor as rapid as those in Lake Sorell which receives inflows from the system's primarily

tributary, Mountain Creek. Given the findings of this thesis (Chapters 6 and 8), sluice gates in this system should be routinely operated to increase the availability and enhance the condition of critical habitats (i.e. spawning substrates). Where possible, such manipulations should include a minimum water level of 802.20 m AHD in Lake Crescent (above which littoral areas of cobble substrate are inundated) during autumn and seasonal variation in water levels (i.e. rises during late autumn – winter and falls during late spring – summer) in both lakes. Even during periods of relatively low water levels, these operating rules will enable access to complex habitats that are important for feeding and refuge, and provide suitable conditions for spawning and recruitment.

Galaxias auratus has coexisted with introduced salmonids for over a century in lakes Crescent and Sorell (Fulton, 1990); however, competition and predation from these fishes limits the abundance (Chapter 4), and growth and age-structure (Chapter 5) of their populations. Salmonid populations in these lakes provide valuable recreational fisheries, but currently management of these fisheries is not guided by robust science (i.e. accurate stock-recruitment relationships or cost-benefit analyses of management practices). Clearly, impacts from salmonids would compound the influence of other threats to *G. auratus* populations (e.g. poor recruitment due to unsuitable hydrological conditions), particularly when they are under stress, such as during periods of low water levels when refuge habitats are scarce. In the long-term, if lakes Crescent and Sorell are to support salmonid fisheries, then the mechanisms that have, to date, allowed these species to coexist should be examined (Stuart-Smith *et al.*, 2007). In light of such information, the compounding affects of the multiple threats to *G. auratus* populations should be investigated. In the mean time, if serious declines in the abundance of *G. auratus* populations occur, the abundance of salmonids should be reduced by altering

stocking strategies, limiting access to their primarily spawning habitat (gravel beds in inflowing tributaries), and if necessary, physical removal.

The *G. auratus* populations in lakes Crescent and Sorell provide unique opportunities to study the ecology of a lacustrine galaxiid. These physically similar, shallow mesotrophic lakes contain abundant *G. auratus* populations which can be sampled at all life stages (i.e. eggs, larvae, juveniles and adults). This thesis has substantially increased our knowledge of this species and provides a platform for further work to investigate population dynamics and fine-scale processes which may impact on certain life stages.

The historically low water levels that occurred during this study prevented assessment of the importance of the littoral wetlands of lakes Crescent and Sorell to *G. auratus*. In the translocated population, *G. auratus* used aquatic vegetation for spawning (S. A. Hardie, unpublished observations), and refuge and feeding (Chapter 3). Therefore, depending on the timing and duration of flooding, the littoral wetlands in the Crescent-Sorell system may significantly increase habitat for all life stages during periods of higher water levels. The flooding of these areas would provide *G. auratus* with a choice of complex littoral habitats (i.e. rock substrates and macrophytes). Further work to examine how *G. auratus* use this additional habitat, especially in relation to reproduction and recruitment, would enable the flexibility of their life history and habitat use strategies to be tested. Furthermore, long-term studies of the dynamics of several life stages could be used to model quantitatively the response of their populations to two impacts which are common to lacustrine galaxiids: (1) altered water level regimes and (2) salmonid competition/predation. Such work would aid sustainable management of water resources, recreational fisheries and populations of other threatened galaxiids which have similar life histories.

At finer spatial and temporal scales, the ability to locate and monitor *G. auratus* spawning sites should be used to identify the variables which trigger spawning and govern the success of this critical life stage. Field and laboratory experiments could be used to examine microhabitat selection, proximal spawning stimuli and the main threat to this life stage that was observed during this study: sedimentation of spawning habitats and incubating eggs. This information would aid management of this species' habitat and provide further insight into the environmental requirements of other galaxiids.

Other lacustrine galaxiids and littoral fishes

Despite recent work (Fulton, 1982; Humphries, 1989; Rowe & Chisnall, 1996a; Barriga *et al.*, 2002; Rowe *et al.*, 2002a; Morgan *et al.*, 2005; Chapman *et al.*, 2006), currently little is known of the biology, ecology and habitat requirements of many galaxiids, including Tasmanian species. Given the findings of this research, identifying habitats that limit populations based on species biology should be a central component of galaxiid conservation programs. Additionally, traits that provide populations with resilience to major perturbations, such as flexibility in recruitment strategies, should also be investigated.

Efforts to conserve galaxiid populations are often poorly resourced; therefore, future research should focus on species (and/or populations) which are likely to have life histories and habitat associations that are common to several species. For example, in Tasmania, there is little information regarding the biology and habitat requirements of the four endemic *Paragalaxias* spp. (a unique lacustrine genus of relatively small benthic fishes). Currently, financial and logistical resources are used primarily to monitor annually the 'status' of all 11 threatened galaxiids that are included in a recovery plan (Threatened Species Section, 2006). In some instances, this work has failed to detect declines of populations until after they have

disappeared (such as *P. mesotes* in hydrologically-altered Woods Lake (Threatened Species Section, 2006)). To make better use of these limited resources, routine monitoring should be scaled-back for species and/or populations that, according to the best available information, are abundant and well protected by the security of areas where they occur (such as populations that occur in World Heritage Areas, e.g. *G. pedderensis*). This would allow life histories and habitat requirements of species that are clearly more vulnerable to be investigated; therefore, enable informed management strategies to be developed for other galaxiid species.

Fish populations may be limited by habitat quantity and quality at more than one life stage, thus, there may be multiple leverage points to increase population size and/or viability (Rosenfeld & Hatfield, 2006). For small-sized epibenthic fishes such as galaxiids, littoral habitats often provide resources for several life stages (Winfield, 2004a); therefore, processes that control the condition of these areas, such as water levels, can be used to manage their populations. Future studies should treat management actions, such as hydrological manipulations, as experiments (Naiman & Latterell, 2005) to investigate the biology-habitat linkages of littoral fishes in whole-lake ecosystems. For example, during wetter years, in the Crescent-Sorell system, sluice gates could be operated to provide appropriately timed water releases from Lake Sorell into Lake Crescent to investigate the microhabitat selection of *G. auratus* for spawning and the importance of flooded littoral wetlands for juveniles and adults. Similarly, hydrological manipulations in impoundments in other regions may also provide opportunities to investigate the roles of hydrological variables in the life cycles of other littoral species in whole-lake ecosystems.

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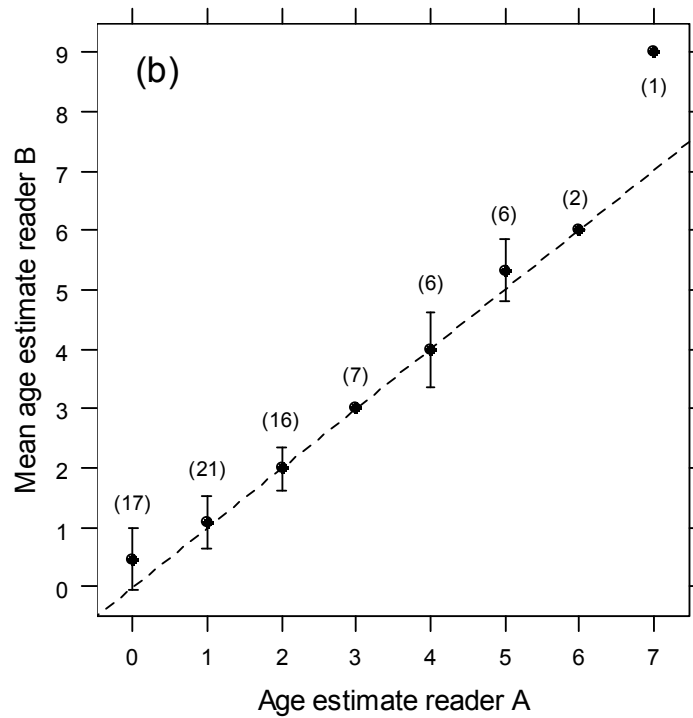
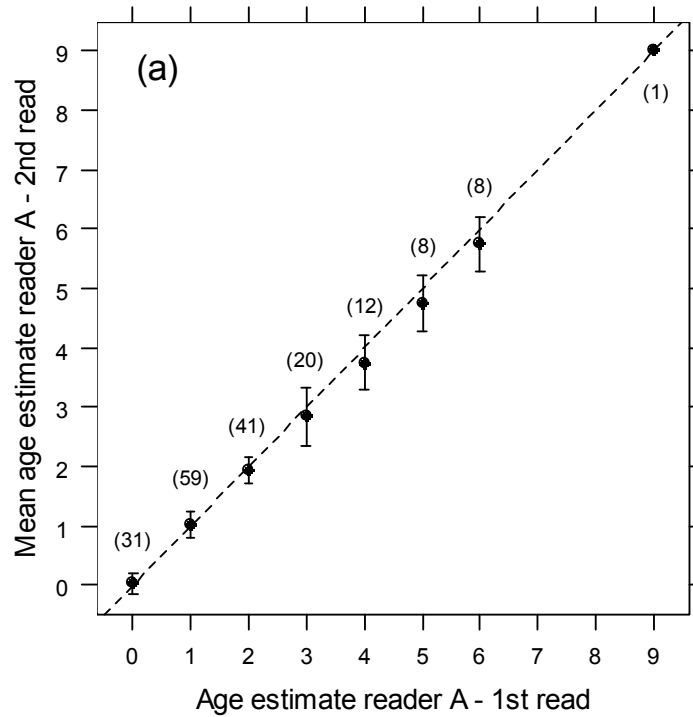
Appendices

Appendix 1. Major threats to galaxiid species in Tasmania*.

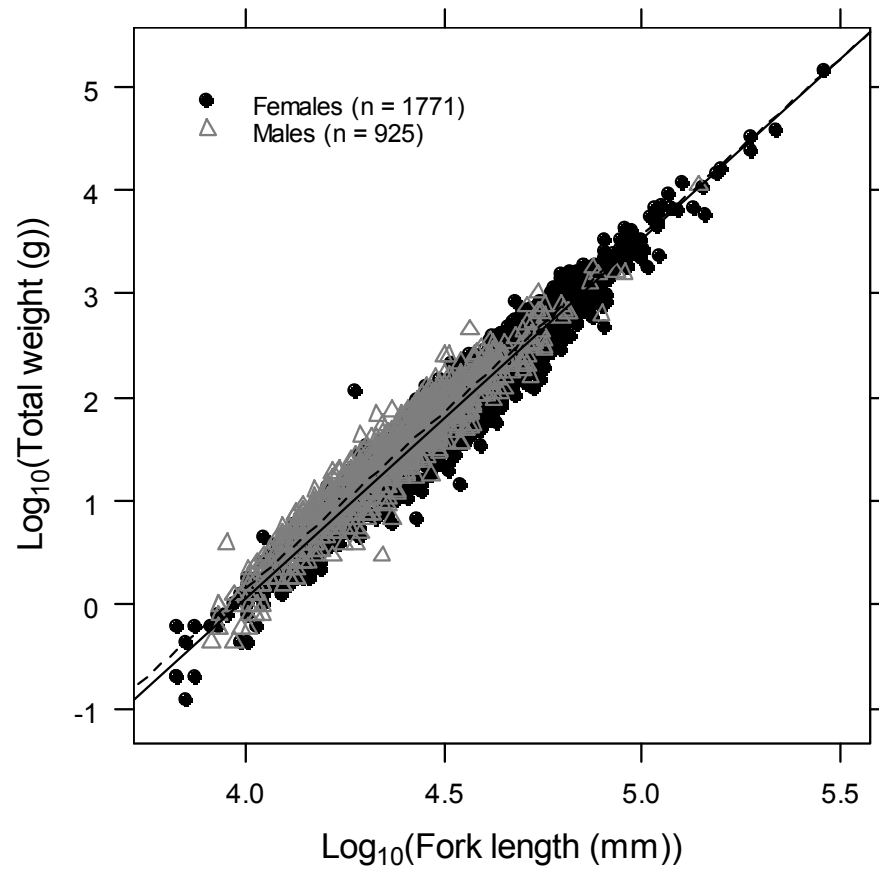
Species	Threat				
	Exotic species	Hydrological manipulation	Restricted distribution	Habitat degradation	Exploitation
<i>Galaxias pedderensis</i>	+	+	+		
<i>Galaxias fontanus</i>	+		+	+	
<i>Galaxias johnstoni</i>	+		+	+	
<i>Galaxias tanycephalus</i>	+	+	+		
<i>Galaxias auratus</i>	+	+	+		
<i>Galaxias parvus</i>	+	+	+		
<i>Galaxias brevipinnis</i>	+	+		+	
<i>Galaxias truttaceus</i>	+	+		+	
<i>Galaxias maculatus</i>	+	+		+	
<i>Paragalaxias mesotes</i>	+	+	+		
<i>Paragalaxias dissimilis</i>	+	+	+		
<i>Paragalaxias eleotroides</i>	+	+	+		
<i>Paragalaxias julianus</i>	+	+	+		
<i>Neochanna cleaveri</i>	+			+	+
<i>Galaxiella pusilla</i>	+		+	+	
<i>Lovettia sealii</i>	+	+			+
Total (no. and (%))	16 (100)	12 (75)	11 (69)	7 (44)	2 (13)

*The three key threats to each galaxiid species were determined based on current available documented information: (Blackburn, 1950; Fulton & Pavuk, 1988; Sanger & Fulton, 1991; Hamr, 1992a; Ault & White, 1994; Hamr, 1995; Chilcott & Humphries, 1996; Crook & Sanger, 1998a, b; Lake, 1998; Hardie, 2003; Koster, 2003; Hardie *et al.*, 2004; Jackson *et al.*, 2004; Threatened Species Section, 2006) and Inland Fisheries Service, Tasmania unpublished data.

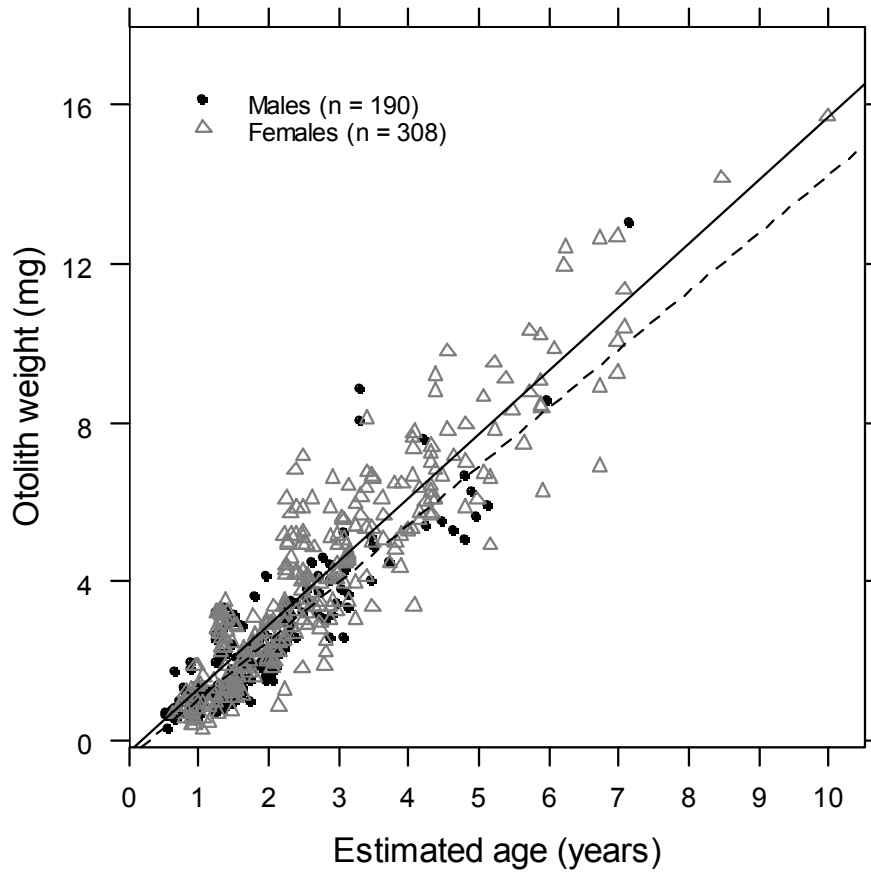
Appendix 2. Age-bias plots of mean (± 1 S.D.) age estimates of *Galaxias auratus* compared to the initial reading by the primary reader: (a) within reader bias and (b) between reader bias. Sample sizes and equal age relationships (dashed line) are indicated.



Appendix 3. Fish length-total weight relationships for male (dashed line) and female (solid line) *Galaxias auratus*. Data from lakes Crescent and Sorell are pooled. Linear model parameter estimates are given in Chapter 5, Table 1.



Appendix 4. Estimated age-otolith weight relationships for male (dashed line) and female (solid line) *Galaxias auratus*. Data from lakes Crescent and Sorell are pooled. Linear model parameter estimates are given in Chapter 5, Table 1.



Appendix 5. Fish length-fecundity relationship for *Galaxias auratus*. Data from lakes Crescent and Sorell are pooled. See Chapter 6 for linear model (dashed line) parameter estimates.

