Fish assemblage response to experimental rehabilitation in the Glenelg River, Victoria, Australia

by

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Submitted in fulfilment of the requirements for the degree of

Doctor of Philosophy

Deakin University

July, 2013
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Acknowledgements

Emabarking on a significant journey such as this and seeing its conclusion, could not have been done without the help of a few fellow travellers along the way. Instigation and completion of this study could not have been undertaken without the assistance of the following people.

First, I would like to especially thank my supervisor’s: Associate Professor Belinda Robson, Dr Ty Matthews and Associate Professor Brad Mitchell. I am greatly appreciative of you always making time to discuss issues at any moment, even when hundreds to thousands of kilometres away and providing much appreciated support, especially throwing work my way to make life more content. Your enthusiasm and assistance was invaluable, and I am especially grateful your wisdom, support, encouragement and of course, friendship.

I am also indebted to the knowledge passed to me from helpful discussions with Dr. Ed Chester, Dr. Alecia Belgrove, Dr. Craig Styan, Professor Gerry Quinn, Associate Professor Laurie Laurenson, Dr. Paul Jones and Dr. Daniel Ierodiaconnou. Together with my supervisors, I have learnt alot and especially grateful for meeting such a great bunch of people who gave me very different insights into the discipline of ecology. Your discussions, experiences and thoughts were invaluable and have immensely assisted my development in the field of ecology. Anonymous reviewers of several chapters of this work assisted in improving earlier drafts, and provided much appreciated external insights into the communication of my thoughts to the wider scientific audience.

Peter Swanson from Glenelg Hopkins CMA who provided water quality and river discharge data on the Harrow rehabilitated reach. Peter and Heinz de Chelard (Earth Tech Pty. Ltd.) also provided informative discussions on the rehabilitation procedure at Harrow, while Wayne Koster from Arthur Rylah Institute, Department of Sustainability and Environment provided before-period fish data. This project would also not be possible from the assisted financial support of Deakin University,
Fisheries Victoria and Glenelg-Hopkins Catchment Management Authority. This research was supported by research permits from the Department of Primary Industries, the Department of Sustainability and Environment and the Deakin University Animal Welfare Committee.

To those volunteers who assisted invaluably in the long weeks of field work, Joanne Dono, David Lambert, Kerrelyn Johnston, Alistair Becker, Mini Hogan, Chris Kennedy, Jake Toe, Mike Truong, Pete Kolotelo, and to Joanne Dono and Rachel Dono for review and editorial assistance, a special thank you as this project could not have been completed without your assistance.

To my family, your support and putting up with me over the years has given me the opportunity to pursue an long-life understanding of fish and their environments. I sincerely hope that shouldering all of the nuances of my personality, the best and particularly worst of times from my efforts, were worth it. To my friends and everyone else who assisted me along the way, a big thank you, your support and 'encouragement' ("when is it going to be finished?") helped to cross the line.

Finally, a extra-special thankyou to Jo. There is no doubt that without you, your wisdom, patience, support and love, I could not have made it.
List of Thesis Publications

These articles have been accepted and published from research completed as part of this thesis.


Summary

A number of historical anthropogenic, press-type disturbances have caused severe and lasting structural changes to the habitat of river-dwelling organisms. Conceptually, habitat restoration may offer a solution to sustained river degradation, but implementing restoration strategies to address these disturbances remains a significant challenge. The Glenelg River in western Victoria is a classic example of an Australian river impacted by multiple press-disturbances. This study examined the response of fish assemblages over a three year period to experimental rehabilitation consisting of re-introduced woody debris and sediment extraction to reconstruct run and pool channel patches.

After three years of monitoring, the application of a reach-scale rehabilitation procedure did not positively influence fish assemblage structure. Unexpectedly, taxon richness and abundance decreased across unmodified and restored locations within six months of works completion, but had recovered to pre-restoration values by the last sampling surveys (summer 2005). Fish assemblage structure was related to significant changes in electrical conductivity, arising from low river discharge associated with the Millenium drought. Although restoration aimed to provide greater resources (e.g. deeper water) for fish under increased stress from low flows in sediment-disturbed reaches, the larger, landscape-scale effect of drought appeared to have masked any potential benefit of the restoration procedures.

The effect of the experimental restoration procedures on fish remained uncertain, especially if sediment extraction and woody debris replacement are both required to induce a response. Fish assemblage responses were hypothesised to depend on woody debris complexity (size and quantity) independent of modification to channel structure. To test this, a multiple Before-After, Control-Impact (MBACI) design approach was undertaken, adding small woody debris (SWD) to runs containing high and low amounts of large woody debris (LWD). High LWD locations had a higher taxa richness, but lower total abundances of fish. Adding SWD further
increased the abundances of several species known to use wood, but the effects differed within and among species. In low LWD locations, SWD additions altered assemblage composition, increasing abundances of two native fish: *Gadopsis marmoratus* and *Philypnodon grandiceps*. However, SWD additions had little effect on assemblage structure in channels containing high LWD. Higher abundances of juvenile *Gadopsis marmoratus* (< 123 mm total length) corresponded with added SWD in low LWD locations, while increases in abundances of adult *Galaxias olidus* (> 42 mm total length) occurred after SWD was added to high LWD locations. Abundances of adult *Philypnodon grandiceps* (> 50 mm total length) increased in response to additions of SWD in both high and low LWD locations, which indicated that the presence of SWD was more important than background amounts of LWD. Wood size-diversity is important for small native fish and it’s use should be considered further in future river restoration programs.

Reasons for the response of native fish to SWD are unclear, but the reaction of multiple species and sizes suggests different mechanisms were involved. Movement is recognised as an important driver of response to restoration, but other processes such as local reproduction and recruitment are seldom investigated. Thus, an experiment was used to investigate the use of rehabilitated reaches for reproduction, particularly the use of different channel types, substrate types and substrate complexity for oviposition. Only spawning of the native *Philypnodon grandiceps* was regularly detected, however the greatest number of spawning events was recorded from the pool edges in unmodified and rehabilitated reaches, respectively. The deeper restored runs also contained substantially more spawning events than unmodified, shallow runs. Added SWD was also used by *P. grandiceps* for oviposition.

Acts to restore sediment-disturbed rivers by reconstructing channels are thought to benefit native fish. Findings from this study support the use of channel modification, particularly sediment extraction and woody debris additions to assist native fish inhabiting sediment-disturbed channels. Results also indicated that detecting positive influences of restoration is difficult when other disturbances
affect the conditions under which resources are used and this may lead to prematurely negative judgements of restoration performance. Improvements in river restoration practices require new knowledge gained from monitoring the impact of management techniques designed to restore damaged ecosystems. This thesis has shown how habitat changes can lead to variable fish responses within short time frames, and that habitat restoration still requires further development, particularly for sand-slugged rivers.
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1. Chapter One: General Introduction

1.1. Importance of ecological restoration and need for monitoring and assessment

Over the past few millennia, river ecosystems have undergone significant changes in response to the development of human societies. Growth in human populations has depended upon resources attained from mining, forestry and intensive agricultural production across large areas of catchments and in response, ecological communities and ecosystems been altered due to these intensive developments. In particular, river ecosystems have endured many structural changes. With the advent of urbanisation, demands for water supply and disposal of waste materials, including pollutants, lasting changes to river ecosystems have extended from their headwater beginnings to their terminal drainages (Allan and Flecker 1993).

Biota associated with river ecosystems, particularly fish, have not fared well under the human stresses placed upon them (Allan and Flecker 1993, Dudgeon et al. 2006). Significant changes to biodiversity and structure of river ecosystems corresponding with environmental degradation, in conjunction with other types of human pressures (e.g. overfishing) has resulted in notable declines in abundance and changes in species distributions, including local extinctions (Burkhead 2012). Recognition of these deleterious impacts and changes has prompted conservation actions, and restoration attempts are now implemented to minimise risks of further losses of fish and other biota from local and larger areas.

Mitigating past environmental impacts is now common practice (Cowx and Welcomme 1998, Wohl et al. 2005). Conceptually, restoration is considered as the returning of a system to a previous ‘undamaged’ or ‘pristine’ state (Bradshaw 1996, Rutherfurd et al. 1999a, Cottingham et al. 2005), however, there is increasing appreciation that this is often difficult and unachievable, and perhaps a more reasonable approach (i.e. stakeholder agreed endpoint) is to rehabilitate ecosystems back towards something that once existed, but dissimilar to its pristine state (Boon 1998, Moerke et al. 2004, Wohl et al. 2005, Palmer et al. 2006). Other
terms (e.g. engineering) may also refer to similar undertakings (Mitsch and Jørgensen 2004), so terminology associated with ecosystem restoration may be used interchangeably within the literature, albeit to largely describe the same intention, which is undertaking action to change the present situation in order to facilitate ecological recovery (Downes et al. 2002, Moerke et al. 2004).

Whether it is referred to as restoration, rehabilitation, or something else, this proactive strategy which aims to rectify past ecosystem degradation to assist biodiversity conservation is being widely implemented by authorities and incorporated into mainstream management policy, such as environmental planning frameworks (Randolph 2003). Similar to other established fields of applied science (e.g. medicine), the most desirable outcomes are achieved using evidence based practice. The value of restoration as an ecological solution and its practical usefulness is underpinned by knowledge of the requirements needed to deliver successful outcomes. Therefore, research has an instrumental role in fostering the development of restoration ecology, considering that it is a relatively young field of scientific enquiry (Hobbs and Norton 1996, Young 2000, Lake 2001, Wohl et al. 2005).

A thorough scientific understanding of the ecological impact of restoration practices is essential for determining the success of such management efforts, but river restoration research is very limited in Australia (Brooks and Lake 2007, Mika et al. 2010). Indeed, restoration of river channels for ecological or conservation purposes is a relatively new concept for many Australian river managers and many current attempts are trials, simply adapting methods used in other locations (e.g. engineered structures, woody debris replacement), or experimenting with largely un-tested techniques (e.g. sediment or willow removal) or recipes (Hilderbrand et al. 2005). Of critical concern is that many of the activities and techniques that were developed and specified elsewhere in the world are often completed without an assessment of the influence on Australian river systems, which are often very different to those overseas (Thoms and Sheldon 2000). This means that intended outcomes for the restoration of Australian lotic environments could differ

Generally, there is little certainty that implemented restoration plans will prove effective (Perrow et al. 2008, Wheaton et al. 2008, Mika et al. 2010). Much of the uncertainty arises from the complex interactions between multiple environmental components (Perrow et al. 2008). For instance, as river degradation often comprises of multiple stressors (Lake et al. 2000, Ormerod et al. 2010), many different procedures are implemented to solve the problem (e.g. channel works, environmental flows, riparian revegetation, fishways, etc.). This means that identifying those procedure(s) that most influence the biota is tricky, especially when a single procedure (e.g. woody debris replacement) can invoke multiple effects that directly affect both habitat and biota (see below, section 1.4). There is limited scope to understand the influence of experimental, complex and often expensive management applications, because there is insufficient or non-existent ecological monitoring of restoration and subsequently, the influence of restoration works on biotic assemblages is often unknown (Lake 2001, Bash and Ryan 2002, Lepori et al. 2005, Wohl et al. 2005, Brooks and Lake 2007, Perrow et al. 2008, Palmer 2009, Howell et al. 2012).

Restoration implies that knowledge to improve habitat conditions or resources for biota is understood, so to ensure that an accurate and fair assessment of the effectiveness of a program can be measured (Bradshaw 1987). However, this is rarely the case because restoration is often undertaken with much uncertainty (Wohl et al. 2005, Perrow et al. 2008, Wheaton et al. 2008). Consequently, rushed efforts to restore ecosystems without understanding the mechanics is risky and may prove costly, especially if partial restoration is perceived as a complete failure of improving ecosystems in the public and political domains (Minns et al. 1996, Wheaton et al. 2008). Insufficient knowledge and the inherent complexity of ecological systems may lead to inappropriate pre-judgement of restoration, before methods of addressing issues are known.
There is much scope to learn from trial restoration procedures that apply current ecological and managerial knowledge. This thesis investigates different aspects of a novel restoration procedure and its influence on a predominantly small-bodied assemblage of freshwater fish in the Glenelg River, western Victoria. Specifically, this thesis aims to determine whether freshwater fish respond to artificial transformations of river structure to re-create natural habitat characteristics of reaches degraded by excessive sedimentation leading to the formation of ‘sand slugs’; a disturbance that is common to river ecosystems of south-eastern Australia (Erskine 1994, Bond and Lake 2005, Downes et al. 2006, Lind et al. 2009, Robson and Mitchell 2010). In addition, this study aims to contribute to the present understanding of restoring river structure and associated habitat for fish in sediment-disturbed rivers, which will ultimately assist managers by contributing towards the development of successful restoration strategies.

The following text introduces and outlines environmental changes, both disturbance and restoration, on river ecosystems and the influence of change and disturbance on shaping fish assemblage structure. Disturbances contributing to river degradation are discussed, with a particular focus on sedimentation, together with an examination of the links between disturbance, environmental variability and assemblage diversity. Key characteristics of woody debris as a contributor to river structure and its importance for fish in rivers are then discussed. Finally, the use of restoration techniques to alter river structure from Australia and overseas is reviewed, with a particular focus on the use of woody debris to influence fish assemblages.

1.2. River structure and degradation: anthropogenic disturbances on lowland rivers

Catchment development and its effect on the structure of river ecosystems are well described through the ecological literature (Welcomme 1985, Allan and Flecker 1993, Taniguchi et al. 2001, Dudgeon et al. 2006). A number of different impacts, many acting simultaneously and mutually, have changed natural river structure and
function (Gordon et al. 2004). Impacts such as the implementation of large water diversion and storage infrastructure, catchment-scale clearing of vegetation and river channel de-snagging has affected conditions and processes, including water quantity, quality and the flow regime (Poff et al. 1997), rates of erosion and sedimentation (Shields 2009) and transport of materials (e.g. supply of organic matter) (Montgomery et al. 2003b). Alteration of these components modifies river structure, leading to substantial changes in channel morphology and thus habitat patches along whole reaches (Bilby 1984, Andrus 2008). These changes are often apparent, particularly in the middle to lower reaches of large rivers, downstream of significant water diversion and storage infrastructure, and adjacent to broad areas of floodplain agricultural lands (Osborne et al. 1993).

Hydrological regulation is perhaps the most serious and widely known perturbation affecting river ecosystems globally (Poff et al. 1997). The widespread construction of reservoirs, weirs and mass-scale water extraction has significantly impacted rivers by reducing discharge, flood frequency, altered flow periodicity, degraded water quality, restricted the movement of biota and materials laterally and longitudinally. This has resulted in a number of direct and indirect effects on river structure and its function (Poff et al. 1997). River discharge is an important driver of lotic processes, particularly for the development and regulation of river channel morphology (Montgomery and Buffington 1997). The mobilisation, transport and deposition of sediment particles corresponds to a number of key structural attributes of rivers (e.g. channel depth and width, substrate composition) ranging from large-scale patterns of channel morphology to small-scale variation in hydraulic conditions (e.g. velocity distribution) (Montgomery and Buffington 1997, Poff et al. 1997, Gordon et al. 2004). High discharge events play a critical role by providing energy to mobilise and transport large volumes of sediment and organic matter particles, which help to form and maintain different habitat patches (see Figure 1.1, page 12), such as discrete channel units (e.g. pools, runs and riffles) and their smaller patches nested within (e.g. interstitial spaces among particles), thereby providing an assortment of spaces that can be occupied by biota (Frissell et al. 1986, Lake et al. 2000). Under significant water extraction (over allocated water
supply) and flood control, rivers have a diminished capacity to re-distribute sediment and thus re-organise bed forms within the channel (Gordon et al. 2004).

Vegetation is another vital component that contributes to the structure of river ecosystems (Kauffman et al. 1997, Pusey and Arthington 2003). Terrestrial vegetation interacts with geology and topography affecting surface and sub-surface waters and soils. Furthermore, the removal of terrestrial vegetation across landscapes is known to cause significant changes to hydrology and water quality, especially concentrations of suspended solids, nutrients and salinity (Brown et al. 2005). Overhanging and fallen vegetative material, even in advanced decay, along river banks and beds is an important natural feature of un-disturbed rivers, as its retention within channels aids in structuring bed forms (e.g. pool-riffle) and providing different resources that support biological communities. With the advent of agriculture and extensive clearing of vegetation, numerous changes to in-stream conditions have occurred, including reduced inputs of organic matter, reduced shading and cover and increases in stream temperature (Pusey and Arthington 2003, Davies 2010). In particular, large-scale vegetation removal destabilises soil structure, prompting widespread erosion and delivery of significant volumes of sediment into river channels (Prosser et al. 2001). Subsequently, the use of channels by aquatic organisms is also altered because some of the former river structure (e.g. deep pools) can be lost (Bisson et al. 1987, Jones et al. 1999, Pusey and Arthington 2003).

Modifications to improve catchment functionality for humans has resulted in a complex interaction of disturbances, river degradation and significant adjustments to biological communities. Hydrological regulation, in conjunction with deforestation of terrestrial vegetation, is responsible for some of the most severe morphological changes to rivers, such as shifts in channel structure resulting from re-adjustments to an altered balance between sediment storage and transport (Alexander and Hansen 1986, Shields et al. 1994, Bunn and Arthington 2002, Shields 2009). Sediment supplies to downstream reaches are severed by man-made dams located in headwater reaches (Gordon et al. 2004). Sediment transport in
downstream alluvial reaches is escalated when a diminished sediment supply combined with augmented discharge releases for hydroelectric or irrigation purposes. This leads to significant bed erosion and bank instability and consequently, channels become enlarged and over-widened with the upstream progression of uncontrolled erosion, obliterating natural bed forms (e.g. pool and riffle bed forms) and producing flat, featureless beds of fine sediment (Shields et al. 1994, Shields 2009). Similarly, excessive loads of fine sediments entering stream channels from degradation of the surrounding landscape (e.g. vegetation removal and erosion) results in significant channel degradation (Alexander and Hansen 1986, Shields 2009). Excessive supply of fine sediment particles combined with prolonged periods of depressed flow from significant water extraction, promotes sediment aggradation and the production of discrete slugs of sediment known as ‘sand slugs’ (Erskine 1994, Rutherford and Budahazy 1996, Bond and Lake 2005, Lind et al. 2009). As sand slugs slowly progress downstream, they fill interstitial spaces in river beds, bury the entire substratum, or in-fill whole channel units such as pools, resulting in the main channel becoming shallow, featureless and stabilised by encroaching vegetation (Erskine 1994, Erskine et al. 1999, Prosser et al. 2001, Bond and Lake 2005, Lind et al. 2009, Shields 2009). Sand slugs entering the mainstem from tributaries cause ‘tributary junction plugs’—discrete slugs that infill the mainstem channel, causing water to backup and forming large expanses of pooling water or wetlands (Lind et al. 2009). In this circumstance, tributary junction plugs may help off-set habitat patches that are degraded by the effect of sediment burying channel patches. This potential positive outcome can make decisions about restoring sand-slug reaches even more difficult (Lind et al. 2009).

Early river interventions to clear sediment build-up, improve the delivery of water resources and to control flooding, largely depended on substantial engineering of natural channels (Erskine 1994, Raborn and Schramm 2003, Gordon et al. 2004). Channels were often artificially enlarged and straightened (channelization) in attempts to increase average flow velocity, channel capacity and greater hydraulic predictability (Raborn and Schramm 2003, Gordon et al. 2004). However, channel degradation inadvertently developed in other reaches, which had subsequent
negative impacts on aquatic biota (Matthews 1998, Shields 2009). Within modified channels, increasing the flow volume redistributes sediment to deeper reaches downstream (Rutherfurd and Budahazy 1996). In addition, further erosion is initiated upstream through promoting channel incision (i.e. headway cutting) and the degradation of bed forms (e.g. pool and riffle) through downward cutting of the bed (Shields et al. 1994, Raborn and Schramm 2003, Shields 2009). Frequently, woody debris was also removed (de-snagging), with or without channelization, as this obstruction was believed to reduce channel capacity, and contribute significantly to an increased likelihood of flooding adjacent lands (Gippel et al. 1996b, Erskine and Webb 2003, Raborn and Schramm 2003). De-snagging was also used to nullify the influence of wood obstructions on the build-up of sediment, or navigation threats to vessels (Shields and Nunnally 1984, Erskine and Webb 2003). However, the removal of woody debris also releases substantial stores of sediment and organic matter, feeding back to exacerbate bed and bank erosion (Bilby 1984, Shields and Nunnally 1984, Erskine and Webb 2003).

1.3. Disturbance and diversity: ecological importance of environmental heterogeneity for fish assemblages

Disturbance is an important natural or human induced feature that has important functional roles in structuring river ecosystems, and may be defined as “any relatively discrete event in time that is characterized by a frequency, intensity, and severity outside a predictable range, and that disrupts ecosystem, community, or population structure and changes resources or the physical environment” (Resh et al. 1988, page 433). The force of environmental disturbance on structuring ecosystems by creating patchiness and its effects on the structure of biological communities are well known (Connell and Sousa 1983, Yount and Niemi 1990, Reice 1994, Mackey and Currie 2000, Haddad et al. 2008, Svensson et al. 2012). Rivers have undergone considerable changes from prolonged disturbance, largely arising from anthropogenic origin, which is recognised as an important contributor to global declines of freshwater biodiversity, particularly fish (Koehn and O’Connor 1990b, Allan and Flecker 1993).
Disturbance, characterised by its intensity, duration, frequency and extent (Petraitis et al. 1989, Lake 2000, Downes et al. 2002), is classified into three types: pulse, press (sensu Bender et al. 1984) and ramp (sensu Lake 2000). Pulse disturbances are short in duration, but vary in intensity, extent and frequency (Downes et al. 2002). Predictability of these disturbances ranges, from repeatable natural events (e.g. annual flooding) to stochastic (e.g. landslides) or rare anthropogenic induced events (e.g. chemical spills). For disturbances that are a natural feature of ecosystems, many organisms have acquired coping mechanisms by developing specific traits to aid resistance or resilience to natural disturbance regimes (Resh et al. 1988). Conversely, press and ramp disturbances are characteristically sustained over longer temporal periods. They are large, infrequent events but their occurrence can extend over large areas and have long lasting effects (Detenbeck et al. 1992). The size and persistence of press and ramp disturbances mean that they are often sustained for longer than the life-cycle of even the longest lived species, while their capacity to significantly alter environments can further influence ecological processes, such as increasing the distance from which re-colonisation can occur (Yount and Niemi 1990).

Knowledge that disturbance is linked to patterns of biotic diversity is well known (Lepori and Hjerdt 2006). Theory relating disturbance and riverine assemblage diversity has developed on two fronts (Lepori and Hjerdt 2006). First, evolutionary histories of organisms expressed through their current day traits closely match present environmental conditions (Southwood 1977, 1988, Townsend et al. 1997). Higher assemblage diversity often corresponds with greater environmental heterogeneity (Reice 1994, Tews et al. 2004). Different taxa exploit a variety of conditions and resources according to their inherent life history traits, such as body size, mobility, or physiological tolerance (Southwood 1977, 1988, Poff and Ward 1990, Townsend and Hildrew 1994, Townsend et al. 1997, Taylor and Warren 2001). Episodic periods of natural disturbance increase environmental heterogeneity by rearranging the environment, unlike anthropogenic disturbances, which tend to homogenise and simplify river structure (Lake et al. 2000, Poff et al. 2007).
Modifying the natural disturbance regime with additional disturbances from anthropogenic sources alters river structure, thereby changing or restricting resources and conditions. Consequently, habitat-driven exclusion of biota from assemblages can occur when environmental stressors act as filters that restrict the colonisation of biota with unsuitable life history traits (Poff and Ward 1990, Tonn et al. 1990, Poff 1997, Lake et al. 2007).

Disturbance also directly affects assemblage structure through the intermittent removal of organisms, which mediates ecological interactions among remaining taxa and provides opportunity for ‘new taxa’ to colonise or use patches (e.g. intermediate disturbance hypothesis; Connell 1978, dynamic-equilibrium hypothesis; Huston 1979). Disturbance in this sense can increase diversity if it promotes inclusion of more new species than those lost (Lepori and Hjerdt 2006), whereas in the absence of disturbance, interactions among individuals intensify and a few superior competitors tend to dominate. Conversely, frequent disturbance selects for fewer, environmentally tolerant colonists (Connell 1978). This concept explains assemblage diversity patterns for sessile taxa or vegetation competing for space, but it may be less useful for assemblages of mobile taxa, as movement may mitigate either the impact of disturbance itself, or the competitive effect between individuals once disturbance passes (Townsend 1989). Another important premise behind this competitive-exclusion pathway is that assemblages are stable, determined by the intensification of biotic interactions among individuals, and facilitated by periods of constant environmental conditions (Resh et al. 1988, Reice 1994). However, this is likely an unusual, or temporary situation in nature (Reice 1994), especially for rivers, where contemporary views regard river ecosystems as non-deterministic, open systems that are in continual states of flux, rather than internally regulated, homeostatic systems exhibiting equilibrium conditions (Resh et al. 1988, Palmer and Poff 1997, Ward et al. 2002). Subsequently, greater emphasis on the distribution of resources and conditions, including access to these areas, in which patch-dynamic models advocating environmental heterogeneity as an important mechanism underlying assemblage structure, has become more widely

Environmental heterogeneity has a central role in the response of assemblages to river ecosystem dynamics (Palmer and Poff 1997). Natural disturbance driving environmental heterogeneity has a defining role in the supply of resources and their distribution among biota, which the maintenance of assemblage diversity depends upon (Reice 1994, Ward et al. 2002). In particular, greater levels of environmental heterogeneity enables organisms to access more resources (Cooper et al. 1997, Palmer et al. 1997, Ward et al. 2002), which may partially explain greater biotic diversity corresponding to increases in environmental variability (Macarthur 1965, Palmer et al. 1997, Tews et al. 2004). Identifying which environmental factors contribute to influencing greater biotic diversity is challenging, considering environmental heterogeneity can mean many things (e.g. patterns, processes, structural versus functional effects) at different scales, and to different organisms (Li and Reynolds 1995, Cooper et al. 1997, Palmer and Poff 1997, Wiens 2000, Tews et al. 2004). Naturally, rivers are hierarchically structured, meaning that there are many different components contributing to habitat patches over a range of different scales (Frissell et al. 1986; Figure 1.1). Furthermore, specific components of environmental heterogeneity may lead to disproportional influences on biodiversity in supporting a larger numbers of species, such as keystone structures (Tews et al. 2004). Woody debris has been suggested as one such keystone structure for forest ecosystems (Harmon et al. 1986, Tews et al. 2004, Shields et al. 2006), and may perform a similar role in rivers given its contribution to river heterogeneity (Shields et al. 2006, Cordova et al. 2007).
Figure 1.1 Hierarchical arrangement of nested habitat patches in a river. Different patches illustrate the range and complexity of environmental heterogeneity occurring across different spatial scales in river ecosystems (from Frissell et al. 1986). As depicted within the reach system, components such as LWD can influence smaller and larger patches within lotic environments.

Environmental heterogeneity, as a component itself, can contribute to complex system behaviour that creates further patchiness by creating positive feedbacks that regulate the intensity or extent of further disturbance events (Reice 1994). Where more resources are available, a greater intensity of disturbance is thought to be necessary to invoke a change in assemblage diversity (Huston 1979, Ward and Tockner 2001). In this sense, environmental heterogeneity assists assemblage resistance to further disturbance, if the resources or conditions provided by environmental heterogeneity from the previous disturbance subsequently reduce future impacts of disturbance (Dutilleul and Legendre 1993). An important example of this is the role of refuges, such as pools, that can be produced by previous flood disturbances. If these newly created pools can hold water for the duration of the dry period, then organisms can use them to assist resistance and resilience to significant disturbances like drought (Sedell et al. 1990, Lake 2000, Fausch et al. 2002, Robson et al. 2008, in press).
For fish, environmental heterogeneity is an important contributor to patterns of assemblage structure, particularly over larger spatial scales such as reaches which are relevant to the management of catchment areas (Grossman 1982, Labbe and Fausch 2000, Oberdorff et al. 2001, Fausch et al. 2002). Freshwater fish have evolved a range of strategies to deal with the demands of various environmental conditions and change (e.g. reproductive traits, Winemiller and Rose 1992; behavioural traits, e.g. movement, Magoulick and Kobzina 2003). Freshwater fish have adapted well to and can recover quickly from short-term, pulse disturbances (Moerke et al. 2004), even catastrophic events that obliterate local populations (e.g. debris flows, Roghair et al. 2002). As evident from their life-history traits, fish may also exploit the characteristics of these events to assist or ensure population persistence (e.g. floods and spawning, year class strength, larvae dispersal, migration) (Matthews 1998, King et al. 2003). However, the effect of press-type disturbances is more serious and fish are generally more sensitive and less resilient to these impacts, especially, anthropogenic press disturbances that result in sustained changes to habitat structure, such as the suite of impacts associated with catchment clearing or hydrological regulation (Detenbeck et al. 1992, Roghair et al. 2002, Pusey and Arthington 2003).

From a review of disturbance and associations with fish assemblages, Detenbeck et al. (1992) concluded that disturbances which do not result in serious habitat change may result in fish returning sometime up to 6 years after the event. With significant and sustained habitat degradation, the recovery period becomes substantially longer, even decades (Yount and Niemi 1990, Detenbeck et al. 1992). Prolonged recovery of fish assemblages from significant changes to habitat structure is likely because degradation of rivers can affect multiple stages of a fish life history (Bunn and Arthington 2002), including eggs (Shelton and Pollock 1966, Acornley and Sear 1999), larvae (Scheidegger and Bain 1995, Humphries and Lake 2000), juveniles (Berkman and Rabeni 1987, Freeman et al. 2001) and adults (Ryan 1991, Nakamoto 1994), along with key processes underpinning each life-history stage such as migration (Pess et al. 2008) and reproduction (Burkhead and Jelks 2001, Hickford and Schiel 2011).
Most anthropogenic disturbances that tend to be detrimental to fish assemblages are those that cause sustained changes, which also simplified environmental heterogeneity. Therefore, it may be possible to reverse habitat degradation by managing disturbance or environmental heterogeneity directly. Modifying degraded environments to improve environmental heterogeneity for the purpose of facilitating greater biodiversity has been coined as the ‘field of dreams’ hypothesis by Palmer et al. (1997). The premise, ‘if you build it, they will come’, suggests that an intervention that aims to provide more or different habitat resources or conditions, leads to the subsequent colonisation by assorted biota and thus the development or persistence of more diverse local assemblages (Bond and Lake 2005). Knowing what to build, or moreover, what’s required to counteract the influence of degradation in order to facilitate biodiversity (i.e. the plan), is central to the field of dreams hypothesis. However, knowledge of how the previous environmental structure facilitated biodiversity, or how disturbance has altered this structure, and whether a source of colonists can move unimpeded, may only be partially known or controlled. Subsequently, restoration proceeds by replicating the structure of other reaches or substituting other successful restoration methods from elsewhere (Hilderbrand et al. 2005). To artificially increase the heterogeneity of degraded river environments, woody debris (or re-snagging) is commonly used in restoration programs around the globe. Woody debris has characteristics that enable it to change river heterogeneity and affect resources and river conditions over a wide range of spatial scales from micro ($10^{-1}$ m) to macro ($10^{3}$ m), which can have significant influences on fish distributions (Crook and Robertson 1999). The following section describes the defining characteristics of woody debris that create environmental heterogeneity, which provides more resources and favourable living conditions for fish.
1.4. Woody debris: key contributor to river heterogeneity and fish habitat

1.4.1. Characteristics and functions of woody debris for structuring river environments

Woody debris is classified as wood pieces shed from trees, but may include whole trees (Gurnell et al. 2002). It is supplied to river channels (Figure 1.2) from surrounding riparian zones via a number of natural (e.g. storms, landslides, fire, flood, disease, Harmon et al. 1986, Swanston 1991, Berg et al. 1998) and anthropogenic processes (e.g. forestry practices, Bisson et al. 1987, Gomi et al. 2001). These processes and others (e.g. weathering) also interact to create diversity in types and forms of woody debris, ranging in size and complexity, from solid, large tree trunks without branches, to whole trees with intact branches and aggregations of woody debris pieces (Wallace and Benke 1984, Abbe and Montgomery 1996, Baillie et al. 1999, Wallace et al. 1999).

![Figure 1.2 Sources and fates of woody debris and leaf litter within river channels (from Gregory 1992).](image)

Size is a commonly used descriptor and discriminator of woody debris pieces, classified according to two categories: large woody debris (LWD) and small woody
debris (SWD) (Plate 1.1). Large woody debris is classified as pieces > 0.1 m in
diameter and typically consists of large trunks or branches (Keller and Swanson
1979, Ward and Aumen 1986, Gippel et al. 1996a). Alternatively, small or fine
woody debris generally consists of small trunks, branches or fragmented pieces
ranging from 0.1 to 0.01 m diameter (Triska 1984, Wallace and Benke 1984, Culp et
al. 1996, Baillie et al. 1999). To separate SWD material from finer grain organic
matter particles, such as leaf and twig matter, some researchers have considered
SWD to contain a lower size limit of 0.01 m diameter (Triska and Cromack 1980,
Baillie et al. 1999, Kraft et al. 2002). Classification of woody debris according to
piece size partly relates to the recognition that large pieces (> 0.1 m in diameter and
1 m in length) can influence macro-scale hydraulic and geomorphic processes and
therefore change habitat structure (Abbe and Montgomery 1996, Gippel et al.
1996b). Larger pieces can become lodged within river channels, effectively storing
sediment and organic matter particles, and causing flow to deviate to reshape the
channel (Montgomery et al. 2003a). For alluvial rivers, particularly those with sand
beds, the interaction between wood, sediment and flow can affect the dynamics of
scour and deposition processes and can increase habitat heterogeneity by creating
discrete channel units such as pools and runs (Bilby 1984, Gippel 1995, Beechie and

Supply of the largest LWD pieces is critical to habitat formation in large alluvial
rivers because the potential for LWD pieces to shape channels depends upon both
channel size and stream power (Abbe and Montgomery 1996, Collins et al. 2002,
Gurnell et al. 2002, Montgomery et al. 2003a). Notably, the average LWD piece size
(i.e. diameter, length and volume) increases as rivers enlarge because wood stability
decreases with increasing stream power (Bilby and Ward 1989, Gurnell et al. 2002,
Montgomery et al. 2003a). Movement of unstable wood by flow still assists in
further channel development through the aggregation of multiple pieces to form
larger wood super-structures, such as logjams and rafts. Even in large channels, the
aggregation of many pieces of wood forming logjams greatly affects channel
structure by increasing both the size and number of larger pools compared to pools
formed by other features (Abbe and Montgomery 1996). Piece size is an important
functional attribute contributing to the formation of these complex woody debris patches and the associated changes to river structure. Development of the jam depends on the aggregation of smaller, unstable wood pieces, while the strength of the structure relies upon the key members, being the largest, most stable pieces of wood that provide anchorage (Nakamura and Swanson 1993, Abbe and Montgomery 1996).

Plate 1.1 Pieces of woody debris located in the Glenelg River. Clockwise from top left: (a) buried large woody debris; (b) large and small woody debris jam; (c) small woody debris in foreground, large woody debris in background on a dry streambed and (d) complex branching of small woody debris pieces.

Apart from shaping channel morphology, woody debris itself offers various habitat features arising from diversity in size. Variations in piece size are governed by natural patterns and processes, including the composition of riparian vegetation (e.g. tree species and age), floods and the transport and mechanical degradation of pieces, as well as effects of biological decomposition (Gurnell et al. 2002). Together, these factors shape individual pieces providing characteristics which are exploited by different organisms. For example, newly fallen large wood pieces contain smaller
pieces of bark, fine branches and leaves attached to larger branches or trunks, which provides a complex substrate that is used by a wide variety of organisms ranging from bacteria to birds (Harmon et al. 1986). In time, smaller parts break off and are redistributed, often becoming entangled in woody debris jams and dams (Gurnell et al. 2002). These small wood fragments assist to increase wood surface area for the colonisation of biofilms (Scholz and Boon 1993, Tank and Webster 1998), and effectively trap finer particles of organic matter (e.g. leaves) to provide resource-rich patches for secondary consumers (Bilby and Likens 1980, Smock et al. 1989). Additionally, older large pieces (e.g. trunks) provide alternative physical characteristics to smaller pieces, such as developing large surface splits, cracks or hollows which are readily used by larger aquatic organisms including macroinvertebrates and fish (Jackson 1978b, O’Connor 1991). Hence, woody debris size has a complex influence on the structure of river environments, generating a hierarchy of environmental patchiness, as it spatially encompasses a number of different scales (Pringle et al. 1988).

Along with size, the amount of woody debris present contributes greatly to river heterogeneity. Wood quantity is expressed as a volume per channel area, reflecting the contribution of larger debris pieces to river geomorphology (Gippel 1995, Gippel et al. 1996a). Typically, loadings of LWD are highly variable (Wallace and Benke 1984, Harmon et al. 1986, Montgomery et al. 2003a, Webb and Erskine 2005), particularly for Australian rivers, where wood loadings can vary up to several orders of magnitude between reaches, rivers and catchments (Marsh et al. 1999, Treadwell 1999, Lester et al. 2006). Wood load variability is attributed to several factors and processes, including the influence of extreme events on wood supply (e.g. landslides, Andrus et al. 1988), rivers traversing ‘wood deprived,’ landscapes such as deserts (Treadwell 1999), or ‘wood rich’ tropical forests, where higher rainfall and temperatures increase the breakdown of organic material (Treadwell 1999, Pusey and Arthington 2003). Previous anthropogenic catchment changes have directly lowered woody debris loads by altering the supply (e.g. vegetation clearing from forestry, agriculture), transport (hydrological regulation) and direct removal of
woody debris pieces by de-snagging (Erskine and Webb 2003, Montgomery et al. 2003a).

The quantity of LWD supplied to channels has a significant effect on bed dynamics. Wood naturally accumulates in undisturbed rivers, and thus greater amounts of LWD are a major determinant of channel heterogeneity (Triska 1984). For example, increasing the number (Richmond and Fausch 1995) and volume (Beechie and Sibley 1997) of LWD pieces is associated with pool formation. Greater amounts of LWD, either as number or size of pieces, relate to increasing volume and area of pools (Bisson et al. 1987, Bilby and Ward 1989), while higher numbers of LWD pieces per reach is also linked to increased pool frequency per reach (Richmond and Fausch 1995, Montgomery et al. 2003a) and thus closer pool spacing (Montgomery et al. 1995, Beechie and Sibley 1997). LWD has its greatest effects on pool formation with intermediate to high river slopes, where the ratio of channel to wood size is smaller (Gurnell et al. 2002, Montgomery et al. 2003a). However, in widened, shallow streams with low gradients and where the amount of woody debris is notably less (Martin 2001, Cordova et al. 2007), woody debris can still force pool-riffle bed forms (Montgomery et al. 2003a). The relationship between LWD quantity and channel heterogeneity can be obscured, because only a small fraction of the LWD volume present in rivers has a specific function of developing pools (Andrus et al. 1988, Richmond and Fausch 1995, Abbe and Montgomery 1996, Berg et al. 1998). Key attributes of individual pieces, such as stability or particular orientation to flow, are important predictors of greater pool formation (Richmond and Fausch 1995, Gippel et al. 1996a). Other factors besides LWD quantity, including the hydrological and sediment regimes are critical to the channel formation over reaches in large rivers (Gurnell et al. 2002). These factors deserve important consideration when examining the potential for LWD to increase channel heterogeneity, particularly for restoration purposes (Richmond and Fausch 1995, Cordova et al. 2007).

Valuing the amount of woody debris strictly based only on wood volumes, could provide a limited indication of the importance of woody debris quantity to habitat heterogeneity and ecosystem function. Stream surveys examining a range of debris
sizes have shown that the highest frequencies of woody debris pieces are skewed towards smaller size pieces (Flebbe and Dolloff 1995, Wallace et al. 2000, Millington and Sear 2007). These classes of debris are rarely, if ever, accounted for in surveys of large debris, yet they may be ecologically meaningful across river landscapes. Bilby and Ward (1991) noted that high frequencies of SWD characterised streams draining old growth forests in Washington (USA). Evans et al. (1993) characterised woody debris in New Zealand streams from differing forest types (ancient, 120 and 10 years old). They noted that most SWD was found in the 120 year old forest and ancient forests compared to the 10 year old forest. Flebbe and Dolloff (1995) also observed that SWD was relatively common in all southern Appalachian streams (USA). These studies suggest that smaller class sizes of woody debris may be numerically important, and thus their ecological value should receive closer attention.

1.4.2. Relationships between fish and woody debris

In rivers, patches of wood are among the richest in fish diversity (Matthews 1998). Crook and Robertson (1999) summarised the relationships between woody debris and fish from research largely conducted in North American upland streams, indicating that the main functions of wood in providing shelter and velocity refuge, predator/prey interactions, foraging locations and possible further roles in navigation and spatial orientation. Woody debris is also directly used by fish during reproduction, either as a substrate for oviposition (Jackson 1978b) or as shelter (e.g. cover) in preparation for spawning (Merz 2001). Cover (viz. as a place of concealment) is widely stated as the reason for fish attraction to woody debris during the day (Inoue and Nakano 1998, Matthews 1998, Crook and Robertson 1999, Taniguchi et al. 2001). Cover is important because biotic interactions are lowered between fishes when places are available for fish to seek shelter (Sundbaum and Näslund 1998). Pieces of woody debris may also act as important refuge from natural disturbances (Harvey et al. 1999, Dolloff and Warren 2003, Bond and Lake 2005) by sheltering fish from high water velocities (McMahon and Hartman 1989, Shirvell 1990, Harvey et al. 1999) or during times of low water levels.
in periods of drought (Bond and Lake 2005). Upland reaches of rivers have featured prominently as locations for studying fish-wood relationships (Crook and Robertson 1999), but their channel structure, debris dynamics and fish assemblages fundamentally differ to lowland reaches, indicating that further studies of the influence of wood in lowland reaches is warranted (Lobb and Orth 1991, Crook and Robertson 1999, Zalewski et al. 2003).

Relationships between woody debris and fishes in river ecosystems are complex and may vary considerably. Fish can respond to the environment that has been shaped by the presence of woody debris (e.g. channel structure, depth, velocity, substrate composition) in addition to the characteristics of the woody debris itself. Furthermore, the response of fish to woody distributions may differ according to the level of scale (Schmetterling and Pierce 1999, Crook et al. 2001). For instance, in catchments where forestry operations occur, rivers have reduced large woody debris loads and fewer pools (Bilby and Ward 1991, McHenry et al. 1998, Hauer et al. 1999), and different fish assemblages than rivers in nearby, non-logged catchments (Flebbe and Dolloff 1995, Connolly and Hall 1999, Jones et al. 1999).

However, within streams that have been historically logged, there are contrasting arguments regarding the influence of woody debris and associated channel units on local fish distributions. For example, some authors consider that at the reach-scale, woody debris itself may explain fish abundances better than the presence of pools (Inoue and Nakano 1998, Neumann and Wildman 2002). Conversely, others have indicated that the presence of pools is much strongly related to fish distributions than the presence of wood (Berg et al. 1998, Harvey 1998). Furthermore, research has shown that pools with more wood support more fish than pools with less wood or none (Flebbe 1999, Wright and Flecker 2004), while pools with and without wood had higher fish occupancy rates than riffles either with or without wood (Flebbe 1999). Spatial relationships are further complicated by periodic dependence at critical life-history periods (e.g. reproduction, Jackson 1978b) or during disturbance events (Quinn and Peterson 1996, Solazzi et al. 2000, Bond and Lake 2005).
Variation in fish associations with wood likely reflects both the importance of the characteristics of the woody debris itself, as well as woody debris influence on the river environment (Monzyk et al. 1997). Since most studies have focused on LWD, disentangling the association of fish directly attributed to the influence of wood (e.g. source of cover) from indirect association with habitat patches it forms (e.g. pools) remains an important area of understanding of habitat function. The effects of SWD on river fish assemblages have not received as much attention as LWD. One explanation is that the small size of SWD pieces is unlikely to affect river channel shape and therefore perhaps unlikely to alter fish habitat directly, even though smaller pieces could provide some of the same direct functions as larger pieces, such as cover. These aspects may be particularly important for small bodied species or juveniles that might be displaced from areas with LWD that are harbouring larger predatory species. Fish have been noted at sites with SWD in rivers (Koehn 1986, Culp et al. 1996, Monzyk et al. 1997, Bond and Lake 2003a), although it is unclear whether the SWD or other factors and their interaction correspond to fish distributions. The natural accumulation of SWD in jams and dams adds more debris surface area at a minimum, as well as the possibility for additional spatial complexity and food resources (i.e. macroinvertebrate assemblages).

1.5. River restoration, woody debris and fish response

1.5.1. Use of woody debris in river restoration

Habitat modification in rivers through the use of in-stream structures to increase local fish abundances has been actively practised in North America since the 1880’s (Thompson and Stull 2002). Evaluations conducted during the mid 20th century indicated that stream sections treated with added structures prompted changes in channel structure and produced notably higher abundances of fish (White 1996, Thompson and Stull 2002). Initial positive responses have encouraged the continued use of in-stream structures to improve river structure and conditions in an attempt to stem the continuing decline of commercially valued salmonid populations (House and Boehne 1985, 1986, Crispin et al. 1993, Slaney et al. 1994, White 1996, Thompson and Stull 2002). More recently, there has been a shift from
enhancing habitat for single, commercially valuable species to an interest in determining the wider effects of management procedures at the fish assemblage level (Roni et al. 2005). However, very few restoration projects are monitored, and so there has been little published, empirical evidence of fish assemblage-level responses to restoration of LWD (Lepori et al. 2005) to support the ‘field of dreams’ hypothesis (Bond and Lake 2005).

As the function of woody debris is widely recognised for river ecosystems, replacement of large wood pieces is often incorporated in river restoration programs to increase habitat heterogeneity after the impacts of channelization, riparian deforestation or sediment aggregation (Roni et al. 2005). To mimic natural river function, LWD is typically introduced in three forms: 1) debris dams or log jams, 2) debris deflectors and 3) shelter/cover debris arrangements (Cowx and Welcomme 1998). Channel alterations have become a priority to improve fish habitat in many rivers, which LWD dams, engineered jams or deflectors are primarily used to increase pool depth, size and frequency, sediment and organic matter retention, or alter substrate composition (House and Boehne 1986, Crispin et al. 1993, Cowx and Welcomme 1998, Larson et al. 2001, Roni et al. 2005). Although successful, effectiveness of LWD channel alterations is reported to be variable and contingent upon larger-scale hydrological and sediment regimes (Shields et al. 1998, Brooks et al. 2004, Shields et al. 2006), which without due consideration can lead to catastrophic failure in unstable channels (Frissell and Nawa 1992, Kondolf 2000). Alternatively, inserting LWD can affect small-scale habitat heterogeneity and fish distributions through creating micro-patches of different amounts of velocity or shade or by directly concealing fish through the provision of cover (Nagayama et al. 2008, 2009, 2012).

Unlike LWD, smaller woody debris does not have the same influence on channel forms and thus are seldom replaced for restoration purposes. The few restoration applications of SWD that have been published include the installation of bundles of brush to trap and store fine sediments and to provide a substrate for plant colonisation on unstable banks (Cowx and Welcomme 1998, Brown 2003). In fact,
SWD can often be perceived as a detriment to existing habitat and some stream improvement strategies have involved SWD removal or ‘de-brushing’ (Roni et al. 2005). Different perceptions of the ecological role and importance of small wood for management may reflect a poor understanding and underestimation of its importance. There is considerable potential for SWD to provide similar functions as LWD (e.g. cover for small fish) under restoration contexts, but this has not been widely investigated. SWD could also be extremely valuable and useful as supplementary woody material where availability of LWD is severely limited, especially if the amount of wood replaced directly corresponds to the size of the fish response to restoration (Roni and Quinn 2001a).

Replacing LWD often successfully alters river channel structure and increases habitat heterogeneity, however, the response of fish varies among and within the species present (Roni and Quinn 2001a, Bond and Lake 2005). In many cases, responses to constructed habitat patches can be significant and rapid, with large increases in species, abundance or biomass within short periods of months to a few years (Table 1.1). Most studies have examined the response of salmonids (Salmo sp., Onchorynchus sp., Salvelinus fontinalis [Mitchill]) to in-stream structures, and have reported more individuals associated with increased pool frequency or size, river depth, reduced velocity, or the presence of suitable substrate or cover after the addition of in-stream structures (House and Boehne 1985, 1986, Slaney et al. 1994, Cederholm et al. 1997, Van Zyll Dejong et al. 1997, Solazzi et al. 2000, Roni and Quinn 2001a, Lehane et al. 2002, Zika and Peter 2002). Salmonid responses to restoration occur seasonally, as more fish may use large woody debris for shelter during periods of greater natural mortality, such as high discharge events during winter (Roni and Quinn 2001a, Nagayama et al. 2009, Antón et al. 2011, Nagayama et al. 2012). Under these circumstances, artificially increasing the load of large woody debris may enhance juvenile survival and result in reaches sustaining more fish (Solazzi et al. 2000, Johnson et al. 2005). For trout (Salmo trutta [Linnaeus] and Onchorynchus sp.), larger fish often represent significant increases in the abundance or biomass as a response to the addition of LWD (Gowan and Fausch 1996, Zika and Peter 2002, Antón et al. 2011), by moving or migrating from adjacent
river reaches (Gowan and Fausch 1996, Antón et al. 2011). Use of restored locations with added LWD may reflect suitable habitat for larger individuals, such as adequate cover (Rosi-Marshall et al. 2006) or an environment conducive to reproduction (House and Boehne 1985, Moerke and Lamberti 2003, Antón et al. 2011). Smaller, younger trout may use restored locations in the absence of larger individuals, suggesting that larger fish may exclude smaller individuals (Rosi-Marshall et al. 2006), or simply that smaller cohorts are taking advantage of sites where fewer larger individuals are present (Lehane et al. 2002).

Less is known about assemblage level responses, especially for non-salmonid species (White 1996), and the premise that an increase in local biodiversity will occur through use of in-stream structures remains relatively uncertain. There is some suggestion that the types of structures developed for salmonids have little effect on other species (Roni et al. 2005, Thompson 2006), which has contributed to debate about the biological effectiveness of different in-stream methods (Thompson 2006). For example, some studies have reported that despite significant changes to channel structure and habitat, there is little or no overall response from fish assemblages to in-stream structures (Pretty et al. 2003, Raborn and Schramm 2003, Lepori et al. 2005, Schwartz and Herricks 2007). These studies, however, used structures consisting of rock and not woody debris, which may elicit a response from fewer species (Pander and Geist 2010). Other studies using woody debris as a material have shown greater responses from multiple species colonising treatment reaches, despite insignificant changes in channel structure. For instance, Shields et al. (1998) showed an increase of up to eight species within two years after restoration in a North American warm-water stream, despite only small changes to the habitat itself. Wright and Flecker (2004) added woody debris to existing pools in a tropical Venezuelan stream and found that up to 13 fish species responded within weeks. Similarly, Hrodey and Sutton (2008) added half-log cover to streams lacking LWD and found species richness and abundance increased at locations where little amounts of LWD existed.
Studies where LWD has been replaced in Australian rivers have also reported that larger species including Murray cod (*Maccullochella peeli peeli* [Mitchell]), trout cod (*Maccullochella maquariensis* [Cuvier]) and golden perch (*Macquaria ambigua* [Richarson]) respond to placed woody debris structures in the Murray River (Nicol et al. 2004) and two-spined blackfish (*Gadopsis bispinopsus* [Sanger]) to rock and woody debris placements in the Ovens River (Koehn 1987, 2005). Smaller native species may also respond to whole reach modifications using LWD. Brooks et al. (2004) indicated initial increases of Cox’s gudgeon (*Gobiomorphous coxeii* [Krefft]) and Australian smelt (*Retropinna semoni* [Weber]) to the replacement of logs in the Williams River, however a follow up study showed that the rehabilitation effect had diminished (Brooks et al. 2006). Similar works incorporating woody debris structures in both pools and riffles in the nearby Hunter River indicated a patch-specific effect of restoration after 16 months, with increased abundances of native *Retropinna semoni* and alien mosquito fish (*Gambusia holbrooki* [Girard]) in riffles, but no post-treatment changes in fish assemblage structure in pools (Howell et al. 2012). Bond and Lake (2005) found that river blackfish (*Gadopsis marmoratus* [Richarson]), mountain galaxias (*Galaxias olidus* [Günther]) and southern pygmy perch (*Nannoperca australis* [Günther]) were temporarily associated with placed LWD structures in Creightons and Castle Creeks before a larger ‘drought effect’ dried the stream, ending the experiment.

1.5.2. Restoration to address the impacts of sand-slugs

Many woody debris re-introduction programs demonstrating fish responses have been undertaken in upland reaches or larger lowland rivers, which contain fish taxa that are relatively large in size and highly mobile (e.g. salmonids, Gowan and Fausch 1996). In these perennial systems, flow is an important determinant of fish response (Shields et al. 1998) as it contributes to both changes in the stream environment (increased depth) and the facilitation of fish movement and colonisation of restored locations (Shields et al. 1998). However, much less is known about the potential response of fish to restoration in hydrological variable systems, especially rivers with sand-slugs, where the risk of bed exposure to desiccation...
increases during droughts (Bond and Lake 2005). Adding LWD to these sand-bed rivers assists in developing channel structure, such as pools (Wallerstein and Thorne 2004), although the extent that LWD influences channel morphology also depends on significant river discharge events like floods or ‘flushing flows’ from reservoirs (Kondolf and Wilcock 1996, Gordon et al. 2004). Where discharge is restricted (e.g. drought), scour-pools may not develop to expectation (Bond and Lake 2005) and therefore other techniques (e.g. sediment extraction) may be required to create deeper water in the shorter-term.

Some freshwater fish have adapted to harsh periods of low flow (e.g. aestivation, small body size), and habitat patches, such as pools that confer organism resistance to low flow disturbance, are important at the assemblage level, although, so too are components that reduce interactions among gathering fishes in these places. Some studies suggest that adding large wood could evoke fish responses for cover as river discharge is declining (Bond and Lake 2005) or when water levels are low (Wright and Flecker 2004).

Other techniques such as sediment extraction are likely to be an important tool in the construction of critical channel refuge (e.g. pools) in rivers with sand-slugs, but the consequence of side-effects (e.g. higher suspended sediment concentrations, deposition of fine sediment, noise and direct mortality of fish) from excavation can be detrimental (Padmalal et al. 2008) or unknown (Hall 1988, Harvey and Lisle 1998, Rutherford et al. 1999b). Fish can be sensitive to increased deposition of fine sediments, particularly during reproduction or early life history phases (Ryan 1991, Wood and Armitage 1997). However, several studies have reported positive responses from fish where sediment extraction has been used. Solazzi et al. (2000) indicated that excavated alcoves provided deep water, winter refuge for Coho salmon (*Onchorhyncus kisutch* [Walbaum]) but warned excavation may have serious impacts on the landscape and should only be reserved for occasions when habitat degradation occurs in sediment-aggrading reaches. Rempel and Church (2009) looked at the response of a fish assemblage to gravel mining over a single bar in the large alluvial Fraser River in Canada, and found that fish assemblage differences
were inconclusive, suggesting that the effect of mining may have fallen within the range of flooding as a natural disturbance. Furthermore, restricted use of excavation to create holes functioning as ‘sediment traps’ above restored locations can reduce or halt sediment depositing into restored reaches. This can lead to an improvement in reach function (Hansen et al. 1983) and increase reproductive use by fish (Avery 1996). Creating pools by removing sediment directly, rather than relying on river discharge to scour the bed, is likely to benefit fishes during critical summer periods when refuge is needed (Bond and Lake, 2005). Therefore, any potential disturbance attributed to sediment extraction, could be outweighed by the requirement of low flow refuge during summer. It is also unclear whether the provision of additional resources (e.g. shelter, food), from the replacement of woody debris may offset any potential disturbance from the sediment extraction procedure.

1.6. Thesis aims and structure

This thesis examines the responses of fish assemblages to experimental restoration procedures consisting of sediment extraction and reintroduction of woody debris in sediment disturbed reaches of the Glenelg River, Victoria, Australia. This introductory chapter has discussed the known role of woody debris in river ecosystems, the relationships with fish, and how it is used in restoration to address challenges of repairing environmental degradation. Chapter Two introduces the Glenelg River and its catchment. Catchment climate, geology, landuse and, catchment alterations are described, as well as changes to the river channel, (e.g. sedimentation) and hydrology are discussed. Fishes inhabiting the catchment and an outline of the restoration procedures are also presented.

Chapters Three, Four and Five investigate different aspects of river restoration techniques contributing to changes in fish assemblage structure and all three chapters have been accepted for peer-reviewed publication. These three chapters address three main questions and associated hypotheses (Table 1.2):
1. Does the implementation of a reach-scale river rehabilitation program alter fish assemblage structure?

2. Does the quantity and size of woody debris influence fish assemblage structure without restoration altering channel morphology?

3. Does rehabilitation consisting of sediment extraction with woody debris replacement influence fish spawning?

In Chapter Three (Howson et al. 2009), the implementation of a reach-scale rehabilitation procedure that alters fish assemblage structure is addressed. Prior to this study, there was little empirical evidence to suggest that alteration of habitat structure in a sand-slugged Australian lowland river would yield positive effects on fish assemblages, in accordance with the field of dreams hypothesis. This Chapter investigated whether sediment extraction combined with LWD replacement to improve river structure would lead to a positive change in fish assemblages (e.g. increased species diversity and abundances). Larger fish species (e.g. Gadopsis marmoratus) were expected to directly benefit from deeper water and added cover, although the influence on several small-sized species was largely unknown, but was assumed to be positive. Therefore, it was expected that the fish assemblage composition would shift to a more diverse assemblage consisting of a greater number of species and higher abundance after restoration. Seasonal variation of the restoration response is explored, particularly hydrological patterns and its effect on water quality, which was an independent factor that may determine fish use of reaches, but it was anticipated that water quality would not relate to location or fish assemblage structure after the completion of restoration.

In Chapter Four (Howson et al. 2012), woody debris becomes a focus for its effect on fish assemblage structure in isolation from altering channel morphology. Published scientific literature has indicated a number of confounding factors that can contribute to restoration outcomes, presenting difficulties in identifying the influence of restoration procedures. Responses of fish to LWD could be due to
effects of large wood on channel morphology, flow patterns or the wood itself acting as cover. The process of restoration, particularly sediment extraction, also creates disturbance and it is unclear as to whether a potential assemblage response reflects changes to habitat or disturbance. This component investigates if the addition of woody debris alone without altering shallow channels affects fish assemblage structure. Secondly, this component examines if fish assemblages vary according to the size and quantity of woody debris. The amount of LWD in rivers is highly variable with assumptions that ‘more is better.’ In contrast, little is known about the importance of the replacement of SWD in lowland rivers, especially the influence of this material when differing amounts of larger wood are present. It was expected that SWD would offer some of the functions provided by LWD (e.g. cover), particularly for small-bodied fish.

Provision of woody debris, together with sediment removal to provide areas of increased river depth (i.e. pools) is assumed to benefit spawning fish, particularly benthic species in sand-slugged rivers. Recruitment of juveniles from local spawning events may provide another mechanism structuring fish assemblages, as opposed to movement and migration into rehabilitation reaches from other areas. However, the effects of river restoration on ecological processes are rarely investigated. These aspects have been examined in Chapter Five (Howson et al. 2010), which describes the distribution of spawning events between runs and pools, within pools and between control (no change) and restored reaches (sediment removed and woody debris added). It was anticipated that spawning frequency will be greater at restored locations compared to control reaches and fish would use re-introduced woody debris (LWD and SWD) for oviposition.

The final chapter (Chapter Six) provides a general discussion that synthesises the results of each data chapter and places the results within the context of the current literature. Implications of this research for the field of restoration ecology and in relation to other sand-slugged rivers are also discussed.
Table 1.1 Published studies since 1996 (seven years before and after data collection in this thesis) that have examined the response of fish species and assemblages to manipulations of LWD in lotic ecosystems.

<table>
<thead>
<tr>
<th>Study</th>
<th>Location</th>
<th>Purpose of restoration</th>
<th>Scale of restoration</th>
<th>Reach type</th>
<th>Woody treatment used</th>
<th>Monitoring employed/design</th>
<th>Monitoring time period</th>
<th>Habitat effect</th>
<th>Fish response</th>
<th>Comments</th>
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</thead>
<tbody>
<tr>
<td>Gowan and Fauch (1996)</td>
<td>Colorado Ck Walton Ck Cache la Poudre River Jack Ck Little Beaver Ck St. Vrain Ck</td>
<td>Experimental restoration undertaken to quantify influence on trout life-history processes.</td>
<td>250m modified section for 5 streams, 1 x 375m modified section for St. Vrain Ck</td>
<td>Upland reach</td>
<td>10 large woody debris, log drop structures placed across channel to form weirs, 25 structures were used in St. Vrain Ck</td>
<td>Before and After sampling: Before period: 2 times, After period: 6 times. A single treatment reach was paired to a single control, either downstream or upstream, except St. Vrain Ck, which had control sites upstream and downstream</td>
<td>Before: summer, 1987-1988 After: summer, 1988-1994</td>
<td>Logs create scour pools downstream and dammed pools upstream. Total cover was greater in restored reaches. Habitat Changes above took between 1-2 years to take place.</td>
<td>Increased adult trout abundances and biomass in multiple streams, but not juvenile trout. Immigration within the first three years accounted for increases in adult trout abundance. Abundance of small Catostomus catostomus were observed to increase over the first two years, but then decline to pre-treatment values. Large Catostomus catostomus declined in both treatment and control reaches. Because juveniles did not increase with treatments, they reasoned other factors were underpinning the increase abundance of adults.</td>
<td>LWD structures.</td>
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<td>Fausch et al. (1996)</td>
<td>Walton Ck Cache la Poudre River Jack Ck</td>
<td>Increase the size and frequency of pools and amount of LWD cover during winter to influence the number of overwintering salmonids.</td>
<td>500 m engineered reach: 133 structures containing 300 LWD logs in total. 5 different structures used, including single logs and log jams to increase pools as well as provide cover for fish. Loggers reach 60 tree beside stream were felled and attached to their stumps using cable</td>
<td>Upland reach</td>
<td>Before and After sampling: Before period: 9 times, After period: 9 times. Two 500m treatment reaches, each 500m had different treatment and were paired to a single 500m reach that served as a control, located immediately upstream of both treatment sites. Sites were placed close together on one stream to keep physical and biological conditions similar. No reference reach</td>
<td>Before period: winter, spring and autumn 1988-1991, After period winter, spring and autumn 1991-1994</td>
<td>LWD amount increased in both types of treated sites. Winter storms added LWD to all reaches, resulting in an increase of pool surface area in treatment reaches, but a decrease in pool area was observed in the control reach. Improved habitat conditions retained fish over winter.</td>
<td>Winter Onchorynchus kisutch abundance increased in both treatment reaches, but not for the control reach. Summer and autumn Onchorynchus kisutch abundance did not differ among treatment and control reaches. After period abundances of age 0+ Onchorynchus mykiss increased in one treatment reach and control reach, but there was no response from age 1 Onchorynchus mykiss.</td>
<td>Fish abundances responded to restoration during low or moderate flow years, but not high flow years.</td>
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<tr>
<td>Cederholm et al. (1997)</td>
<td>North Fork Porter Ck</td>
<td>Increase the size and frequency of pools and amount of LWD cover during winter to influence the number of overwintering salmonids.</td>
<td>500 m engineered reach: 133 structures containing 300 LWD logs in total. 5 different structures used, including single logs and log jams to increase pools as well as provide cover for fish. Loggers reach 60 tree beside stream were felled and attached to their stumps using cable</td>
<td>Upland reach</td>
<td>Before and After sampling: Before period: 9 times, After period: 9 times. Two 500m treatment reaches, each 500m had different treatment and were paired to a single 500m reach that served as a control, located immediately upstream of both treatment sites. Sites were placed close together on one stream to keep physical and biological conditions similar. No reference reach</td>
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<td>Fish abundances responded to restoration during low or moderate flow years, but not high flow years.</td>
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<td>Van Zyll Delong et al. (1997)</td>
<td>Joe Farrells Brook, Eastern Pond Brook, Newfoundland, Canada</td>
<td>Restoration undertaken to increase the amount and diversity to improve resources available for salmonids</td>
<td>40m sites engineered reach: 133 structures containing 300 LWD logs in total. 5 different structures used, including single logs and log jams to increase pools as well as provide cover for fish. Loggers reach 60 tree beside stream were felled and attached to their stumps using cable</td>
<td>Upland reach</td>
<td>Before and After sampling: Before period: 1 time, After period: 2 times. One restored reach, containing several sections with different restoration types. 4 V-dam sites and 2 half-log sites were sampled. A single control reach was located on a separate river. No reference reach</td>
<td>Before period: summer 1993, After period summer 1994, 1995</td>
<td>V-dam had greatest effects on depth and increase in % area of pool. Half-log covers greatly improved cover in bare pools. Control reach did not change during experiment</td>
<td>V-dam, a year after restoration, higher abundance of ages 0+, 1+ and 3+ Salmo salar were detected in treatment reach, compared to the control reach. V-dam had little effect on abundance of Salvelinus fontinalis as their abundance declined in the post-treatment period. Half-log cover: Salvelinus fontinalis did not respond to this cover, with a after period, treatment abundance increases only for 0+ Salmo salar density.</td>
<td>Brook trout didn't respond to V-dams as predicted, and it may be because salmon were numerically abundant.</td>
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<td>Study</td>
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<td>Purpose of restoration</td>
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<td>Shields et al. (1998)</td>
<td>Mississippi, Ohio, Bobo Bayou, Goodwin Ck, Tidby Tulby Ck, Mississippi, North America</td>
<td>Improve habitat heterogeneity by developing pools</td>
<td>1000m per stream 4 x 100m sections within each stream selected for sampling</td>
<td>Lowland reach</td>
<td>LWD willow stakes up to 0.3 diameter were added to existing stone bed protection measures.</td>
<td>Before period: 4 times, After period: 6 times. 2 restored rivers, 2 control river, 1 reference river</td>
<td>Before period: spring and autumn, 1991-1992, After period: spring and autumn, 1993-1995</td>
<td>Deep scour holes occurred in two treatment reaches immediately after placement (3-4 weeks). Hilotphia Ck structures had little effect on habitat. Goodwin Ck structures produced large pools that covered the reach.</td>
<td>Only in Hilotphia Ck, largest effects on species composition within first 2 years. 8 new species (72%) increased were present on average in the after period. Before after increases in abundance occurred at Hilotphia Ck reach, but not different to after period control reaches. Hilotphia Creek reach showed biggest change to fish assemblage, despite having smaller change in habitat from restoration.</td>
<td>Authors believe that access by colonisers was significant in producing patterns. Fish colonisation after two large flow events. Restoration did not address a major disturbance-variable hydrology.</td>
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<td>Solaz et al. (2000)</td>
<td>Yellowstone National Park, East Fork River, Bear Lodge Creek, Upper Bear Lodge Creek, Moon Creek, East Creek, Oregon, North America</td>
<td>Increase amount and complexity of winter cover</td>
<td>3.2 km modified in Upper Lobster Creek, 2.4 km modified in East Creek</td>
<td>Upland reach</td>
<td>Anchored LWD pieces to create large pools and assist scour within pools. Rootwads and smaller trees were also added to increase habitat complexity. Off-channel rearing ponds excavated.</td>
<td>Before period: before period 3 to 4 times; After period: 4 to 5 times. Sampling years within Before-after periods varied between the two treatment rivers and among response variables within treatment rivers. Surveys consisted of 2 restored reaches and 2 Control streams. Restored streams were paired to Control streams. No reference location</td>
<td>Before period: winter 1986-1990, After period: winter 1989-1994</td>
<td>Constructed pools averaged 3 times larger than natural pools. Creation of slower flowing water and addition of LWD as shelter was important to increasing winter habitat.</td>
<td>Increasing winter habitat assisted with survival, corresponding to abundance increase for Oncorhynchus kisutch. Excavated alcoves and dammed pools were successful as winter refuge from high velocities.</td>
<td>Habitat changes designed for Oncorhynchus kisutch, also were used by other salmonids suggesting restoration can provide benefits at the assemblage level.</td>
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<tr>
<td>Roni and Quinn (2001)</td>
<td>Oregon and Washington, USA</td>
<td>Restoration aimed to change fish habitat to reverse declining salmonid populations</td>
<td>70 to 130 m long</td>
<td>Upland reach</td>
<td>Restored sites consisted of various different applications of LWD structures.</td>
<td>After period: no before period data; sites sampled twice after restoration, once in summer and once in winter. Restoration locations paired to nearby control locations on the same river, to control for large-scale geomorphologic effects on channel morphology</td>
<td>After period: Winter and Summer, between 1996 and 1999. Winter sampling occurred in 1997 and 1998</td>
<td>Treated stream reaches exceeded reference reaches in total wetted area, total number of habitat units, total pool area, and total number of pools during both summer and winter</td>
<td>Juvenile Oncorhynchus kisutch 1.81 times higher in treated during summer, and 3.2 times higher in treated during winter. Age 1+ Oncorhynchus clarki and Oncorhynchus mykiss were 1.7 and 1.73 times more abundant in treatment reaches during winter.</td>
<td>Difference in the number of wood pieces between control and restored sites positively correlated to the size of the response of Oncorhynchus kisutch.</td>
</tr>
<tr>
<td>Lehane et al. (2002)</td>
<td>Dublin, Ireland, Cork, Ireland</td>
<td>To mitigate the influence of forestry practices</td>
<td>2 x 200m reaches. Each reach has 2 x 25m sections of treatment</td>
<td>Upland reach</td>
<td>Partial LWD debris dams, constructed using 4-6 trunks with roots attached.</td>
<td>Before and After sampling. Before period: 1 time; After period: 3 times; 2 replicates of treatment and 6 replicates of controls per 300m reach; no reference location</td>
<td>Before period: spring 1998, After period: autumn 1998, spring 1999, spring 2000</td>
<td>Localized increase in depth, reductions in stream velocity and channel constriction. Redistribution of fine sediment</td>
<td>Salmo trutta density and biomass increased 1 to 2 years after adding LWD treatment. Smaller fish (age 1+) increased in abundance than older 2+ fish, but more age 1+ were present.</td>
<td>No wood present before restoration.</td>
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<td>Zika and Peter (2002)</td>
<td>Mühlebach River, Forstentum, Liechtenstein</td>
<td>LWD inserted to improve salmonid habitat in a channelised stream</td>
<td>250m reaches</td>
<td>Upland reach</td>
<td>Large trees cut down beside the channel, every 20m, dropping 45 trees into the river. Wood load: 3.5 to 4 pieces per 100 m. No wood present before restoration.</td>
<td>Before and After sampling. Before period: 1 time; After period: 2 times. Only one site restored and control site was selected for sampling from 5 restored and control reaches. Length of sites sampled was 150m. No reference site</td>
<td>Before period: 1995, After period: yearly during winter in February 1996 and March 1997.</td>
<td>Adding LWD increased depth, and reduce velocity in the treatment reach. Channel reaches with greater amounts of wood had more, and larger (higher volume) pools.</td>
<td>Abundance and biomass of Salmo trutta and Oncorhynchus mykiss increased in treated sites, after 86 days and 29 days respectively. Brown trout minimum, average and maximum size progressively increased after LWD addition. Greater numbers of larger trout &gt; 200 mm. also observed after LWD addition.</td>
<td>Trout used added LWD for cover during winter and for refuge during summer.</td>
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<tr>
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<td>Moerke and Lambert (2003)</td>
<td>Juday Ck, Potato Ck, Indiana, North America</td>
<td>Increase fish diversity</td>
<td>Restoration</td>
<td>Woody</td>
<td>Woody treatment used</td>
<td>Monitoring employed/design</td>
<td>Monitoring time period</td>
<td>Habitat effect</td>
<td>Fish response</td>
<td>Comments</td>
</tr>
<tr>
<td>Shields et al. (2003)</td>
<td>Little Topshaw Ck, Mississipi, North America</td>
<td>Stop-channel incision and accelerate recovery of habitat and fish assemblages</td>
<td>Restoration</td>
<td>Reach</td>
<td>Woody treatment used</td>
<td>Monitoring employed/design</td>
<td>Monitoring time period</td>
<td>Habitat effect</td>
<td>Fish response</td>
<td>Comments</td>
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<tr>
<td>Mooie et al. (2004)</td>
<td>Juday Ck, Indiana, North America</td>
<td>Increasing habitat heterogeneity</td>
<td>Restoration</td>
<td>Woody</td>
<td>Woody treatment used</td>
<td>Monitoring employed/design</td>
<td>Monitoring time period</td>
<td>Habitat effect</td>
<td>Fish response</td>
<td>Comments</td>
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<td>Nicol et al. (2004)</td>
<td>Murray River, Victoria, Australia</td>
<td>LWD restoration undertaken to improve fish habitat</td>
<td>Restoration</td>
<td>Reach</td>
<td>Woody treatment used</td>
<td>Monitoring employed/design</td>
<td>Monitoring time period</td>
<td>Habitat effect</td>
<td>Fish response</td>
<td>Comments</td>
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<td>Wright and Flesher (2004)</td>
<td>Rio Las Marcos, Venezuela</td>
<td>LWD restoration tackled due to impacts of Simbar</td>
<td>Restoration</td>
<td>reach</td>
<td>Woody treatment used</td>
<td>Monitoring employed/design</td>
<td>Monitoring time period</td>
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<td>Study</td>
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<td>Scale of restoration</td>
<td>Reach type</td>
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<tr>
<td>Bond and Lake (2005)</td>
<td>Castle Cr, Creighonts Cr, Victoria, Australia</td>
<td>To mitigate influence of sediment on habitat</td>
<td>100 m sections of stream restored and sampled</td>
<td>Lowland reach</td>
<td>Individual LWD pieces: 1 or 4 pieces per site</td>
<td>BACI design, Before period: 3 times; After period: 4 times; 3 replicates for each treatment level, 3 control locations per stream; no reference location</td>
<td>Before period: November 2000 to March 2001; After period: September 2001 to August 2002</td>
<td>Local increases in depth around wood pieces but scour pools were small and stream dried due to drought. LWD treatments effective in accumulating organic matter.</td>
<td>No effect of restoration on species composition.</td>
<td>Indeed small bodied fish.</td>
</tr>
<tr>
<td>Johnson et al. (2005)</td>
<td>Ten Mile Cr, Cummins Cr, Oregon, North America</td>
<td>Quantify the change in fish production</td>
<td>LWD additions approximately occurred over 6 miles of river</td>
<td>Upland reach</td>
<td>LWD &gt; 0.2m in diameter, 30-35m long, 0.75m diameter</td>
<td>BACI design; Number of sampling events varied from 1 per year for summer fish populations, to once per week (March to June) for smolt migration over 12 years</td>
<td>Before data: summer 1991-1995; After data: summer 1997-2001</td>
<td>Increases in the amount of wood in the treatment reach from storm and restoration resulted in a higher area and depth of pools in the upper two reaches.</td>
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<tr>
<td>Brooks et al. (2006)</td>
<td>Williams River, NSW, Australia</td>
<td>Improve channel stability and recreate habitat diversity</td>
<td>1.100 m treated 550 m control</td>
<td>Lowland reach</td>
<td>20 engineered LWD log jams, placed in difference positions along channel; LWD volume inserted 0.034 m$^3$/m$^2$</td>
<td>BACI design, Before period: 2 times; After period: 5 times; 1 treatment reach paired to a control reach; no reference location</td>
<td>Before period: April 2000, September 2000, After period: December 2000, April 2001; then on an annual basis: winter 2002, autumn 2004, autumn 2005</td>
<td>LWD structures increased the area of pools and riffles, and reach depth. Structures were successful in retaining sediment, unlike control reaches which transported sediment</td>
<td></td>
<td>Structures persisted through 5 large flood events</td>
</tr>
<tr>
<td>Shields et al. (2006)</td>
<td>Little Topsham Cr, Toby Tubbay Cr Mississippi, North America</td>
<td>Accelerate recovery of habitat and fish assemblages</td>
<td>2000m restored; 2 x 150m sections selected for sampling</td>
<td>Lowland reach</td>
<td>LWD &gt; 0.3m in diameter, 1168 logs placed into 72 log jam along the reach</td>
<td>BACI design, Before period: 2 times (once annually). After period 4 times (once annually). 2 treatment sections sampled, 1 upstream control section sampled; 1 reference TT Cr</td>
<td>Before period: summer and Autumn in 1999 and 2000</td>
<td>Treated habitat failed to produce deeper and more retentive functions. Only 64% of wood structures survived after 1 year, because of this wood structures failed to retain smaller organic matter</td>
<td></td>
<td>Treated habitat failed to produce deeper and more retentive functions – though fish populations still moved towards larger individuals as per reference stream, suggesting that use of wood as cover was more important, than its function in altering channel morphology.</td>
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<tr>
<td>Study</td>
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<td>Rosi-Marshall et al. (2006)</td>
<td>Cook’s Run, Michigan, North America</td>
<td>Restoration landslide</td>
<td>100 m reaches</td>
<td>NA</td>
<td>LWD added as skybooms (artificial cover) and K-dams or log dam; T1 reach consisted of 2 K-dam structures; T2 reach received skyboom structures</td>
<td>Before and After sampling, Before-period 1 time, After-period 2 times. Two 100m treatment reaches were sampled; one control reach located upstream and no reference reach.</td>
<td>Before period: 1998, After period: 2000 (after K-dams) and 2002 (after skybooms).</td>
<td>Both techniques increased depth and Organic matter amount</td>
<td>After 1 year, abundance of larger, harvestable trout were 2 x control for the K-dam reach and 3 x control for the skyboom reach. Although, overall trout abundances did not increase. Adult trout habitat was suspected of being limited in abundance, which restoration rectified.</td>
<td>After channel remanagement fish assemblage changed from lentic to lotic. Adding woody debris as flow refugee and cover can additionally benefit fish assemblages.</td>
</tr>
<tr>
<td>Hrodey and Sutton (2008)</td>
<td>Upper Wabash River, Indiana, North America</td>
<td>Restoration undertaken to increase habitat diversity</td>
<td>25m sites</td>
<td>Lowland reach</td>
<td>108 half log structures added to the stream. Four half logs placed in each 25m site and arranged in pairs.</td>
<td>Before and after sampling, but only After-period data examined. After period: 10 times (4 times in 2003 and 6 times 2004). Sampling occurred in 9 streams within 3 stream types: agriculture, forest and followed-field use. Each stream site had 3 paired treatment and control sites.</td>
<td>Before period: June 2003, After period: monthly between July-October 2003 and April-September 2008</td>
<td>Addition of LWD assisted in increasing the area of cover. During floods, added LWD assisted as velocity refuge. Only 2 LWD structures were lost due to flooding.</td>
<td>An increase in species richness, abundance and biomass was detected in treatment sites, however, this depended on stream as well as existing amounts of other cover.</td>
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</tr>
<tr>
<td>Nagayama et al. (2008)</td>
<td>Shibetsu River, Japan</td>
<td>Restoration landslide</td>
<td>470 m treatment reach</td>
<td>Lowland reach</td>
<td>4 LWD structures, each consisting of 3 trees (8-13m long) and rootwads. LWD structures floated freely in the current</td>
<td>NA</td>
<td>After period: (July) in 2005</td>
<td>LWD structures reduced near bank velocities</td>
<td>Oncorhynchus masou associated with LWD. Large fish were associated with dense cover, while smaller fish used a range of different cover types.</td>
<td>Within a year of adding treatments, juvenile and adult Oncorhynchus masou colonized during summer. In winter, Leuciscus cus and Oncorhynchus keta inhabited structures. Areas with LWD structures contained more species than without LWD in both Autumn and winter. No fish observed in lateral scour pools where LWD was not added.</td>
</tr>
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<td>Nagayama et al. (2009)</td>
<td>Shibetsu River, Japan</td>
<td>Restoration landslide</td>
<td>470 m treatment reach</td>
<td>Lowland reach</td>
<td>4 LWD structures, each consisting of 3 trees (8-13m long) and rootwads. LWD structures floated freely in the current</td>
<td>After period: Sampled once during summer in 2005. Sites surveyed consisted of 4 LWD structures, 4 areas without LWD and no reference site.</td>
<td>After period: summer (July) in 2005</td>
<td>LWD structures reduced near bank velocities</td>
<td>Gondorhynchus masou associated with LWD. Large fish were associated with dense cover, while smaller fish used a range of different cover types.</td>
<td>Within a year of adding treatments, juvenile and adult Oncorhynchus masou colonized during summer. In winter, Leuciscus cus and Oncorhynchus keta inhabited structures. Areas with LWD structures contained more species than without LWD in both Autumn and winter. No fish observed in lateral scour pools where LWD was not added.</td>
</tr>
<tr>
<td>Paner and Geist (2010)</td>
<td>Guest River, Germany</td>
<td>Restoration landslide</td>
<td>1080m sampled that had been restored from a channelized, stream</td>
<td>Upland reach</td>
<td>Large LWD bundles inserted diameter of 1-1.2m and 25m in length</td>
<td>After period sampling. After-period 2 times sampled. Sites consisting of 30m site lengths for each of 4 habitat types, 3 using boulders, placed at strategic vegetation sites. One treatment included was placed LWD bundles. Nine replicates of each site type were sampled, with a 100m separating sites</td>
<td>Sampling conducted 2 years after completion of restoration procedures, once in February 2008 (winter) and once in June 2008 (Summer)</td>
<td>Added woody debris only provided more cover for fish</td>
<td>Woody debris treatments contained the highest species richness and abundance compared to 3 other boulder-vegetation cover combinations. Higher species richness occurred in summer for all treatment types. Densities were consistent between seasons for different boulder treatments, except woody debris, where higher densities of small-size fish species were associated with woody debris placements in winter. Densities in woody debris were 3.4 to 8.6 times larger than boulder substrate treatments.</td>
<td>Woody debris treatments contained the highest species richness and abundance compared to 3 other boulder-vegetation cover combinations. Higher species richness occurred in summer for all treatment types. Densities were consistent between seasons for different boulder treatments, except woody debris, where higher densities of small-size fish species were associated with woody debris placements in winter. Densities in woody debris were 3.4 to 8.6 times larger than boulder substrate treatments.</td>
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<td>Scale of restoration</td>
<td>Reach type</td>
<td>Woody treatment used</td>
<td>Monitoring project</td>
<td>Monitoring time period</td>
<td>Habitat effect</td>
<td>Fish response</td>
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<td>Anton et al. (2011)</td>
<td>Atseginsoro River, Malbazar River, Lates River, Anarbe River, Basque, Spain</td>
<td>LWD inserted to increase habitat complexity and biodiversity</td>
<td>100 m reaches except Anarbe River, which was 400m in length</td>
<td>Upland reach</td>
<td>Restoration consisted of trees being cut down besides the river and pulled in using winches. LWD was not anchored to the bed</td>
<td>BACI design, Before-period 2 times, After-period 4 times.  Single control site above each treatment reach in 4 rivers. No reference reach.</td>
<td>Before period: winter 2006 and summer 2007. After period: summer 2008, winter 2008, summer 2009 and winter 2009</td>
<td>Log addition resulted in sudden changes to habitat, especially the number and depth of pools. Eventually, leaf litter, finer and coarser sediments were retained by logs leading to a reduction in depth.</td>
<td>Adult trout abundance remained 53% higher in treatment reaches compared to control reaches after 21 years. However, adult trout abundance was not significantly different to the first post-treatment period (1987-1994). Most log structures were still in good condition after 21 years despite being exposed to largest floods on record.</td>
<td>Consistent response of adult trout hypothesized to relate to a limitation in pool habitat.</td>
</tr>
<tr>
<td>White et al. (2011)</td>
<td>Colorado Creek, Walton Creek, Cache la Poudre River, Jack Creek, Little Beaver Creek, St. Vrain Creek</td>
<td>Experimental restoration undertaken to quantify influence on trout life-history processes.</td>
<td>250 m modified section for 6 streams. 1x 375m modified section for St. Vrain Creek</td>
<td>Upland reach</td>
<td>Large woody debris, log drop structures placed across channel to form weirs.</td>
<td>Before and After sampling. Before period: 2 times. After period: 6 times a year. Sites were resampled again in 2009. A single treatment reach was paired to a single control, either downstream or upstream, except St Vrain Creek, which had control sites upstream and downstream.</td>
<td>Before: summer, 1987-1989. After: summer 1988-1994 (Gowan and Fauch 1996) then resampled in 2009 as for this study.</td>
<td>Logs create scour pools downstream and dammed pools upstream. Total cover was greater in restored reaches. After 21 years pool volume remained more than 3 times higher in restored reaches, although it was not different to the first after period (1987-1994). Most log structures were still in good condition after 21 years despite being exposed to largest floods on record.</td>
<td>Adult trout abundance remained 53% higher in treatment reaches compared to control reaches after 21 years. However, adult trout abundance was not significantly different to the first post-treatment period (1987-1994). Most log structures were still in good condition after 21 years despite being exposed to largest floods on record.</td>
<td>Consistent response of adult trout hypothesized to relate to a limitation in pool habitat. To authors knowledge, study is the longest of instream habitat studies published to date.</td>
</tr>
<tr>
<td>Howell et al. (2012)</td>
<td>Hunter River, NSW, Australia</td>
<td>Improve channel stability and recreate habitat diversity</td>
<td>Restoration was undertaken over a 10km, while whole pool, and 90-100 m long riffle sites selected for sampling units</td>
<td>Lowland reach</td>
<td>0.008 to 0.038 m³ of wood added to riffles.</td>
<td>BACI design: Before period: 3 sampling times, After period: 6 sampling times. Sites sampled included 3 treatment riffles, 3 no control riffles, 2 far control riffles, 3 treatment pools and 1 control pool. No reference reach</td>
<td>Before period: January, April and July 2004. After period: October 2004, January, April, July and October 2005, and January 2006</td>
<td>Wood volume in the treated reach doubled the existing amount. Refill jams did not arrive in the channel structure, except they created zones of low flow refuge.</td>
<td>Addition of LWD to riffles increased the abundance of Retrophrya semoni and Gambusia holbrooki. Fish assemblages did not respond to additions of LWD structures in pools. Fish abundances changed seasonally, with lowest abundances recorded in winter and highest during summer.</td>
<td>Fish responses to wood differed between warmer and cooler seasons.</td>
</tr>
<tr>
<td>Nagayama et al. 2012</td>
<td>Shibetsu River, Japan</td>
<td>LWD added to increase habitat diversity</td>
<td>470 m treatment reach, Site treatments were 4 to 13m debris patches or 4 m no-wood control patches</td>
<td>Lowland reach</td>
<td>SAV structures &gt; 4 woody pieces and U structures &gt; 3 wood pieces. These sites were 4 to 12 m long.</td>
<td>After period: Sampled once during autumn and winter in 2006. Sites surveyed consisted of 25 treatment sites in autumn and 34 sites in winter. 5 and 8 no-wood control sites were sampled in autumn and winter, respectively.</td>
<td>After period: autumn (September) and winter (December) in 2006</td>
<td>Sites with large wood were deeper and contained finer substrates. In winter, sites with more large wood (U sites) had slower velocities and finer bed materials than sites with little or no large wood.</td>
<td>Sites with wood had between 2.5 to 5 times the number of species than sites without wood for both autumn and winter. Abundance of the four most dominant species was higher with more large wood pieces during winter. Two of the dominant species present had higher abundances with greater amounts of large wood in autumn.</td>
<td>Fish responses to wood differ between warmer and cooler seasons.</td>
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<td>Question</td>
<td>Null hypotheses</td>
<td>Chapter</td>
<td>Project publications</td>
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</table>
| 1. Does the implementation of a reach-scale river rehabilitation program alter fish assemblage structure? | $H_0$: Fish assemblage composition at the rehabilitated reach does not diverge from un-manipulated locations, within two years after the completion of works  
$H_0$: Species richness, individual or group, and total fish abundances does not differ between rehabilitated and un-manipulated reaches after the completion of works  
$H_0$: Water quality parameters do not vary among rehabilitated and un-manipulated locations, nor does a relationship exist with fish assemblage composition | 3       | Howson et al. (2009) Fish assemblage response to rehabilitation of a sand-sluged lowland river, River Research and Applications, 25, 1251-1267.          |
| 2. Does the quantity and size of woody debris influence fish assemblage structure without restoration altering channel morphology? | $H_0$: The amount of LWD present in channels impacted by sedimentation does not correspond with fish assemblage structure (composition, richness, total and taxa abundance).  
$H_0$: Adding SWD to different amounts of LWD in sedimented channels, does not alter assemblage composition, increase species richness, total abundance, individual species abundances, or, adult and juvenile abundances within species  
$H_0$: Other habitat components, including channel width and depth, water quality and velocity does not vary significantly among LWD and SWD groups | 4       | Howson et al. (2012) Size and quantity affects fish assemblages in a sediment-disturbed lowland river, Ecological Engineering, 40, 144-152.                    |
| 3. Does rehabilitation consisting of sediment extraction with woody debris replacement influence fish spawning? | $H_0$: Spawning frequency does not differ among channel types (pools and runs) and reach types (rehabilitated and un-manipulated)  
$H_0$: In pools, spawning frequency does not vary between deep and shallow zones, or among rehabilitated and un-manipulated reaches  
$H_0$: Adding woody debris to sedimented channels is not used as a spawning substrate  
$H_0$: Substrate complexity of woody debris and artificial substrata does not influence the frequency of spawning events  
$H_0$: Habitat characteristics do not differ among channel types and reaches | 5       | Howson et al. (2010) Patch specific spawning is linked to restoration of a sediment-disturbed lowland river, south-eastern Australia, Ecological Engineering, 36, 920-929. |
2. Chapter Two: The Glenelg River Catchment–Impacts, Restoration and Fish

2.1. Catchment description

The Glenelg River is situated in far southwestern Victoria, beginning in the Victoria Range (Grampians National Park) and then flowing south, entering the Southern Ocean (Figure 2.1). The Glenelg River catchment covers an approximate area of 12,700 km² and the river is the longest in western Victoria and amongst the largest in the state. Several major tributaries drain into the Glenelg River. The Wannon River is the largest and also has headwaters beginning in the Grampians ranges, flowing southwest and joining the Glenelg River near the town of Casterton. Larger tributaries including the Crawford, Stoke, Wando, Henty and Chetwynd River’s, Pidgeon Ponds and Mathers Creek, which contain water over much of the year owing to remnant pools that from a ‘chain of ponds’ (Erskine 1994, Rutherfurd and Budahazy 1996). Many other tributaries entering the mainstem river are low order streams draining straight from the adjacent surrounding hills. These tributaries are often intermittent, flowing only during the wetter seasons of winter and spring (T. Howson personal observation).

The Glenelg River catchment is situated in a dry, Mediterranean type climate, with cool wet winters with warm dry summers (Soil Conservation Authority 1980, Department of Water Resources 1989). At the town of Hamilton (centrally located in the catchment) long-term mean maximum monthly temperatures over summer (December to February) range between 23.3 to 25.7 °C, while long-term mean minimum monthly temperatures of winter (June to August) range between 4.5 to 5.2 °C (Bureau of Meteorology 2007). Highest rainfall occurs during late winter and spring with mean annual rainfall regionally varying from 500 to 700 mm in the central and coastal catchment areas, while higher falls of 900 mm occur in the Grampians ranges (Department of Water Resources 1989).
Figure 2.1 The Glenelg River catchment, Victoria, Australia.
The catchment landscape varies from high mountainous terrain in the Grampians, to the deeply divided Dundas and Merino tablelands, to the low relief basaltic plains of the southeast and the coastal plains in the south. Rivers and creeks in each of these regions drain different geologies, resulting in significant changes to channel morphology. In the lower reaches, the river cuts through softer sandstones and limestone yielding a deeply incised channel and forms a gorge in the estuary (Rutherfurd and Budahazy 1996). In the Dundas tablelands where this thesis is set, rock and sediment consist of coarse and fine textured deposits (black earths and cracking clays, sandy loams), sedimentary (sandstones), volcanic rocks, granites and gneisses (Soil Conservation Authority 1980, Department of Water Resources 1989, Erskine 1994). Channel morphology in this region consists of large pools (Plate 2.1), interdispersed with braided runs. In some places pools extend over a kilometre in length and reach depths of 8 m (McGuckin et al. 1991) and are suspected to play important roles as refuges for variety of taxa particularly during dry summer periods (Mitchell et al. 1996). Along several reaches, deep pools are interdispersed by anastomosing channel sections (multiple channels separated by floodplain), where the main channel is flanked by several parallel channels, but only the main channel is usually wet and shallow (< 1.5 m deep) with all other channels remaining dry, only becoming inundated during floods (T. Howson personal observation). Anastomosing channels in the in the upper catchment are not rare but are considered quite unique for a humid river system and regarded as unusual and important (Rutherfurd and Budahazy 1996).

Land use across the catchment is predominantly farming with stock grazing and broad acre cropping consisting of 67.9% of the catchment area. The dominant vegetation across the catchment is low forest, native grassland and heath land scrub consisting of a variety of mixed species covering 28.0% of the catchment area (Department of Water Resources 1989). Remnant patches of forest still persist across
Plate 2.1 Large, deep pools such as these near the Five Mile outlet (top) and Crawford River (bottom) were once prevalent in the Glenelg River. Many large pools have now been filled by sand (see Plate 2.2).
the middle catchment but are somewhat scattered over crown land, as opposed to
the well forested upper reaches flowing through the Grampians National Park.
Much of the Glenelg River from Rocklands reservoir to Casterton is considered to
contain continuous riparian vegetation (Department of Water Resources 1989) as
the river is currently tree-lined with large River Red gum (*Eucalyptus camaldulensis*
[Dehnhardt]) and Swamp gum (*Eucalyptus ovata* [Labillardiere]). The understory is
largely absent but where found, it consists of small tree genera including *Melaleuca,
Leptospermum* and *Acacia* spp.. Other land uses in the region include forestry with
Pine (*Pinus radiata* [D. Don]) and hardwood (*Eucalyptus* spp.) plantations.

2.2. Catchment impacts

The Glenelg River catchment has been subjected to many disturbances since
settlers first reached the area in the 1830’s, after being attracted by abundant
grasslands providing prime grazing territory. Further, cultivation of cereal crops
were carried out on the surrounding hill slopes and in valley bottoms, where
furrows were often ploughed in order to encourage rapid drainage. As early as the
1850’s, settlers reported a rapidly changing landscape. Stock feed was lost due to
the disappearance of various vegetation types leaving bare earth in its place,
landslips and eroding hill-slopes were prevalent where furrows were cut and springs
of salt water became an emerging problem. Deep ruts caused by extensive soil
erosion (up to 3 m deep) that extended well into gullies made it difficult for
landholder’s to travel across the landscape, even on horse back (Soil Conservation
Authority 1980).

Over two thirds of the Glenelg River catchment has been cleared of its vegetation
for agriculture development (Department of Water Resources 1989). As indicated
by the early settlers, severe sheet and gully erosion has developed throughout the
catchment in response to early farming practices. The removal of vegetation
combined with stock trampling of the banks has led to channel incision and head-
ward cutting of tributaries, resulting in the excessive export of sediment to tributary
and mainstem reaches and the loss of the original ‘chain of ponds’ channel
structure (Rutherfurd and Budahazy 1996). Characteristically, the Glenelg River is
most renowned as an example of a large Australian river that has undergone
significant geomorphic change from excessive sediment input (Erskine 1994,
Rutherfurd and Budahazy 1996). Significant volumes of sediment have entered the
system in the order 10,000 to 50,000 m$^3$ per kilometre of channel (Rutherfurd and
Budahazy 1996). This excessive amount of sediment has resulted in extensive
changes to river morphology, reducing channel capacity and depth (Plate 2.2). Many
large pools that were once several meters deep are now filled in having become
long, shallow, homogenous lengths of channel while other components deemed
important to biota such as woody debris and undercut banks have been buried
(Rutherfurd and Budahazy 1996).

Sand deposited into the Glenelg River from surrounding tributaries is believed to
have peaked after the 1946 floods, with sediment distribution changing little since
this time (Rutherfurd and Budahazy 1996). Concern over the potential for increased
amounts of sediment blocking the channel and increasing flood frequency during
the 1950’s has led to desnagging operations in the Glenelg River catchment (Erskine
and Webb 2003). Early attempts at sand management from the development of the
Glenelg River Improvement Trust during the 1960’s involved active desnagging of
the mid reaches of the Glenelg River in order to increase flow velocities to increase
the sediment carrying capacity of the river and to transport sediment rapidly
downstream. At the time, desnagging was successful in increasing flow velocity
from 0.45 ms$^{-1}$ to 0.54 ms$^{-1}$ downstream of Casterton potentially flushing sediment
further downstream (Erskine and Webb 2003). However, this management practice
has inadvertently caused further problems in that sediment is now advancing
towards the now heritage listed, Lower Glenelg River National Park. In an attempt
to alleviate the risk of sediment reaching the heritage-listed lower reaches,
commercial sand mining is undertaken in several areas of the catchment and is
recommended as a long-term sediment control strategy (Rutherfurd and Budahazy
1996).
Plate 2.2 The Glenelg River near Bourke’s Bridge, a photograph of the typical middle to upper reaches where sediment has greatly infilled the channel. According to figure 3.2 in Rutherfurd and Budhazy (1996), the original channel bed lies a depth of 3-4 m beneath the current bed height.

A large water reservoir, Rocklands Reservoir is located on the Glenelg River near the township of Balmoral in the upper catchment. Completed in 1953, Rocklands Reservoir forms part of a large water inter-basin transfer system known as the Wimmera-Mallee stock and domestic supply system. Water is harvested from the Glenelg River catchment via Rocklands Reservoir and a number of headwater off-takes and transported north through a series of channel networks to the Wimmera-Mallee. With a total capacity of 348,000 ML, the reservoir was designed to store 3 times the mean annual discharge of 110,000 ML (Mitchell et al. 1996, Lind 2004). The impact of Rocklands Reservoir has significantly altered the downstream hydrology of the Glenelg River, with reductions in mean annual discharge below the reservoir from 71,500 ML to 22,000 ML (approximately 70% reduction, Mitchell et al. 1996, Lind 2004). As a consequence, an environmental flow regime has been
established for the Glenelg River in an attempt to improve river health (Mitchell et al. 1996, Lind et al. 2006), however, during periods of drought environmental flow releases are restricted (Lind 2004, Lind et al. 2006).

Catchment clearing and alteration of system hydrology is believed to have contributed to increased salinities in the Glenelg River catchment (Soil Conservation Authority 1980, Cameron and Jekabson 1992). Groundwater in the region is naturally salty owing to ancient inundation by the sea. However, deep-rooted perennial vegetation (e.g. *E. camaldulensis*) is believed to have partially controlled the groundwater table levels through removal of excess water by processes such as evapotranspiration (Soil Conservation Authority 1980). Removal of vegetation resulted in up to 200 mm of excess water entering deeper geological zones of the Dundas tablelands, raising the groundwater table in this region as much as 20 m in the past century resulting in saline water pouring out from drainage lines (Soil Conservation Authority 1980). Saline water from groundwater intrusion is also present in deeper pools, where water conductivities over 10,000 μS cm⁻¹ have been recorded (McGuckin et al. 1991, Cameron and Jekabson 1992, Coates and Mondon 2009). The presence of high salinity concentrations in pools is likely to become a physiological barrier to some taxon, especially during critical summer periods where a reduction in the amount of space available to taxa could occur in pool refuges (Mitchell et al. 1996, Lind 2004).

The legacy of past catchment practices has resulted in significant degradation of the Glenelg River catchment. Mitchell (1990) regarded the Glenelg River catchment as amongst the poorest in Victoria in terms of its environmental condition. Subsequent surveys in 1994 indicated that river condition was still regarded as moderate to poor (Davidson et al. 1994). Implementation of the state-wide Index of Stream Condition (ISC) in 1999 and 2004, using different methods and scoring systems still indicated that much of the Glenelg River is classified in a moderate or poor condition, indicating little has changed. The current, continuing degraded condition of the catchment has prompted authorities to include river rehabilitation activities
as part of catchment management and river health strategies for this region (GHCMA 2004).

2.3. Rehabilitating sediment-disturbed reaches of the Glenelg River

The reach-scale rehabilitation procedures used in this thesis were initiated and funded by Glenelg-Hopkins Catchment Management Authority and conducted by Earth Tech Pty. Ltd. Initially, rehabilitation was undertaken at a single reach on the Glenelg River behind the township of Harrow during February 2003, although, further opportunity arose to study a partial completion of rehabilitation works at Casterton during late 2004. In both these projects, sediment extraction by excavation and large woody debris (LWD) replacement was predominantly undertaken approximately over a 1500 m reach (Plate 2.3). Diagrams with photographs of rehabilitation procedures undertaken at both Harrow (Figures 3.2, 5.2) and Casterton (Figure 5.2) are shown in Chapters Three and Five, respectively.

Sediment extraction was undertaken using an excavator, where sediment was removed in two stages. Firstly, sediment extraction lengthened pools and deepened runs (Plate 2.3a and 2.3b) in an attempt to reintroduce a definitive channel and to improve connectivity between existing larger pools. Secondly, a sediment trap consisting of a 1500 m³ (50 m long x 15 m wide x 2 m deep) hole dug into the bed immediately upstream of the rehabilitated reach enabled the collection of sediment moving into the reach (Photograph A, Figure 3.2). As part of ongoing maintenance, the sediment trap is annually ‘cleaned’ by excavation.

Once sediment extraction was completed, extra-large size pieces of woody debris were placed within the newly re-constructed channels to induce scouring and further deepening of the bed, prevent the localised build up of sediment and protect eroded banks from collapse and to provide additional habitat for biota along the reach. At Harrow, multiple pieces were also placed together in piles or in an engineered type arrangement (Photograph B, Figure 3.2). In one instance, several logs were layered in a vertical arrangement against a bank to prevent...
further erosion (Photograph G, Figure 3.2). Engineered log jams consisted of approximately 12-16 large-trunk logs layered in 3 rows, with each row containing 3-4 logs. These structures are useful in deflecting flow and reducing velocity during high flow periods, thus assisting in bank protection (Shields et al. 2006). Criteria of Gipple and White (2000) were used in order to select timber for the woody debris replacement. This included selection of pieces larger than 0.3 m in diameter as large pieces have greater longevity than small pieces and less likely to fail during large floods (Earth Tech 2002). Replaced pieces of LWD were orientated at various angles, though, most were placed with the trunk orientated perpendicular to bank and slightly facing upstream or downstream. These orientations were used to ensure wood pieces occupied the largest amount of channel cross-sectional area, thereby increasing flow resistance, which induces greater bed scour when logs are slightly facing upstream, or, assist in bank protection by deflecting flow when logs are slightly facing downstream (Cherry and Beschta 1989, Gippel 1995).

Debris loadings for Harrow and Casterton were 0.011 and 0.013 m$^3$ m$^{-2}$ of channel, respectively, which is comparable to the LWD loading of 0.01 m$^3$ m$^{-2}$ specified by Treadwell et al. (1999), which was suggested as an initial starting point for woody debris re-introduction where no information on natural debris quantity is available. True LWD loads for the Harrow rehabilitated reach are likely to be much higher given several deep pools containing quantities of large woody debris are present. Observations over the initial 6 months after the completion of rehabilitation works indicated only localised bed scouring under very few wood pieces, however, most of the replaced LWD structure had been colonised by biofilm and by shrimp (*Paratya australiensis* [Kemp]).

Additionally at Harrow, three vehicle river crossings were also lowered (2 altered, 1 modified into a basic rock-ramp fish ladder) to improve the connectivity of the reach (Figure 3.2). Two vehicle crossings that spanned across lateral floodplain channels, were excavated to lower their height (ca. 0.5-1 m), to allow water to flow into these small channels. Subsequently, crushed bassalt rock (ca. 50-250 mm in diameter) was added to stabilise the bed of these crossings, presumably to allow
safe passage of vehicles to access the river during times of emergency (e.g. bushfire). Similarly, a third low height vehicle crossing spanning the Glenelg River at the downstream end of the rehabilitated reach, was excavated (8 m length) to lower the bed height (ca. 0.5 m), in which, crushed basalt rock and large boulders (150 to 600 mm in diameter) were used to stabilise the bed and to provide flow refuge for fish during crossing of the barrier.

Plate 2.3 River rehabilitation of the Harrow reach. Photographs (a) before-period and (b) after-period of sediment extracted from a degraded run. Photographs (c) single log structure and (d) racked-member log jam. Photographs (e) tree stump staked to bed with wood piles and (f) modified vehicle crossing. Note: All photographs (except e) were taken at low flow during February 2003. At time of sampling (winter 2003 onwards) woody debris structures and vehicle crossings were completely submerged.
2.4. Fishes of the Glenelg River Catchment

Despite the reported poor condition of the catchment, 19 native freshwater fish species, representing five orders and 10 families have been recorded (Table 2.1). Six species are recognised for their high conservation value on the *Advisory List of Threatened Vertebrate Fauna in Victoria* (Department of Sustainability and Environment 2007), while five species have been listed under state (*Flora and Fauna Guarantee Act 1988*) and federal (*Environmental Protection and Biodiversity Conservation Act 1999*) legislation. For some species, such as the Variegated pygmy perch (*Nannoperca variegata*, [Kuiter and Allen], Plate 2.4), the Glenelg River is the only catchment in Victoria where this species is found. In addition, both Yarra pygmy perch (*Nannoperca obscura*, [Klunzinger]) and Southern pygmy perch (*Nannoperca australis*) are also common to this system, making the Glenelg River catchment the only place in Victoria where these three species co-exist.

Other rare species considered under state and federal legislation, such as the Australian grayling (*Prototroctes maraena*, [Günther]) and dwarf galaxias (*Galaxiella pusilla*, [Mack]), also reside in the Glenelg River catchment (Close et al. 2003). For even some of the more abundant fish species encountered across Victoria, unique populations occur within the Glenelg River Catchment. Recent evidence for *Retropinna semoni*, suggests that despite evidence of mixing in coastal Victorian populations, the Glenelg River catchment is an evolutionary significant management unit (Hammer et al. 2007). Variations in the body forms of mountain galaxias (*Galaxias olidus*, Lower Glenelg River form) and river blackfish (*Gadopsis marmoratus*, Upper Wannon River form) have also been recently noted suggesting the likelihood of potential new species or sub-species (Close et al. 2003, Department of Sustainability and Environment 2007). Considering the significant local diversity of freshwater fishes, crayfish (Johnston and Robson 2009) along with other unique aquatic fauna (e.g. Glenelg freshwater mussel, *Hyridella glenelgensis*, [Dennant]), the continuing occurrence of environmental degradation represents a significant risk to conserving important regional biodiversity (Robson and Mitchell 2010).

<table>
<thead>
<tr>
<th>Order</th>
<th>Family</th>
<th>Species</th>
<th>Common Name</th>
<th>Origin</th>
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<td>Oncorhynchus mykiss</td>
<td>Rainbow trout</td>
<td>Exotic (Introduced)</td>
</tr>
</tbody>
</table>
Native species diversity within the catchment also consists of non-endemic species, such as Macquarie perch (*Macquaria australasica*, [Cuvier]) and golden perch (*Macquaria ambigua*), which have been historically translocated to the catchment in attempts to expand populations and create recreational angling opportunities. A native carp gudgeon(s) (*Hypseleotris* sp.) has been noted to occur in the Glenelg River system (Jackson and Davies 1983, Close et al. 2003), but not in the other regional coastal river systems (*i.e.* Hopkins, Curdies, Gellibrand and Aire Rivers, Department of Water Resources 1989). This suggests carp gudgeon has been potentially introduced to the Glenelg River catchment, given that carp gudgeons are common to inland drainages (*e.g.* Murray-Darling Basin, Balcombe and Closs 2000, Bertozzi et al. 2000).

Other recent discoveries of introduced fish taxon include common carp (*Cyprinus carpio*, [Linnaeus]) in 2001 and Australian bass (*Macquaria novemaculeata*, [Steindachner]) in 2003. The recent discovery of common carp is particularly unfortunate given the Glenelg River catchment was one of the last large ‘carp-free’ catchments in the state, given that carp have been implicated with degradation (*e.g.* increased turbidity) of waterways (King et al. 1997). The origin and timing of arrival of Australian bass into the catchment is unclear, but because no natural populations occur within several hundred kilometres the presence of Australian bass is likely due to human-assisted dispersal. The presence of Australian bass poses a risk to the genetic integrity of the western Victorian population of estuary perch (*Macquaria colonorum*, [Günther]), since hybridisation occurs between these two closely related species in eastern Australian catchments (Jerry et al. 1999).

Several other common exotic species to Victorian rivers, including: brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*, [Walbaum]), redfin (*Perca fluviatilis*, [Linnaeus]), tench (*Tinca tinca*, [Cuvier]), goldfish (*Carrassius auratus*, [Linnaeus]) and mosquito fish (*Gambusia holbrooki*) can be found within the catchment (Jackson and Davies 1983, Close et al. 2003). Most of the introduced species are larger, aggressive or piscivorous and are therefore likely to threaten smaller native fish through competition for space or resources or predation. During
dry periods when pools may become the only refuge available to fish, these biotic interactions may intensify (MacDonald et al. 2012). However, very little is known about the effects of exotic species on Australian native fish assemblages and particularly whether these alien fish species are reduced or enhanced by restoration (Nicol et al. 2004).

While many native and exotic fish species are present in the Glenelg River catchment, a distinctive assemblage feature of species inhabiting the mid to upper reaches of the Glenelg River, is that many are small in size. The absence of an large bodied indigenous fish species is peculiar for a large river system (Close et al. 2003), which may represent an absence of a natural piscivorous predator. Even short-finned eel (*Anguilla australis*, [Richardson]) are rarely detected in the upper Glenelg River catchment (Close et al. 2003), despite their common capture across multiple western Victorian catchments (T. Howson unpublished data). Redfin (*Perca fluviatilis*) is widespread in the upper catchment, particularly in deeper pools and therefore may have filled this large-body predator niche. The northern form of river blackfish (*Gadopsis marmoratus*) is the largest indigenous species that is commonly captured in the mid to upper regions (Close et al. 2003). Although small bodied fish species largely comprise the assemblage in the mid to upper Glenelg River, these patterns may not necessarily reflect indigenous species or possibly processes unique to the Glenelg River catchment, but likely to reflect evolutionary processes and the pool of freshwater species over larger spatial scales (Matthews 1998), considering a high proportion of native fish species (71%, 31 species) in Victorian rivers in general, commonly attain lengths of under 200 mm (Koehn and O’Connor 1990a). Nevertheless, body size distributions can be an important ‘taxon-free’ descriptor of organism relationships with their surrounding environment (Robson et al. 2005) and may yield important insight into the requirements of organisms.
Plate 2.4 Adult variegated pygmy perch (*Nannoperca variegata*) from the 'The Gorge,' Glenelg River. Note: largest fish is approximately 53 mm (TL).

2.5. Fish Sampling

The mid to upper reaches of the Glenelg River was chosen for this study, primarily because most of the sediment occupying the channel is located in this region (Rutherfurd and Budhazy, 1996) and because this is where restoration works have taken place. Initially, sediment-affected reaches with remaining deeper pools were of particular interest between the townships of Balmoral and Harrow (Figure 2.1). Within this region, there were limited deeper pools remaining, but many shallower runs were available where sediment had been deposited within the channel.

At the beginning this study, the initial restoration reach at the township of Harrow had been chosen and planning for restoration works was underway. However, there was a greater choice over the position of control locations in relation to the restored reach. Selection of control sites were decided on several factors including the similarly of location features to the rehabilitated reach before restoration, the proximity of control locations to rehabilitated reaches (i.e. spatial and temporal independence) and the availability of before-period data including access to
locations in after-period. Careful consideration of these factors is required, given, they ultimately affect the type of monitoring design used and thus the strength of inference obtained from monitoring (Downes et al. 2002).

The similarity of control reach locations to the rehabilitated reach and to each other, form an important component on deciding where potential control locations could be selected. After conducting several reconnaissance trips across the catchment, most of the accessible deeper pools were located upstream of the rehabilitated reach, which is favourable, when the negative impacts of sediment extraction (e.g. increase in suspended solids) downstream are largely unknown. Channel characteristics between Balmoral and Harrow were similar, with deeper pools interdispersed by areas of shallow runs, and the river bed consisting of largely sand or fine silt substrates (Lind et al. 2006, Turner and Erskine 2005). Woody debris was present in an assortment of sizes and forms (e.g. single logs, log and debris jams) while dominant macrophyte species including *Phragmites australis* ([Cavanilles] Trinius ex. Steudel) and *Typha domingensis* (Persoon), *Triglochin procerum* (Brown) and *Potamogeton ochreatus* (Raoul) cover the pool edges and runs (Lind et al. 2006). Downstream of Harrow, the river characteristically changes as the channel widens and becomes increasingly shallow (Rutherfurd and Budhazy, 1996).

Ensuring total independence between locations in an essential continuous ecosystem can be difficult, but using mark and recapture techniques can sometimes assist with determining whether fish move between control and restored locations (Gowan and Fausch, 1996; Howell et al. 2012). However, this was beyond the scope of the current these because mark and recapture techniques are inherently more difficult to undertake for small-bodied fish, especially, if they are not recognised migrators (i.e. individuals detected in other environments, such as the sea). Mark-recapture techniques also largely depend on recovery of marked individuals, and likely futile considering low numbers of fish were captured within the restored reach (Chapter Three). Furthermore, knowledge that the assemblage consisted of species that were not or unlikely to be migratory (e.g. *Gadopsis marmoratus*, *Nannoperca australis*, *Philypnodon grandiceps*; Close et al. 2003, Koehn and O’Connor 1990a, Cook et al.
2007, Koster and Crook 2007) suggested that locational independence may be achieved by separating rehabilitated and control reaches by a reasonable distance in the order of several km's.

A previous study (Close et al. 2003) examined fish distributions with salinity along several mid to upper reaches of the Glenelg River provided an opportunity to use valuable before-period data, which is preferred considering it can provide greater inferential strength than just post-rehabilitation sampling alone (i.e. BACI design, Downes et al. 2002). Close et al. (2003) sampled before the commencement of this project, and therefore, provided a description of the fishes present within the restored and control reaches within a short-time prior to rehabilitation. However, available data for trips (twice: once in autumn, once in winter) was limited to a few key pool locations, the rehabilitated reach at Harrow and suitable control reaches located upstream at ‘The Gorge’ and Five Mile outlet. For the present study, additional sampling was undertaken during summer, as well as winter, for these locations. An additional reach located on the Crawford River was also included as a higher quality ‘reference’ location—a reach that was largely un-impacted by sediment.

Fish sampling methods in the current study were kept consistent with those used by Close et al. (2003) so that the data collected could be used as the ‘after-period’ comparison. These methods included the use of fyke nets, bait traps and additional boat electrofishing. Further details on the amount of effort applied is provided in Chapter Three. Fyke nets and traps are useful passive fish collecting methods (Hubert 1996), particularly for small bodied species (Ruetz et al. 2007) and were deployed overnight as per Close et al. (2003). Fyke nets were set with the codend tied to a wood stake above the water line to allow an air pocket to ensure any trapped air-breathing bycatch to be released unharmed. The opening and wing of the net was set perpendicular to the bank with the wing positioned to ‘fish’ into the deeper water. Bait traps (baited with tinned cat food) were deployed in shallower water, ranging from enough water to cover the entrance of the trap to about 1.5 m
in depth. Bait traps were useful in both open water, bare substrates as well as in areas where macrophytes or woody debris was present.

Boat and backpack electrofishing were undertaken as a primary method of fish capture (see Chapters Three and Four). Electrofishing is based on the principal of fishes responding to electric current, in which the use of direct current (pulsed) enables a directional orientation and movement (galvanotaxis) towards the anode (+ve) (Kolz et al. 1998). Attraction to the anode can occur from several meters, particularly for boat electrofishing, where two Wisconsin-array anodes (SAA-6, Smith-Root Inc. Vancouver, Washington, USA) were used. Once fish are close to the anode (< 1 m), the fish enters narcosis and netted. Pulsed current was set at a higher frequency of 120 Hz, which is useful for collecting small or more active swimming fishes (Kolz et al. 1998). Power control settings were adjusted to cope with variations in water conductivity, but set to minimum levels needed to induce galvanotaxis to ensure fish recovered immediately upon capture. This was undertaken by trialling the electrofisher at adjacent areas nearby before undertaking sampling (Kolz et al. 1998). After capture, fishes were allowed to recover in aerated tubs for at least 30 mins prior to counting and measuring length and weight.

Backpack electrofishing was performed using a Smith-root® Model 15D backpack electrofisher (Smith-Root Inc. Vancouver, Washington, USA) with one operator and one assistant dipnetter. Electrofishing stopnets consisting of ca. 2 mm mesh, were deployed at the start and finish of the site. Triple passes were conducted to ensure accurate estimates of the most common species present and detection of rarer species (Pusey et al. 2006). A brief period of 10 mins between each successive pass was allowed to enable fine sediments suspended by wading to settle. Electrofishing was conducted by beginning at the most downstream point of the site and moving upstream.

Boat electrofishing was undertaken using a 4.5 m aluminum punt coupled with a Smith-root® Model 7.5 GPP electrofisher (Smith-Root Inc. Vancouver, Washington,
USA). Electrofishing was conducted in a series (10 per site) of 3 minute 'shots' of power-on (i.e. time the electrodes were energised). Ten shots enabled sampling to cover each site effectively (Close et al. 2003). Similar to back pack electrofishing, power control settings were adjusted with variations in water conductivity, but again, only set to minimum levels needed to induce galvanotaxis. There was one dip netter per vessel and fine mesh (ca. 3 mm) was used on all dip nets.

Artificial substrates, an underwater camera and snorkelling were trialled to determine fish spawning (e.g. parental guarder) and presence of oocytes. Artificial substrates consisted of polyvinyl chloride pipe and two configurations of small woody debris. Fish use of these artificial substrata for spawning and details on sampling gears and their deployment are provided in Chapter Five. Additionally, to examine the use of replaced LWD pieces for oviposition, a manipulation involving *Philypnodon grandiceps* oocytes was conducted to determine the likelihood of detecting the presence of oocytes using an underwater camera and snorkeling. However, these trials were unsuccessful and failed to reliably detect the presence of oocytes, so they were abandoned. Further details of these methods are found in Appendix Seven.
3. Chapter Three: Fish assemblage response to rehabilitation of a sand-slugged lowland river

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3.1. Abstract

The impact of excessive sediment supply on river channels has been described in many areas of the world. Sediment deposition disturbance alters habitat structure by decreasing channel depth, changing substrate composition and burying woody debris. River rehabilitation is occurring worldwide, but information is scant on fish assemblage responses to rehabilitation in sediment-disturbed lowland rivers. Sediment removal and large woody debris (LWD) replacement were used to experimentally rehabilitate habitat along a 1500 m stretch of the Glenelg River in western Victoria, Australia. Using an asymmetrical before-after control-impact (BACI) design, fish were captured before and after the reach was rehabilitated, from two control reaches and from a ‘higher quality’ reference reach. After two years post-rehabilitation monitoring, the fish assemblage at the rehabilitated reach did not differ from control reaches. Temporal changes in taxa richness and the abundance of *Philypnodon grandiceps*, *Nannoperca* spp. and three angling taxa occurred after rehabilitation (winter 2003) compared with the before period (winter 2002), but these effects did not differ between rehabilitated and control locations. Highest taxa richness and abundances occurred at the reference location. High salinity coincided with the timing of rehabilitation works, associated with low river discharges due to drought. The negative effects of other large-scale disturbances may have impaired the effectiveness of reach-scale rehabilitation or the effects of rehabilitation may take longer than two years to develop in a lowland river subjected to multiple environmental disturbances.

key words: fish habitat; sediment; BACI; restoration ecology; large woody debris; snags
3.2. Author Contributions

Travis Howson:
- Involved in project conception
- Design of field experiment and conducted fieldwork
- Conducted all statistical analyses
- Wrote the manuscript

Belinda Robson:
- Assisted in result interpretation
- Provided significant comments on draft manuscripts
- Assisted in editing the manuscript for publication

Brad Mitchell:
- Conceived the project in collaboration with Glenelg-Hopkins CMA and assisted in obtaining project funding
- Assisted in obtaining project animal ethics and sampling permits
- Provided comments on draft manuscripts
3.3. Introduction

Excessive loads of sand-size sediment particles exported into channels has simplified geomorphic complexity in southeastern Australian rivers (Erskine 1994, Rutherfurd and Budahazy 1996, Walling 1999, Rutherfurd and Bartley 2005) and rivers elsewhere in the world (Waters 1995, Shields et al. 2003). A variety of catchment-scale processes, including past vegetation clearing (Prosser et al. 2001), bushfire (Moody and Martin 2001) and mining (Ryan 1991, Walling 1999, Prosser et al. 2001) have all contributed to the deposition of sand in large volumes along reaches characterised by a low gradient or where stream power has been substantially reduced (Rutherfurd and Budahazy 1996, Prosser et al. 2001). This situation is exacerbated in rivers where flow and stream power have been reduced by impoundment; in such systems, multiple disturbances act simultaneously.

Large volumes of sand are stored in river channels, migrating downstream at slow rates as discrete ‘slugs’ and hence the term ‘sand slug’ is commonly applied (Bond and Lake 2005, Downes et al. 2006). Without intervention, sand slugs will last for centuries (Rutherfurd and Budahazy 1996, Prosser et al. 2001, Prosser et al. 2002), continuing to disturb rivers in a sustained way (O'Connor and Lake 1994). Sand-slugs may limit important components of freshwater fish habitat by creating relatively homogenous reaches with reduced depths and depth variation, increased width to depth ratio, reduced channel slope and little variety of cover because of woody debris, undercut banks and coarse substrata have all been buried by sand (Waters 1995). These impacts to river morphology are obvious; however, the response of biota to such impacts is less clear (Downes et al. 2006). Catchment managers faced with the possibility that these impacts are long lasting and potentially threatening to stream biota, are actively removing sediment in attempts to rehabilitate these rivers.

Often, the rationale for conducting river rehabilitation programmes is that recreating elements of habitat structure will yield positive responses from biota (field of dreams hypothesis, sensu Palmer et al. 1997). Recreating important fish
habitat patches such as pools in sediment-disturbed rivers, may yield a suitable test of this concept. However, there are few published examples of positive responses by fish assemblages to rehabilitation techniques in sand-disturbed rivers (Shields et al. 2006), probably because the applied techniques are still experimental (Bond and Lake 2005, Shields et al. 2006) or project evaluation is inadequate or non-existent (Lake 2001, Alexander and Allan 2006). Nevertheless, fish assemblage responses to some river rehabilitation techniques (e.g. large woody debris placement) can be rapid, occurring over time scales ranging from months to several years after the completion of works (Koehn 1987, Lehane et al. 2002, Nicol et al. 2002, Zika and Peter 2002) with large post-project increases in abundance often characteristic for large taxa such as salmonids (House and Boehne 1986, House 1996, Cederholm et al. 1997, Roni and Quinn 2001b, Lehane et al. 2002).

Therefore, we aimed to determine the responses of a lowland river fish assemblage to a river rehabilitation technique consisting of sediment removal and large woody debris replacement. The aim of the rehabilitation was to improve access for recreational users and to improve fish habitat for anglers by removing sand (and associated emergent macrophytes) and replacing large woody debris. Woody debris, a key component of river ecosystems, has long been associated with fish distributions in lowland river systems, playing several roles including refuge from natural disturbances (Harvey et al. 1999, Bond and Lake 2005), reducing predation risk (Everett and Ruiz 1993, Crook and Robertson 1999), assisting in spawning (Jackson 1978b, Merz 2001, Zimmer and Power 2006) and providing patches of high food abundance (Benke et al. 1985, O’Connor 1991). The addition of woody debris to sand-slugged rivers has been suggested to improve habitat structure for fish (Shields et al. 2003, Bond and Lake 2005). Therefore, at least for species with known associations with woody debris (e.g. Gadopsis marmoratus, Jackson, 1978; Bond & Lake 2003b; Nannoperca australis, Bond and Lake 2003b; Galaxias olidus, Bond and Lake 2003b; Anguilla australis, Koehn et al., 1994), a relatively rapid response to this rehabilitation technique was expected, consisting of increased taxa abundance and richness, and producing a different assemblage composition compared to very similar reaches where no rehabilitation has been applied. The individual responses
of some fish species to the rehabilitation were also assessed, including a group of species comprising ‘angling’ taxa that were anticipated to increase in density as a result of rehabilitation works.

3.4. Methods

3.4.1. Catchment description

The Glenelg River catchment is situated in far western Victoria, Australia (37° 30’ S, 143° 30’ E) and covers area of approximately 12 700 km² (Figure 3.1). The catchment climate is considered as a dry, Mediterranean-type, with long-term maximum air temperatures in the centre of the catchment ranging between 23.3 to 25.7 °C for summer and 4.5 to 5.2 °C during winter. The mean annual rainfall varies from 500 to 700 mm across the catchment; however, high rainfall (> 900 mm) occurs in the headwaters of the Glenelg and Wannon Rivers (Department of Water Resources 1989). A large water storage, Rocklands Reservoir, situated on the Glenelg River near the town of Balmoral, harvests water from the upper catchment and diverts it north to the Wimmera-Mallee region. Rocklands Reservoir has significantly altered the hydrology of the Glenelg River. The pre-dam mean annual discharge of 71 500 Ml has reduced to 22 000 Ml (ca. 70% reduction) since reservoir completion (Mitchell et al., 1996). Currently, an environmental flow regime has been established for the Glenelg River in an attempt to improve river health (Lind et al., 2007).

Agriculture dominates the catchment land use, with over two thirds of the Glenelg River catchment cleared of its vegetation predominantly for dryland grazing and cropping (Department of Water Resources 1989, Ierodiaconou et al. 2005). Land clearing has resulted in significant catchment changes, including severe land erosion and mass transport of soil into tributaries and mainstem reaches (Erskine 1994, Rutherfurd and Budahazy 1996). It is estimated that between 10 000 to 50 000 m³ of sediment (predominantly sand-size particles) per kilometre of channel (Rutherfurd and Budahazy 1996). The effect of sand is significant, transforming the
original ‘chain of pools’ channel structure to shallow, flat and over-widened bed forms (Rutherfurd and Budahazy 1996). Large pools that were once several metres in depth have now been in-filled and become long, shallow and homogenous channels, and other habitat components important to fish (e.g. woody debris, undercut banks) are partially or fully buried (T. Howson personal observation). Early attempts at sand management in the Glenelg River involved the removal of large woody debris along several sections to increase the channel capacity and enable greater flow through channels to move sediment (Erskine 1994, Erskine and Webb 2003). A few remnant large pools (up to 1 km long and 6 m deep) remain and perhaps provide refuge for flora and fauna, although some pools are affected by saline groundwater intrusion with high water conductivities over 10 000 μS cm\(^{-1}\) being recorded in pool bottom waters (McGuckin et al. 1991, Cameron and Jekabson 1992). Runs are the dominant type of connecting channel between pools and typically vary between 5 to 20 m in width, tens to hundreds of metres in length and between 0.3 to 1.2 m deep. In pools and runs, wood is dominated by river red gum (Eucalyptus camaldulensis, Dehnhardt), wattle (Acacia spp.) and tea-tree (Leptospermum spp.), and the macrophytes cumbungi (Typha spp.), common reed (Phragmites australis, [Cavanilles] Trinius ex. Steudel), blunt pondweed (Potamogeton ochreatus, Raoul) and triglochin (Triglochin procerum, Brown) are also present.

3.4.2. Rehabilitation procedure

The river rehabilitation procedure was commissioned by Glenelg-Hopkins Catchment Management Authority and undertaken by an environmental engineering firm (Earth Tech Pty. Ltd.) over a 1500 m length of the Glenelg River at the township of Harrow. Before rehabilitation, this reach consisted of a series of perennial pools connected by shallow runs that dried out during the summer period, leaving large sections of riverbed exposed. Rehabilitation works
Figure 3.1 The Glenelg River catchment, western Victoria, where Rehabilitated (square), Control 1 (triangle), Control 2 (inverted triangle) and Reference (diamond) sites were located. Stream gauge locations are also shown.
remodelled the entire 1500 m reach, primarily by excavating sediment to reconstruct deeper pools and to improve connectivity between pools by constructing a defined channel in the existing riverbed. After the removal of sediment, large woody debris (LWD) at an estimated load of 0.011 m$^3$ m$^{-2}$ was placed within the channel to create localised bed scour around LWD pieces, thereby keeping significant pool depth (Earth Tech 2002). The amount of wood added is similar to wood loadings reported for other river rehabilitation projects (Brooks et al. 2004, Nicol et al. 2004), and relative loadings recorded for several Victorian lowland rivers (Gippel et al. 1996a, Bond and Lake 2003a, Webb and Erskine 2005).

Different configurations of LWD were used including single whole red gum ($E.\ camaldulensis$) logs and rack-member (engineered log jam) wood jams (Earth Tech, 2002) (Figure 3.2). This was undertaken to increase habitat complexity whilst also facilitating the geomorphic function of woody debris (e.g. reduce bank erosion, assist pool formation, Abbe and Montgomery 1996, Brooks et al. 2004). Immediately above the rehabilitation reach, a sediment trap consisting of a 1500 m$^3$ hole (ca. 50 x 15 x 2 m deep) was excavated to further stop the upstream supply of sediment entering the reach and filling-in constructed pools. In addition, two barriers (vehicle crossings) that prevented water entering the floodplain were modified by lowering the existing crossing height, then crushed basalt rock inserted to stabilise the bed (Figure 3.2). A third crossing that spanned the main channel was similarly modified, but a rock-ramp type fishway consisting of a layered bed (5 x 8 m) of crushed basalt rock and large basalt boulders (150 to 600 mm in diameter) was also installed to assist fish access to the rehabilitated reach from downstream (Figure 3.2). In total, the rehabilitation procedure took several weeks to complete during February 2003. Reaches surrounding the rehabilitation reach were similar to the before-rehabilitation conditions and contained similar fish assemblages (Close et al., 2003), therefore we expected that fish would be easily able to recolonise the rehabilitated reach within the time frame of this project.
Figure 3.2 Map of the rehabilitated reach with photographs outlining the procedures undertaken: sediment extraction and construction of a sediment trap, woody debris replacement and modification of vehicle crossings. Photograph letters correspond to labelled points on diagram: A, sediment trap upstream end of rehabilitated reach; B, constructed rack-member log jam; C, piles of extracted sediment; D, F, modified vehicle crossings; E, placed single logs; G, bank protection logs and H, modified vehicle crossing with fishway. Photographs were taken at low flow during February 2003. Woody debris structures and vehicle crossings were inundated during subsequent fish sampling trips.
3.4.3. Experimental design

Designs constructed for the assessment of environmental impacts may be adopted for the assessment of rehabilitation procedures (Chapman 1999, Chapman and Underwood 2000, Lake 2001, Downes et al. 2002). In both cases, there is usually only a single ‘impact’ (rehabilitated) location, necessitating an asymmetrical design. Therefore, a before-after control-impact (BACI) type design (Green 1979) was used to assess the rehabilitation procedure, using before-period fish assemblage data collected by the Arthur Rylah Institute for Environmental Research (Close et al., 2003). Therefore, sites and sampling techniques were restricted to those used by Close et al. (2003). We have no reason to believe that these sites are not representative of this reach of the Glenelg River more generally. Two control locations were used, Five Mile (Control 1) and The Gorge (Control 2), both located upstream of the rehabilitated reach and separated from each other and the rehabilitated reach by more than 10 km (Figure 3.1). Finally, a reference reach on the Crawford River, a tributary of the Glenelg River, was included in the design, to serve as a higher quality standard to compare the rehabilitated reach after procedures were complete (Simenstad and Thom 1996, Chapman 1999, Grayson et al. 1999, Chapman and Underwood 2000, Lake 2001, Downes et al. 2002). The reference reach was relatively unimpacted by sediment and flow regulation, contained intact riparian and aquatic vegetation, and fish species present were identical to that of the mainstem Glenelg River.

3.4.4. Fish surveys

The before-period data consisted of two sets of samples taken during winter 2002. Each set of samples consisted of 10 bait traps (600 x 300 x 300 mm, 50 mm opening, bait: tinned cat food), up to 20 fyke nets (5 m wing, 600 mm opening, mesh size: 2 mm) and 10, 3 minute boat electrofishing shots (Smith-Root 7.5GPP, 120 Hz, 340–500 volts, pulsed DC, duty cycle: 10–20%). A variety of gear was used to maximise the number of species collected and to reduce selectivity by any one method. The number of nets set (range: 10–20) and net deployment time varied (range: 14–17 h.
overnight), so fish abundance was standardised to both the number of nets used and deployment time per site (catch per unit effort data) prior to analysis.

The after-period data consisted of two sets of samples in winter 2003 and again in winter 2004, using exactly the same type and number of gear within each site as was used in 2002. Each set of two samples was a minimum of four weeks apart. These data were also standardised to both the number of nets used and deployment time per site prior to analysis. The only difference in sampling methods between the before and after-periods was that boat electrofishing was only carried out once in the after-period. In addition, in the after-period, two sets of samples using fyke nets and bait traps (same effort, methodology as above) were collected in summer 2004 and summer 2005 to identify any seasonal response to the rehabilitation procedure (e.g. recruitment pulses), which may have been missed by only sampling in winter.

To analyse these data using a BACI design, we compared the before data to each set of after data, separately. As is usual with BACI designs, we used the two sets of samples within each season as replicates to generate variances upon which to test the significance of the before versus after rehabilitation treatment effect.

All fish sampling occurred in the littoral areas of river pools. Upon gear retrieval, collected individuals were identified to species with total length and weight recorded in the field. Note that we have assumed that the selectivity and capture efficiency of each gear type has remained constant over the life of the study.

3.4.5. Physicochemical and river discharge variables

During each sampling trip, river height was noted at the river gauges located immediately downstream of Rocklands Reservoir and at the townships of Balmoral and Harrow. Average daily discharge and electrical conductivity data for the water quality monitoring stations 238224 (Glenelg River at Fulham’s Bridge), 238210 (Glenelg River at Harrow) and 238235 (Crawford River at Lower Crawford) were
obtained from the Victorian Water Resources Warehouse database and Glenelg-Hopkins Catchment Management Authority.

A physiochemical depth profile was taken at approximately 10:00 am on the morning before the hauling of nets and traps. Several variables were recorded including temperature, conductivity and the concentration of dissolved oxygen, pH and salinity. Each variable was recorded at 0.5 m depth increments from the surface to the bottom along with the maximum depth of each profile.

3.4.6. Statistical analysis

Catches from bait traps and fyke nets were pooled to provide a single estimate of catch-per-unit-effort from passive gear at each site. In addition, the three species from the genus *Nannoperca*, namely *Nannoperca australis*, *Nannoperca variegata* and *Nannoperca obscura*, were pooled to form the category *Nannoperca* spp. as numbers of individuals caught were low. The targeted angling species category comprised *Gadopsis marmoratus*, *Perca fluviatilis* and *Tinca tinca*. After the application of a fourth root transformation, assumptions of heteroscedasticity and normality were met for all analyses. Among group differences in fish assemblage composition, taxa richness and abundance was examined using constructed asymmetrical analysis of variance models. For assemblage composition, Bray-Curtis dissimilarity was selected for the similarity measure using the program PERMANOVA (Anderson 2005) to construct models and to partition group variation. Multivariate dispersion was assessed visually to ensure exchangeability among sample groups.

Boat electrofishing samples were analysed separately using the same groupings of dependent variables as for the passive gear (above), with the exception that sufficient numbers of *G. marmoratus* were captured for them to be analysed separately. Multivariate analyses were not carried out for boat electrofishing data, but the same ANOVA models (described below) were used to analyse these data and those from the passive gear.
The asymmetrical BACI design used in this study was calculated using the instructions outlined in Underwood (1992, 1996) and is constructed from two factorial analysis of variance models. In this study, the first model encompasses all the variation with the terms: Times (fixed, 2 levels: Before and After), Locations (random, 3 levels: Control 1, Control 2 and Rehabilitated) and Times × Locations (random, 6 levels). The second model is constructed from the control data with similar terms: Times (fixed, 2 levels: Before and After), Control locations (random, 2 levels: Control 1 and Control 2) and Times × Control locations (random, 4 levels). Here the term Times is designated as a fixed factor as there can only be two periods, before and after the rehabilitation procedure. The before-period contained one time period (winter 2002); however, several separate time periods were classified in the after-period (winter 2003, summer 2004, winter 2004 and summer 2005). Therefore, several asymmetrical models were constructed for each possible before-after period comparison. The two sets of samples within each time period provided the error variance. Where possible, post-hoc pooling of the Time × Control location and MS_residual were undertaken (Underwood, 1992) to provide a more powerful test of the Times × Rehabilitated term. To alleviate the potential problem of increased likelihood of Type-II errors, the Times × Control location term was only combined with the Residual when differences between the MS_time × Control location and MS_residual were $p > 0.25$ (Winer et al. 1991, Glasby 1997, Underwood 1997, Quinn and Keough 2002). For all comparisons, significance levels were set at $\alpha = 0.10$, to reduce the chance of making a Type-II error, which is of greater concern than the Type-I error rate in this case (Downes et al., 2002).

Spatiotemporal assemblage composition patterns were also examined using non-metric multidimensional scaling (nMDS), again using fourth root transformed data and Bray–Curtis dissimilarity as the distance measure. Differences between a priori groups (After-period winter versus summer) were tested using analysis of similarities (ANOSIM) with species contributing to assemblage composition discriminated using the similarity percentages (SIMPER) routine in the package PRIMER® (Plymouth Routines In Multivariate Ecological Research, version 5.0, PRIMER-E Ltd.).
Variation in the values of several physiochemical variables between seasons and locations was expected due to intrinsic relationships with temperature and flow events. As no before-period data existed, before-after water quality comparisons could not be made, but after-period samples were useful for gauging responses of fish to reach treatments. From the water column profile, only data from the surface to the three-metre depth was included in analyses as these were the depths over which the fishing gear was deployed. Means were calculated for each variable across locations and times, with a natural log transformation applied to improve data normality. Factorial analysis of variance models were constructed for each variable using the factors location (random, 4 levels; Control 1, Control 2, Rehabilitated and Reference) and Season (fixed, 2 levels; winter and summer). A principal components analysis (PCA) using Euclidian distance was used to identify which variables and samples were contributing to patterns. Finally, the multivariate relationship between the physiochemical and fish similarity matrices was explored using a permutation test (RELATE function in the program PRIMER®)(Clarke and Gorley 2001).

3.5. Results

3.5.1. River discharge

River discharge varied seasonally across the mid-upper Glenelg (controls and rehabilitated sites) and Crawford (reference site) rivers (Figure 3.3). Maximum annual discharge occurred during late winter–spring, reducing to base flow discharges during summer. Higher average daily flows were observed for the Crawford River during winter–spring 2002, and 2004, but discharge was similar between the two rivers in winter–spring 2003. For both rivers, gradual increases in the winter–spring discharges occurred from the before-rehabilitation to the after-rehabilitation periods. The collection of fish samples during the before period was undertaken in an unusually low flow year and an extreme low flow event occurred concurrently with the undertaking of rehabilitation works (Figure 3.3).
Figure 3.3 Recorded average daily flow for the station 238224 on the Glenelg River (black line) and station 238235 on the (unregulated) Crawford River (grey line) from the period of August 1990 to August 2005. River discharge has been log10 transformed. The broken line indicates the completion of rehabilitated site works; arrows indicate individual fish sampling trips.

3.5.2. Physicochemical variables

Principal components analysis successfully separated patterns among the physicochemical variables, with the first two principal components explaining 88.4% of the variation (Figure 3.4). Further examination of the eigenvectors revealed temperature positively and dissolved oxygen negatively correlated with component 1, while conductivity negatively and pH positively correlated with component 2. The first principal component displayed a seasonal axis with significantly higher temperatures ($F_{1, 3} = 478.500, p < 0.001$) encountered across all sites during summer periods. Lower temperatures during winter also corresponded with significantly higher dissolved oxygen concentrations ($F_{1, 3} = 20.244, p < 0.05$) across all locations. The second principal component axis represents location differences in conductivity and pH with significantly lower conductivity values ($F_{3, 3} = 22.838, p < 0.05$) and
higher pH values ($F_{3, 3} = 9.000, p < 0.05$) at the reference compared to control and rehabilitated locations, but no difference between control and rehabilitated sites. PCA revealed outlying rehabilitated site and control site samples characterised by high conductivity values for the first after-period during winter 2003 (Figure 3.4) when river discharge was quite low. Furthermore, gauging data (238210 Glenelg River at Harrow) also confirmed high electrical conductivity values from December 2002.

![Figure 3.4](image)

**Figure 3.4** Principal component analysis (fourth root transformed data) of the factors temperature, dissolved oxygen, pH and specific conductivity ($\mu$S cm$^{-1}$ at 25 °C) for samples collected during the after-rehabilitation period. Symbols represent locations with diamond = Reference, triangle = Control 1, inverted triangle = Control 2 and square = Rehabilitated location.
June 2003 (Figure 3.5) suggesting that rehabilitation works occurred when elevated salinities existed. The small but significant ($Rho = 0.207, p < 0.01$) multivariate relationship between the physicochemical variables and the fish assemblage data indicated that fish distributions were partially related to physiochemical conditions.

**Figure 3.5** Average daily discharge (black line) and specific conductivity (grey line) recorded over the period of December 2002 to July 2005 at the rehabilitated reach (station 238210). Electrical conductivity was monitored at 0.8 m depth from the surface at the common river stage height of 0.44 m. The broken line indicates the completion of rehabilitated site works; arrows indicate individual fish sampling trips.
3.5.3. Fish surveys

Fish surveys collected a total of 16 species (Table 3.1) from 10 families; many of these species were not abundant and were patchily distributed across locations and times. Six species accounted for 83% of the catch: *Philypnodon grandiceps*, *Perca fluviatilis*, *Gadopsis marmoratus*, *Tinca tinca*, *Nannoperca australis*, *Nannoperca variegata* and *Nannoperca obscura*, where *P. grandiceps* was the most abundant species contributing to 61% of the overall catch for passive gear. The same six species comprised 71% of the assemblage in the boat electrofishing samples, with *P. grandiceps* only contributing to 37% of the catch composition. The dominance of *P. grandiceps* in samples is probably explained by selectivities in gear types. Interestingly, *Galaxias maculatus* was only captured at the Reference location, even though they are known to be present in the upper reaches of the Glenelg River (T. Howson unpublished data). Species contribution to the assemblage was similar for both passive gear and electrofishing (Table 3.1).

3.5.4. Influence of the rehabilitation procedure on fish assemblage composition

Non-metric multidimensional scaling indicated substantial temporal and spatial variability among samples (Figure 3.6). Differences in assemblage composition were observed between the before-period winter 2002 and the after-period winter 2003, summer 2004 and summer 2005. However, considerable trip-to-trip variation among locations within time periods obscured the detection of significant Time, Location or Time × Location terms (Table 3.2). Seasonal variation in assemblage composition was prominent (Global R = 0.242, p = 0.001) and similarity percentages revealed high summer abundances of two gudgeon species, *Philypnodon grandiceps* and *Hypseleotris* spp., as largely responsible for (35%) these seasonal differences. Further, the spatial separation of rehabilitated and reference samples is much clearer during summer compared to winter (Figure 3.6).
Table 3.1 Fish taxa captured in Glenelg and Crawford river pools using fyke nets, bait traps and electrofishing from April 2002 to May 2005.

<table>
<thead>
<tr>
<th>Order</th>
<th>Family</th>
<th>Species</th>
<th>Common name</th>
<th>Number captured (passive gear)</th>
<th>Number captured (boat electrofishing)</th>
<th>Total number of individuals captured</th>
</tr>
</thead>
<tbody>
<tr>
<td>Anguilliformes</td>
<td>Anguillidae</td>
<td>Anguilla australis</td>
<td>Short-finned eel</td>
<td>4 (0.3)</td>
<td>2 (0.4)</td>
<td>6</td>
</tr>
<tr>
<td>Perciformes</td>
<td>Bovichtidae</td>
<td>Pseudaphritis urvillii</td>
<td>Tupong</td>
<td>0 (0.0)</td>
<td>2 (0.4)</td>
<td>2</td>
</tr>
<tr>
<td>Cypriniformes</td>
<td>Cyprinidae</td>
<td>Tinca tinca</td>
<td>Tench</td>
<td>15 (1.2)</td>
<td>21 (4.7)</td>
<td>36</td>
</tr>
<tr>
<td>Perciformes</td>
<td>Eleotridae</td>
<td>Hypseleotris spp.</td>
<td>Carp gudgeon</td>
<td>70 (5.8)</td>
<td>1 (0.2)</td>
<td>71</td>
</tr>
<tr>
<td>Perciformes</td>
<td>Eleotridae</td>
<td>Philypnodon grandiceps</td>
<td>Flathead gudgeon</td>
<td>852 (70.6)</td>
<td>160 (35.5)</td>
<td>1012</td>
</tr>
<tr>
<td>Perciformes</td>
<td>Gadopsidae</td>
<td>Gadopsis marmoratus</td>
<td>River blackfish</td>
<td>110 (9.1)</td>
<td>91 (20.2)</td>
<td>201</td>
</tr>
<tr>
<td>Salmoniformes</td>
<td>Galaxiidae</td>
<td>Galaxias maculatus</td>
<td>Common galaxias</td>
<td>29 (2.4)</td>
<td>82 (18.2)</td>
<td>111</td>
</tr>
<tr>
<td>Salmoniformes</td>
<td>Galaxiidae</td>
<td>Galaxias olidus</td>
<td>Mountain galaxias</td>
<td>1 (0.1)</td>
<td>0 (0.0)</td>
<td>1</td>
</tr>
<tr>
<td>Salmoniformes</td>
<td>Galaxiidae</td>
<td>Galaxiella pusilla</td>
<td>Dwarf galaxias</td>
<td>1 (0.1)</td>
<td>0 (0.0)</td>
<td>1</td>
</tr>
<tr>
<td>Petromyzontiformes</td>
<td>Geotriidae</td>
<td>Mordacia mordax</td>
<td>Short-headed lamprey</td>
<td>0 (0.0)</td>
<td>1 (0.2)</td>
<td>1</td>
</tr>
<tr>
<td>Perciformes</td>
<td>Nannopercaeida</td>
<td>Nannoperca australis</td>
<td>Southern pygmy perch</td>
<td>10 (0.8)</td>
<td>20 (4.4)</td>
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<td>Perciformes</td>
<td>Nannopercaeida</td>
<td>Nannoperca obscura</td>
<td>Yarra pygmy perch</td>
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<td>8 (1.8)</td>
<td>37</td>
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<td>Perciformes</td>
<td>Nannopercaeida</td>
<td>Nannoperca variegata</td>
<td>Variegated pygmy perch</td>
<td>11 (0.9)</td>
<td>21 (4.7)</td>
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<tr>
<td>Perciformes</td>
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<td>Perca fluviatilis</td>
<td>Redfin</td>
<td>29 (2.4)</td>
<td>11 (2.4)</td>
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<td>Osmeriformes</td>
<td>Retropinnidae</td>
<td>Retropinna semoni</td>
<td>Australian smelt</td>
<td>2 (0.2)</td>
<td>17 (3.8)</td>
<td>19</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Total 1207 (100)</td>
<td>451 (100)</td>
<td>1658</td>
</tr>
</tbody>
</table>

Note: The proportion of individuals captured expressed as a percentage of the total catch is detailed in parentheses.
Table 3.2 Variation in fish assemblage composition data collected using combined fyke net and bait traps as detected by permutated multifactor analysis of variance conducted on fourth root transformed species abundance data, Bray–Curtis dissimilarity distance measure and no sample standardisation.

<table>
<thead>
<tr>
<th>Source</th>
<th>Win 02 versus Win 03</th>
<th>Win 02 versus Su 04</th>
<th>Win 02 versus Win 04</th>
<th>Win 02 versus Su 05</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>DF</td>
<td>MS</td>
<td>F</td>
<td>DF</td>
</tr>
<tr>
<td>Time</td>
<td>1</td>
<td>3083.132</td>
<td>3.948</td>
<td>1</td>
</tr>
<tr>
<td>Location</td>
<td>2</td>
<td>1265.105</td>
<td>1.620</td>
<td>2</td>
</tr>
<tr>
<td>Rehabilitated versus Controls</td>
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<td>1.168</td>
<td>1</td>
</tr>
<tr>
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<td>1618.185</td>
<td>2.299</td>
<td>1</td>
</tr>
<tr>
<td>Time × Location</td>
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<td>1.192</td>
<td>2</td>
</tr>
<tr>
<td>Time × Rehabilitated</td>
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<td>857.846</td>
<td>1.296†</td>
<td>1</td>
</tr>
<tr>
<td>Time × Control</td>
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<td>703.860</td>
<td>1.074</td>
<td>1</td>
</tr>
<tr>
<td>Error</td>
<td>6</td>
<td>655.180</td>
<td>6</td>
<td>552.884</td>
</tr>
</tbody>
</table>

Note: † denotes the F ratio is tested using a pooled Time × Control–Residual term with 1, 7 df, when Time × Control is tested at \( p > 0.25 \). The before period sample was compared to each after period sample individually to determine before-after period changes in assemblage composition.
Figure 3.6 Non-metric multidimensional scaling plot (fourth root transformed Bray–Curtis dissimilarity) of fish samples from combined fyke nets and bait traps. Symbols represent locations with diamonds = Reference, triangles = Control 1, inverted triangles = Control 2 and squares = Rehabilitated locations. Symbol shades and fill represent different temporal periods with closed dark grey = winter 2002 (before period), closed light grey = winter 2003 (after period), open black = summer 2004 (after period), closed black = winter 2004 (after period) and open light grey = summer 2005 (after period).

3.5.5. Effects of rehabilitation on species richness and abundance: passive gear samples

Species richness varied among locations, over the before-after periods (Figure 3.7a). Significant changes occurred between locations from winter 2002 to winter 2003 (Times × Location; $F_{2, 6} = 5.250, p < 0.1$) reflecting before-period separation among control locations (Times × Control; $F_{1, 6} = 8.100, p < 0.05$). After the completion of rehabilitation procedures, all locations became more similar to each other throughout the two years of after-period sampling (Figure 3.7a). Even though species richness increased over the after-period, there was no evidence of the rehabilitated location demonstrating higher species richness compared to
unmodified control reaches. At the reference site, species richness varied greatly and was highest during summer. Highest species richness was expected at the Reference location, but on three occasions (winter 2003, summer 2004 and summer 2005) it was more similar to control locations.

![Graphs](image)

**Figure 3.7** Patterns of mean: (a) species richness (untransformed data), (b) total abundance (fourth root transformed data), (c) angling taxa abundance (fourth root transformed data) and (d) *Nannoperca* spp. (fourth root transformed data) varying across the factors location (Control 1, Control 2, Rehabilitated & Reference) and time (winter 2002, winter 2003, summer 2004, winter 2004 and summer 2005). Passive gear sampling. Individual points are means with error bars representing ±1 standard error.
Total abundance of all species captured did not significantly differ among control and rehabilitated locations, nor between any before and after period, although the rehabilitated reach contained the lowest mean abundances of all sites (Figure 3.7b). The reference location only showed the highest total abundances during summer 2004 (Figure 3.7b).

Although *Philypnodon grandiceps* was the most abundant species captured, there was no evidence that this species responded to the rehabilitation procedure. The abundance of angling taxa changed strongly between before-after periods, but did not vary between control and rehabilitated locations (Figure 3.7c). After-period decreases in angling species abundance occurred for winter 2003 ($F_{1, 2} = 30.277, p < 0.1$), summer 2004 ($F_{1, 2} = 81.75, p < 0.1$) and winter 2004 ($F_{1, 2} = 10.866, p < 0.1$) suggesting that river rehabilitation did not ‘attract’ larger species. Again, the Reference location was predicted to contain the highest values for angling taxa abundance; this was true for the summer 2004 and 2005 samples, but it had the lowest abundances in winter 2004.

The abundance of *Nannoperca* spp. did not differ significantly between the before-and after-periods at the rehabilitated site (Figure 3.7d). Higher *Nannoperca* spp. abundances were observed at control and reference locations; however, their distribution among times and locations was patchy. Abundances at both controls significantly increased (Times × Control, $F_{1, 6} = 28.000, p < 0.05$) during summer 2005 with a particularly large increase observed at Control site 1.

3.5.6. Effects of rehabilitation on species richness and abundance: boat electrofishing samples

Boat electrofishing collected 14 species, including the two larger species *Mordacia mordax* (short-headed lamprey) and *Pseudaphritis urvillii* (tupong) that were not found during netting surveys. Species richness did not change between the before-and after-periods at the rehabilitated location (Figure 3.8a), but varied among control locations over the before-after period ($F_{1, 3} = 6.255, p < 0.1$) as a result of an after-period increase in species richness at Control site 1. As expected, boat
electrofishing revealed highest species richness at the Reference location where an average of 10 species was collected.

Total abundance significantly increased \( (F_{1, 2} = 148.75, p < 0.05) \) in the after-period, although it did not differ between control and rehabilitated locations. Highest fish numbers occurred at the reference site (Figure 3.8b). The abundance of angling species differed spatially and temporally \( (\text{Times} \times \text{Location}, F_{2, 3} = 8.848, p < 0.1) \) with planned comparisons indicating a temporal change between rehabilitated and control locations \( (F_{1, 4} = 9.565, p < 0.05) \). During the after-period, higher abundances were observed at the reference and control locations compared to the rehabilitated site, perhaps indicating that the rehabilitation procedure limited increases in angling species abundance (Figure 3.8c). \textit{Gadopsis marmoratus} abundance varied significantly \( (F_{2, 3} = 58.33, p < 0.05) \) among locations over the before-after period (Figure 3.8d). However, this change reflected differences between control locations \( (\text{Times} \times \text{Control}, F_{1, 3} = 111.667, p < 0.05) \), rather than between controls and the rehabilitated site.

A clear temporal increase in \textit{Nannoperca} spp. abundance occurred across all locations during the after-period (Figure 3.8e). \textit{Nannoperca} species abundances did not differ across control and rehabilitated locations \( (F_{1, 2} = 0.357, p > 0.1) \) but the reference reach contained the highest abundances. Boat electrofishing produced differing results to that of the passive gear for \textit{Nannoperca} spp. at Control 1 (before-period) and rehabilitated locations (after-period) as actively searching sites with electrofishing captured higher numbers across some locations.

For \textit{Philypnodon grandiceps}, boat electrofishing only yielded a before-after temporal effect \( (F_{1, 2} = 35.85, p < 0.05) \) with higher abundances observed across the rehabilitated and control locations during the after period (Figure 3.8f). Interestingly, differences between the reference site and other locations were non-significant for this species, unlike observations for other species.
Figure 3.8 Patterns of mean: (a) species richness (untransformed data), (b) total abundance (fourth root transformed data), (c) angling taxa abundance (fourth root transformed data), (d) *Gadopsis marmoratus* abundance (untransformed data), (e) *Nannoperca* spp. Abundance (fourth root transformed data) and (f) *Philypnodon grandiceps* (fourth root transformed data) varying across the factors location (Control 1, Control 2, Rehabilitated & Reference) and time (before: April and July 2002, after: May 2005) for boat electrofishing samples. Individual points are means with error bars representing ± 1 standard error.
3.6. Discussion

3.6.1. Detecting the response of fish assemblage structure to habitat rehabilitation

We anticipated spatial and temporal changes in fish assemblage structure as a result of the rehabilitation measures, consisting of increased abundance of those fish species with a known dependence on woody debris and increased species richness. However, rehabilitation did not yield a clear positive response at the individual species level, nor at the assemblage-level during two years of post rehabilitation sampling. It is difficult to know whether this pattern will continue over the longer term or whether fish abundance and species richness at the rehabilitated reach will increase over subsequent years.

The comparatively short time frame of this study is one of several limitations to detecting a response by the fish assemblage to rehabilitation. We still know relatively little about the capacity for dispersal in Australian native fish that would enable them to colonise rehabilitated sites or how long this might take. For example, one recent radiotracking study shows that *Gadopsis marmoratus* may be more mobile than previously thought (Koster and Crook, 2007). Therefore, there is considerable uncertainty as to whether barriers to movement external to rehabilitated sites limit their colonisation or whether the spatial extent of the rehabilitated site (1500 m length in this case) is sufficient to induce a response. In the present case, the limited differences among the control, rehabilitated and reference sites may indicate that the fish assemblage had not been greatly affected by sand slugs. Conversely, it may also indicate that modifying 1500 m of river length by extracting sand and reinstating pools and woody debris is insufficient to increase fish abundances within the larger context of several hundred kilometres of sand-slugged river. This is despite the fact that some of the species we recorded are known to respond positively to the presence of woody debris (Jackson, 1978; Koehn et al., 1994; Bond and Lake, 2003b). If 1500 m is too small a length of river for this type of rehabilitation, it does not bode well for the future because of the prohibitive cost of remodelling longer lengths of river.
The assemblage of native fish in the Glenelg River is diverse (for southern Australia) despite the presence of exotic species. More diverse assemblages may have a greater capacity to respond to environmental change (Elmqvist et al. 2003) including rehabilitation. Our use of a single reference site is a limitation in this regard as we could not assess the levels of variation in sites unimpacted by sediment slugs and flow regulation. However, the detection of temporal trends in fish species richness and abundance, especially before versus after-rehabilitation, shows that our design had sufficient statistical power to detect changes in the fish assemblage (of the order of three species or 0.4 fish caught per hour). Therefore, we would have detected any changes in community structure of rehabilitated sites, had they occurred. We conclude that the rehabilitation procedure documented here was not associated with detectable increases in the abundances of angling or other fish species, but may be so in the future (requiring further sampling).

Several studies have shown reach or river-scale fish responses to the placement of instream structures (Gowan and Fausch 1996, Lehane et al. 2002, Zika and Peter 2002, Brooks et al. 2004). Such results may reflect well-connected rivers, containing species that are highly mobile (e.g. salmonids, Gore and Shields 1995, Gowan and Fausch 1996) in situations where habitat is the key limiting factor. We could not detect a rapid response from the fish assemblage at the reach level and there are other recent examples reporting the failures of such programmes for fish (Pretty et al. 2003, Bond and Lake 2005, Thompson 2006) and macroinvertebrates (Larson et al. 2001, Harrison et al. 2004, Lepori et al. 2005). The reasons for failure vary widely (Frissell and Nawa 1992, Minns et al. 1996, Bond and Lake 2003b, Harrison et al. 2004) but one clear trend is the continuing influence of simultaneous large scale disturbances (e.g. flood and drought) and degradation (e.g. riparian vegetation removal, degraded water quality) (Mitchell et al. 1996, Larson et al. 2001, Pretty et al. 2003, Harrison et al. 2004, Bond and Lake 2005, Cottingham et al. 2005, Lepori et al. 2005) that limit the capacity of the fauna to respond to rehabilitation of a relatively small area of habitat. In the present study, an extended drought and associated changes to river hydrology and water chemistry, especially salinity (e.g.
Lind et al., 2006) coincided with the application of the rehabilitation procedure and with significant decreases in both abundance and species richness of fish, suggesting that these effects may have limited the capacity of the fish assemblage to respond to rehabilitation. However, fish may also have redispersed to other unsampled patches (Minns et al. 1996, Bond and Lake 2003b) for biological reasons (e.g. spawning).

Detection of a small, significant relationship between fish assemblage composition and water quality values may reflect selection or avoidance of better or worse quality patches, indicating the potential for salinity to impact fish assemblages as well as invertebrates in the Glenelg River (Close et al. 2003). Although, the adults of many of the fish species (except Gadopsis marmoratus) are tolerant of salinity concentrations above 10 g L$^{-1}$ (ca. 14700 μS cm$^{-1}$) (Clunie et al. 2002, Hart et al. 2003) the critical, early life-history stages are less tolerant (Hart et al. 1991, Clunie et al. 2002, James et al. 2003, Nielsen et al. 2003). The effects of salinity may thereby limit the capacity of the fish assemblage to respond to rehabilitation.

Consideration of other large scale confounding factors such as angling is also important. It is possible that angling removed fish attracted to the rehabilitation site thereby obscuring the effect. Gowan and Fausch (1996) and Thompson (2006) discuss the importance of recording information on angler use of rivers surrounding control and rehabilitated locations, a component often absent from current assessment of programmes. Recreational fishing was observed at sites on a few occasions, but we did not collect quantitative information on angling effort. Informal discussions with local residents revealed that the rehabilitation site is targeted for Perca fluviatilis and Gadopsis marmoratus after significant flow events, during spring and summer periods. Therefore, it is possible that any increased standing crop of these fish was removed by anglers.

The present study shows the difficulties of making effective assessments of rehabilitation even with before-treatment data and more than a single year of post-treatment data. While design improvements may assist, ultimately the problem lies
with using survey-based data to disentangle the effects of multiple impacts operating at multiple scales within the context of seasonal variability. Instead, direct evidence of the modification of ecological processes by rehabilitation works, such as an increased food supply that is consumed by fish, is needed to substantiate the positive effects of rehabilitation.

3.6.2. Lessons for future attempts to rehabilitate fish habitat in sand-slugged rivers

Established relationships between fish and habitat structure, along with high community willingness to combat river degradation, will ensure the continuing popularity of habitat rehabilitation programmes. However, the current extent of river degradation, particularly sediment deposition, means that many systems are still experiencing stress even after problems have long been identified (Karr and Chu 1999). Many disturbances are difficult to stem and occur concurrently, and rehabilitation efforts are therefore being attempted under conditions of continuing environmental degradation. In the context of the present study, the press disturbance of habitat loss may be addressed via rehabilitation programmes, but this approach will not address the simultaneous press disturbance of flow modification and the pulse disturbance of salinity (sensu Lake 2000). It may be unrealistic to expect present rehabilitation strategies focused solely on habitat reinstatement to address problems in systems with multiple disturbances, particularly within short time frames (Ziemer 1999).

In considering the role of habitat modification towards alleviating environmental degradation, it is crucial that rehabilitation strategies aiming to modify or create ‘new habitat patches’ in sediment-disturbed rivers are viewed for what they are: small areas nested within a backdrop of a larger landscape. Thus, an assessment of the influence of the larger surrounding environment on habitat function is critical during planning; especially the impact of continuing stressors and disturbances across the landscape (Bond and Lake 2005) that may swamp expected rehabilitation effects.
Improving the process of successful rehabilitation in sand-slugged rivers will depend on the *a priori* knowledge obtained about the system (Rutherfurd et al. 1999a). Although the multiple components which contribute to the habitat have been identified, links between component quantity and arrangement in space (i.e. how much and where to put it, in what shape?) on assemblage structure are not well understood. It is imperative that these questions are addressed as they can increase the efficiency of the effort placed into rehabilitation (e.g. to place all resources into one reach or spread over multiple reaches?). A mechanistic understanding of the impact and the level of degradation influencing biota is required prior to conducting rehabilitation as this will set the boundaries for the rehabilitation works and will identify the spatial and temporal scales required for a monitoring programme (Hobbs and Norton 1996, Chapman and Underwood 2000, Lake 2001, Downes et al. 2002, Bond and Lake 2003a). As has been found for stream rehabilitation aimed at improving macroinvertebrate communities (Brooks et al. 2002), simply rebuilding habitat for fish in lowland rivers does not necessarily mean they will come.
4. Chapter 4: Size and quantity of woody debris affects fish assemblages in a sediment-disturbed lowland river

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4.1. Abstract

Responses by fish assemblages to individual restoration actions among a suite of channel modifications are not well understood. We investigated whether increasing woody debris abundance, without significant change to channel morphology, would increase native fish abundance and species richness in a sediment-disturbed river channel (Glenelg River, Victoria, Australia). We conducted a Before–After, Control-impact design experiment at twelve locations containing either a high (n = 6) or low (n = 6) quantity of large woody debris (LWD). We added small woody debris (SWD) to half (n = 6: 3 high, 3 low LWD densities) of the locations to increase woody debris complexity without the impacts on channel morphology associated with LWD manipulations. Fish species richness and abundance was quantified using electrofishing surveys before (4 sampling trips) and after (3 sampling trips) SWD addition. Fish species richness was not associated with high or low quantities of LWD or with types of woody debris (LWD or SWD). Addition of SWD altered fish assemblage composition but the effect depended on LWD quantity. SWD additions to locations with low LWD quantities increased abundance of two, wood affiliated species: *Philypnodon grandiceps* and *Gadopsis marmoratus*. SWD additions to locations with high LWD quantities increased abundance of *P. grandiceps* and
**Galaxias olidus.** Fish body size was important in detecting a response to added SWD because for two species, only certain size classes responded: adults of *P. grandiceps* (> 50 mm TL) and juveniles of *G. marmoratus* (< 123 mm TL). Fish assemblages responded positively to increased density of SWD through local increases in abundance, despite channel sedimentation. Unlike LWD, SWD is relatively cheap to place in rivers because it does not require heavy machinery and can be obtained without tree mortality. The use of SWD to assist in habitat restoration, especially for small species of native fish and juvenile fish, should be considered as a strategy in river restoration.

Keywords: Coarse woody debris; Habitat complexity; Large woody debris; Restoration ecology; Sedimentation; Small woody debris
4.2. Author Contributions

Travis Howson:
- Design the field experiment and conducted fieldwork
- Conducted all statistical analyses
- Wrote the manuscript

Belinda Robson:
- Provided significant comments on draft manuscripts
- Assisted in extensive editing the manuscript for publication

Ty Matthews:
- Provided significant comments on draft manuscripts
- Assisted in editing the manuscript for publication

Brad Mitchell:
- Provided comments on draft manuscripts
- Assisted in gaining project animal ethics and sampling permits
4.3. Introduction

Restoration of fish habitat often focuses on modifying river channels by adding artificial structures, usually to address habitat changes from disturbances such as excessive sediment deposits (Bond and Lake 2005, Howson et al. 2009, 2010), channelization (Pretty et al. 2003) or channel erosion (Shields et al. 2006). Frequently, large pieces of woody debris (LWD, > 0.1 m in diameter) are added to rivers, reflecting its usefulness for reinstating key channel features when natural LWD loads are reduced (Kauffman et al. 1997, Erskine and Webb 2003). Higher fish abundances may then result from changes to channel depth or substrate dynamics (Montgomery et al. 2003a) or river velocity (McMahon and Hartman 1989). Fish may also use LWD directly for cover (Crook and Robertson 1999), as a refuge from disturbance (Bond and Lake 2005) or for oviposition (Howson et al. 2010). Thus, replacing LWD could initiate various responses at different times, leading to different recovery pathways and highlighting the need to identify which habitat alterations shape fish assemblages.

Despite its benefits, LWD is a scarce resource shared with terrestrial ecosystems (Harmon et al. 1986). Limited availability and the high cost of installation may restrict the extent of restoration undertaken, particularly if many pieces are required to recreate natural debris jams. Therefore, increased understanding of the relationship between wood quantity, complexity of arrangement and fish assemblages is needed to maximise the effectiveness of river restoration using LWD. This includes identifying alternative sustainable wood resources that are suitable for fish. In particular, small woody debris (SWD) is logistically and financially easier to place in rivers and streams (Lester et al. 2006) and may deliver ecological benefits for fish.

Small pieces of woody debris may be more important for fish than is presently recognised. ‘Small’ or ‘fine’ woody debris are defined as pieces ranging from 0.01 to 0.1 m in diameter (Triska and Cromack 1980, Culp et al. 1996, Giannico 2000, Wallace et al. 2000). It dominates counts in wood surveys, especially in streams.
draining old growth forest (Bilby and Ward 1991) or lands deforested of larger trees (Wallace et al. 2000). In rivers, SWD retains leaf litter and fine organic matter (Bilby and Ward 1989, Muotka and Laasonen 2002) and supports high macroinvertebrate densities (Lester et al. 2007, Schneider and Winemiller 2008). Subsequently, restricting SWD availability can result in significant decreases (47-50%) in macroinvertebrate abundance, biomass and productivity (Wallace et al. 1999). However, the importance of SWD to fish in rivers is relatively unknown, as most fish studies have focussed on LWD (Culp et al. 1996, Monzyk et al. 1997). Fish have been observed in reaches containing SWD (Culp et al. 1996, Monzyk et al. 1997, Welcomme 2002, Bond and Lake 2003a) and experiments isolating SWD as cover have demonstrated positive associations for single species in simple artificial stream channels (Lonzarich and Quinn 1995, Spalding et al. 1995), or small upland streams (Culp et al. 1996, Giannico 2000). Schneider and Winemiller (2008) found that small fish species responded to added SWD in lowland reaches, but this depended on location. Furthermore, it is unclear if the presence of other habitat components within a reach (e.g. LWD and macrophytes) affects fish assemblage response to SWD addition. Clarification is needed to determine whether added SWD will influence fish assemblages.

Here, we used a manipulative field experiment to investigate whether the quantity and size of woody debris affected fish assemblage structure within the shallow, sediment-disturbed channel of the Glenelg River, western Victoria, Australia. We questioned whether the fish assemblage was influenced by the addition of woody debris, without concomitant changes to channel morphology. If so, were combinations of woody debris size and amount more influential to fish, or would adding SWD influence fish assemblages regardless of pre-existing quantities of LWD? By adding SWD to shallow, sediment-disturbed channels containing high or low pre-existing amounts of LWD, it was possible to observe fish responses to changes in woody debris, without confounding with changes in channel morphology (Howson et al. 2009). It was expected that assemblage structure (composition, species richness, abundance) would respond positively to the addition of SWD. In particular, species known to use woody debris such as river blackfish (Gadopsis
marmoratus), mountain galaxias (Galaxias olidus) and flathead gudgeon (Philypnodon grandiceps) were expected to increase in abundance at the experimental locations.

4.4. Materials and methods

4.4.1. Site description

This study was conducted in the Glenelg River, western Victoria, Australia (Figure 4.1). The river is regulated and affected by sedimentation in its middle reaches (Lind et al. 2009), but it has an effective environmental flow (Lind et al. 2007) and supports considerable biodiversity (Robson and Mitchell 2010) especially native fish (Howson et al. 2009, 2010). Twelve, 20 m sections of run-type channel were selected from several reaches of the Glenelg River, downstream of Rocklands Reservoir (Figure 4.1). Each run was separated by deeper pools and represented the range of channel widths impacted by sediment deposition. Discharge patterns varied among years with discharge peaking during August and September (Figure 4.2) although the experiment was conducted during a drought (Howson et al. 2009).

4.4.2. Experimental design

We used a Before-after control-impact design with multiple control and treatment locations and times. Of the twelve runs, six contained high pre-existing levels of LWD and six contained lower levels (LWD: fixed factor, 2 levels: high and low) (Figure 4.1). Addition of SWD was a randomly allocated treatment (with a control comprising no addition (fixed factor, 2 levels: added and not added)) crossed with the level of LWD. Lastly, time was a factor (fixed, 2 levels: before and after SWD added). Interaction terms in this design, particularly SWD × Time or LWD × SWD × Time are indicative of a potential SWD effect. Sites with high amounts of LWD contained a spatial coverage of > 15% or more than 14 pieces of LWD (Figure 4.1) and differed significantly from low LWD sites ($F_{1,10} = 28.691, p < 0.001$). The SWD treatment, consisting of 10 branches, was added after an initial ‘before’ period of fish sampling.
<table>
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<td>High</td>
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<td>High</td>
<td>Control</td>
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</tbody>
</table>

**Figure 4.1** Site locations in the Glenelg River.
Figure 4.2 Average daily discharge at Fulham Bridge gauging site (#238224; January 2002 to January 2005). Solid and broken arrows indicate times of sampling and velocity measurements, respectively. Solid and broken lines indicate times of the addition of SWD and conducting habitat surveys.

Seven sampling trips (trip, random factor, 7 levels nested within time) were undertaken between June 2003 and July 2004 (Figure 4.2). Before-period data consisted of 4 trips (winter-spring) with 3 trips conducted in the after-period (late summer to winter). Flooding prevented the fourth after-period trip and a third round of sampling at one site (Balmoral Low). Because smaller pieces of woody debris break down faster than larger pieces, and are more easily washed downstream during flooding, one year duration was deemed to be a sufficient and realistic timeframe in which to detect effects of SWD placement on fish assemblages. We did not anticipate that the experiment would quantify any effects of SWD on recruitment as it was only there for approximately six months.
Interestingly, most pieces of SWD placed for the experiment persisted in the river until major floods in winter 2010 (T. Howson personal. observation).

SWD was sourced locally and consisted of mixed branches, ranging 0.01–0.1 m in diameter. In each SWD treatment location, 10 branches were randomly allocated with the branch position recorded relative to the nearest bank, nearest existing SWD piece and LWD piece. These characteristics differed little between sites reflecting treatment homogeneity (see Appendix One). No pieces of SWD placed for the experiment moved or were lost during the experiment.

Habitat structure and water quality variables were measured during the study. Water quality variables differed marginally between sites over time (Appendix Two), so are not reported further. Despite random allocation of SWD treatments to locations, there were some differences in channel morphology among treatment combinations. Channels in the low LWD, SWD treatment locations were shallower ($LWD \times SWD; F_{1, 8} = 4.878, p < 0.05$) and wider ($LWD \times SWD; F_{1, 55} = 17.381, p = 0.05$) (chart’s a and c respectively, Appendix Three). Mean depth was similar between wood groups over trips, thus seasonal flow variability had little influence on spatial patterns of depth (chart b, Appendix Three). Mean stream width varied among SWD treatment and control groups in the before period, but not in the after period ($SWD \times Time; F_1, 55 = 8.191, p < 0.05$, chart d, Appendix Three). Water velocity was slow and similar among woody debris groups, with site velocities averaging 0.21 ms$^{-1}$ or less. Sand-size particles were the predominant substratum at most sites. Bank undercutting was present at all sites, except Five Mile low 2. Spatial coverage of emergent macrophytes was significantly greater in the low LWD, SWD control group ($LWD \times SWD; chart e, Appendix Three$), with a significantly greater proportion of submerged macrophytes covering low LWD locations (chart f, Appendix Three). However, subsequent results indicate that these differences had no detectable effect on fish assemblages.
4.4.3. Fish sampling

Fish collection was undertaken using a Smith-root® Model 15D backpack electrofisher (Smith-Root Inc. Vancouver, Washington, USA.) with 100-400 V of pulsed direct current (120 Hz, 1 ms⁻¹ pulse width, 12% duty cycle). Stop-nets (ca. 2 mm mesh) were placed across upstream and downstream ends to prevent fish movement. Three passes were made consecutively to remove fish from each site. Fish were lightly sedated (clove oil, 1 ml per 5 L of water) prior to species identification and length (mm) and weight (g) measurements. Fish recovery was monitored for a period of 20 min then fish were released to point of capture.

4.4.4. Discriminating juvenile and adult length

Fish lengths were classified into adult and juvenile categories to determine the distribution of abundance according to size. Classification of individuals into mature/immature groups could only be undertaken for common species (e.g. *G. olidus*, *G. marmoratus*, *P. grandiceps*, and *Nannoperca variegata*). Discrimination of adults was based on a combination of length and age at maturity information from the literature (Merrick and Schmid 1984, Koehn and O'Connor 1990a), examination of length-frequency distributions and inspection of ripe individuals collected during sampling. Where age at maturity is known to occur at a young age (1+), the modal progression analysis procedure, NORMSEP (FISAT II, FAO 2005) was employed to discriminate 0+ and 1+ age cohorts. Sizes (total lengths) used to discriminate mature individuals were: *G. olidus* 42 mm; *G. marmoratus* juveniles < 123 mm, adults > 150 mm; *P. grandiceps* 50 mm; *N. variegata* 30 mm.

4.4.5. Statistical analyses

Non-metric multi dimensional scaling (nMDS) and permutation analysis of variance (PERMANOVA, Anderson 2005), based on fourth-root (x + 0.0001) transformed CPUE data and Bray-Curtis dissimilarity coefficient, examined sample variation and tested hypotheses of among-group differences in assemblage composition (Clarke and Warwick 1994). One before-period sample was randomly selected and removed (Trip 1) to ensure the PERMANOVA was conducted on balanced data. Factor level
comparisons were conducted upon finding significant model terms (PERMANOVA, Anderson 2005). Similarity Percentages routine (SIMPER, PRIMER® version 5) was employed to identify species primarily accounting for group differences in assemblage composition (Clarke and Gorley 2001).

Analysis of variance (SYSTAT® version 10.0) was used to compare the distribution of species richness, total and species abundances, juvenile and adult abundances across the factors LWD, SWD, Time and Trip. Residual plots revealed that only four species *G. marmoratus, P. grandiceps, G. olidus* and *N. variegata* satisfied the assumptions of homogenous variances and normality. Abundance data was transformed (fourth-root) to meet the assumption of homogeneity of variance (Quinn and Keough 2002). All tests were conducted using Type III Sums of Squares. Pre-planned comparisons were used to examine the relationship between SWD and Time within each LWD group (Quinn and Keough 2002).

4.5. Results

4.5.1. Influence of woody debris size and quantity on fish assemblage composition

In all, 3754 individuals were captured representing eleven species; nine native species and two exotic species: *Gambusia holbrooki* and *Perca fluviatilis*. Four species were common to all woody debris groups, namely: *G. marmoratus, P. grandiceps, G. olidus* and *N. variegata*. Other species including *Nannoperca australis, Hypseleotris* sp., *Nannoperca obscura, P. fluviatilis, G. holbrooki* and *Retropinna semoni* were patchily distributed among sites. A single *Galaxias maculatus* was captured.

Assemblage composition varied among woody debris treatments over time (LWD × SWD × Time; $F_{1, 4} = 6.399$, $p < 0.05$, Figure 4.3a, see Table 4.1) with planned comparisons indicating composition differed between SWD treatment and control levels, for the low LWD group in the after-period only ($p < 0.05$, Table 4.1). Temporal change in composition was observed for both LWD groups (Figure 4.3b and 4.3c), but varied less in high than low LWD for both time periods, indicating a
more distinct or stable assemblage at high LWD locations. The after-period composition difference among low LWD, SWD groups resulted from changes in the abundance of three species: *N. australis, G. marmoratus* and *P. grandiceps*. Abundances of *G. marmoratus* and *P. grandiceps* increased in the SWD treatments (Figure 4.4).

4.5.2. Influence of woody debris size and quantity on fish species richness and abundance

Species richness varied among woody debris groups and times (LWD × SWD × Time; Figure 4.5a, Table 4.1). At high LWD locations, species richness increased with time for both the SWD treatment ($F_{1, 55} = 9.044, p < 0.05, n = 21$) and control groups ($F_{1, 55} = 16.950, p < 0.001, n = 20$), but SWD treatment and control groups did not differ in the after-period ($F_{1, 55} = 1.646, p > 0.1, n = 18$). At low LWD locations, species richness also increased with time but only for the SWD control group ($F_{1, 55} = 48.932, p < 0.001, n = 20$), resulting in greater numbers of species in the after-period SWD control ($F_{1, 55} = 15.626, p < 0.001, n = 17$).

Total abundance varied independently across the terms LWD, SWD, time and among trips (Table 4.1). Highest total abundances were recorded in the high LWD level, SWD treatment and after-period (Figure 4.5b). Variation among trips reflected differences among before–after periods (Figure 4.5c).

Species and size-class abundances varied substantially among woody debris groups (Tables 4.1 and 4.2). *G. marmoratus* increased in abundance during the after-period at low LWD locations with the SWD treatment ($F_{1, 55} = 5.411, p < 0.05, n = 41$, Figure 4.4a). Largest *G. marmoratus* (> 150 mm, TL) were most abundant in the after period and locations with more large wood (Figure 4.4b), but did not respond to SWD despite a significant LWD × SWD interaction ($F_{1, 55} = 4.376, p < 0.05, n = 41$, Figure 4.4b).
Figure 4.3 Non-metric multidimensional scaling plots of assemblage composition: a) LWD × SWD × Time interaction, b) SWD × Time levels within the High LWD location and c) SWD × Time levels within the Low LWD location. Open and closed symbols represent High and Low LWD samples, respectively. Square and triangle symbols represent before-period, SWD treatment and control groups respectively. Circle and diamond symbols represent after-period, SWD treatment and control groups respectively.
Figure 4.4 Mean relative abundance of a) *G. marmoratus*, *P. grandiceps*, *G. olidus*, and *N. variegata* and b) size-specific relative abundance of *G. marmoratus*, *G. olidus* and *P. grandiceps*, as a response to the presence (treatment) or absence (control) of SWD, before and after additions within high and low LWD locations. Note: scale of dependent variable is fourth-root transformed, where error bars represent one standard error.
Figure 4.5 Mean ± 1 standard error of a) number of species captured across factor levels of LWD × SWD × Time; b) abundance distributed across levels of the factors LWD, SWD and Time; c) abundance (dark) and *P. grandiceps* (light) mean abundance distributed across trips. Note: y-axis scale represent fourth root transformation.
Locations with greater quantities of LWD were also associated with increased numbers of intermediate sizes of *G. marmoratus* (123–150 mm, TL). However, the smallest size class (< 123 mm, TL), which contained the largest numbers of individuals, used low LWD runs with the added SWD treatment more frequently (LWD × SWD × Time; $F_{1, 55} = 3.206, p < 0.05, n = 17$, Figure 4.4b).

Abundances of *P. grandiceps* varied among trips and among LWD × SWD groups (Figure 4.5c, Table 4.1). There were higher abundances in the low LWD, SWD treatment group ($F_{1, 55} = 17.476, p < 0.001, n = 41$, Figure 4.4a). Examination of adult (> 50 mm) abundances revealed a further significant SWD × Time term with highest abundances occurring in the after-period, SWD treatment group ($F_{1, 55} = 13.193, p < 0.001, n = 35$, Figure 4.4b) showing that this species responded strongly to added SWD.

Abundances of *G. olidus* varied significantly among LWD, SWD and time groups (Tables 4.1 and 4.2); a pattern largely driven by abundances of adults (> 42 mm TL, Figure 4.4b). Temporal increases in *G. olidus* abundance occurred for both LWD groups (Figure 4.4a and 4.4b), but after-period responses to the SWD treatment depended on the amount of LWD. At high LWD locations, higher abundances were associated with the SWD treatment ($F_{1, 55} = 3.192, p < 0.1, n = 18$), unlike the low LWD locations where significantly lower abundances were associated with the SWD treatment ($F_{1, 55} = 8.035, p < 0.05, n = 17$). This response contrasts with all other species and suggests adding SWD may have positive or negative effects depending on LWD amount.
Table 4.1 Variation in Assemblage composition, species richness, total abundance and abundances of *G. marmoratus*, *P. grandiceps*, *G. olidus* and *N. variegata* across levels of the factors LWD, SWD, Time and Trip(Time). Note: Abundance data used is fourth-root transformed, standardised CPUE data.

<table>
<thead>
<tr>
<th>Source</th>
<th>Composition</th>
<th>Species Richness ($R^2 = 0.491$)</th>
<th>Total Abundance ($R^2 = 0.603$)</th>
<th>G. marmoratus ($R^2 = 0.427$)</th>
<th>P. grandiceps ($R^2 = 0.488$)</th>
<th>G. olidus ($R^2 = 0.414$)</th>
<th>N. variegata ($R^2 = 0.387$)</th>
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* *, **, *** denotes significant differences (p-values based on F distribution) detected at $p = 0.1$, 0.05 and 0.001
†, ††, ††† denotes significant differences (monte carlo generated p-values, Anderson 2005) detected at $p = 0.1$, 0.05 and 0.001
‡, ‡‡, ‡‡‡ denotes significant differences (multivariate t-statistic, Anderson 2005) detected at $p = 0.1$, 0.05 and 0.001
NS denotes no significance
**Table 4.2** Variation in the abundance of adult and juvenile *G. marmoratus*, adult *P. grandiceps* and adult *G. olidus* distributed across levels of the factors LWD, SWD, Time and Trip(Time). Note: Abundance data used is fourth-root transformed, standardised CPUE data.

<table>
<thead>
<tr>
<th>Source</th>
<th>$G. \text{marmoratus} &lt; 123 \text{ mm}$ ($R^2 = 0.434$)</th>
<th>$G. \text{marmoratus} 123-150 \text{ mm}$ ($R^2 = 0.515$)</th>
<th>$G. \text{marmoratus} &gt; 150 \text{ mm}$ ($R^2 = 0.238$)</th>
<th>$P. \text{grandiceps} &gt; 50 \text{ mm}$ ($R^2 = 0.461$)</th>
<th>$G. \text{olidus} &gt; 42 \text{ mm}$ ($R^2 = 0.395$)</th>
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* *, **, *** ($p$-values based on $F$ distribution) denotes significant differences detected at $p = 0.1, 0.05$ and $0.001$. 
Abundances of *N. variegata* varied among the LWD × SWD and SWD × Time groups (Table 4.1). Comparisons partitioning the LWD × SWD term indicated significantly higher abundances ($F_{1, 55} = 5.314, p < 0.05, n = 42$) in the high LWD, SWD treatment (Figure 4.4a).

Analysis of the SWD × Time term revealed no significant after-period difference between SWD treatment and control groups ($F_{1, 55} = 0.941, p > 0.1, n = 35$), but significant before–after temporal effects for both SWD treatment and control (Table 4.1). Therefore, abundance of *N. variegata* in the SWD treatment was different to SWD control in the before-period (Figure 4.4a).

### 4.6. Discussion

#### 4.6.1. Temporal dynamics of the fish assemblage

Before considering the impact of woody debris addition on fish assemblages, the MBACI design allows us to assess temporal changes that occurred, but were not attributable to added SWD. In the after-period, mean species richness increased with similar numbers of species recorded for both LWD groups, except the low LWD, SWD addition treatment. Lower species richness in this latter group resulted from the absence of two species: *R. semoni* and *P. fluviatilis* which occurred rarely throughout the study.

Temporal increases in abundance across woody debris groups were associated with both juveniles and adults, reflecting the influence of both movement and recruitment processes in structuring assemblages. Increased adult abundances, particularly of *P. grandiceps*, most likely resulted from movement but this requires mark-and-recapture information to be confirmed. Young of several species were present in after-period samples, although only significant numbers of the largest species, *G. marmoratus*, were captured. Low capture numbers of juveniles may reflect the sampling procedure used because the juvenile size of most species in the Glenelg River is small (< 30–50 mm in total length except *G. marmoratus*) and electrofishing is generally biased towards larger-bodied fish (Kolz et al. 1998).
Despite our awareness of this bias, and our best attempts to avoid it (e.g. use of maximum voltage possible, higher frequency, fine mesh on dip nets), inferences concerning changes to assemblage structure are restricted to adults; other methods are required to assess the influence of recruitment on fish assemblage structure.

4.6.2. Effect of added SWD on assemblage structure

The spatial structure of fish assemblages in shallow channels depended on the addition of SWD branches where LWD quantity was low. At locations with more LWD, adding SWD did not influence assemblage structure (composition, richness or total abundance), perhaps reflecting an adequate quantity of woody debris available for fish. At low LWD locations, adding SWD produced higher abundances of wood-associated species (e.g. *P. grandiceps* and *G. marmoratus*); but, withholding SWD resulted in lower abundances of wood-associated species and greater abundances of species less dependent on wood (e.g. *N. australis*). In a Venezuelan stream, Wright and Flecker (2004) increased the LWD quantity in pools containing a low amount, finding after 2 weeks that at least 8 fish species had increased in abundance with an assemblage composition that resembled natural pools with higher amounts of LWD. Similarly, Schneider and Winemiller (2008) also noted 3 fish species and many invertebrate taxa responded to added SWD after 2 weeks in a temperate lowland river (Brazos River, Texas, USA), however, only few fish colonised added SWD treatments. Nevertheless, these studies suggest that replacing wood may have an immediate effect, but it appears that longer periods are needed before greater numbers of fish use added wood (e.g. months: Welcomme 2002), which may arise from life-history processes driving abundances over longer, seasonal scales. Our study corroborates these studies, demonstrating that adding even small pieces of woody debris to areas with little LWD can promptly influence multiple fish species, without the necessity for woody debris to significantly change channel morphology.

The effect of adding small woody debris on fish abundance varied among species and size classes. Abundances of *G. marmoratus* were expected to increase with
SWD additions because they are known to be associated with LWD (Jackson 1978a, Koehn et al. 1994, Bond and Lake 2005, Koster and Crook 2007). Initially, adding SWD appeared not to have influenced G. marmoratus abundance (non-significant SWD × Time interaction), but accounting for LWD quantity and fish size, it became evident that juvenile abundances increased at low LWD, SWD treatment locations. Small G. marmoratus have been observed using branches as cover (Koehn et al. 1994) or feeding areas (Davies 1989) and size-related use of different types of cover by G. marmoratus has been noted (Koehn et al. 1994). Larger fish (123-150 and > 150 mm in TL) did not respond to the SWD treatment, probably because they could not be concealed by individual SWD pieces; or, had colonised or defended other locations for residence, as they do not move far along river channels (Kahn et al. 2004, Koster and Crook 2007).

In contrast, abundances of only the larger, adult P. grandiceps (> 50 mm), responded to added SWD irrespective of the amount of LWD present. This was surprising, considering the known association of P. grandiceps with ‘tree trunks’ and ‘large wood pieces’ (Merrick and Schmida 1984). But again, the effect of added SWD was unclear until fish size was accounted for. Abundance of adult P. grandiceps consistently increased in response to added SWD, suggesting specific use of small pieces, perhaps for oviposition, as reproduction coincided with the after-period of the experiment (Howson et al. 2010). SWD could also be used as cover (Culp et al. 1996), or as areas associated with feeding (Benke et al. 1985), but even single fish species can display complex responses to combinations of food and cover (Giannico 2000), suggesting that further experimentation is needed to separate and identify the different contributions that SWD has on fish assemblage structure.

4.6.3. Effects of wood quantity on fish

Assemblage composition at High LWD sites was more diverse, dominated by native species and less-variable among times, reflecting a more stable or perhaps distinct assemblage structure compared to low LWD locations. Furthermore, greater native fish abundances where there is more LWD and when discharge was subsiding may
reflect the value of larger pieces as a resource in shallow water, such as a low-flow refuge (Lisle 1986, Bond and Lake 2005) or for predator avoidance (Everett and Ruiz 1993). Higher abundances, typically of salmonids, are associated with reaches containing channel features corresponding to greater LWD loadings (e.g. Flebbe and Dolloff 1995, Rosenfeld et al. 2000). But, we did not observe substantial differences in channel characteristics attributed to LWD quantity, as the channel gradient was low (Beechie and Sibley 1997) and sedimentation from shifting sand-slugs is continuous (Lind et al. 2009). Alternatively, higher abundances of fish may simply arise from the increased resources provided by greater amounts of the wood itself. In streams, greater wood quantity is associated with higher numbers of insectivorous or xylophagous species (Lehtinen et al. 1997, Wright and Flecker 2004) and faster growth rates of young, insectivore species (Quist and Guy 2001). Woody debris can moderate biotic interactions between fish (Crook and Robertson 1999) and thus higher amounts of large wood could segregate individuals, making existing resources more available (i.e. reduced territorialism). Our results suggest that in shallow lowland channels, more woody debris directly contributes to larger abundances of fish (e.g. by supplying food and shelter, Crook and Robertson 1999).

Most *G. marmoratus*, particularly larger individuals (> 123 mm TL), were found in high LWD locations supporting previous observations of higher abundances with more woody debris in central Victorian catchments (Koehn et al. 1994, Bond and Lake 2005). The use of cover by *G. marmoratus* partly relates to the diurnal behaviour of this species, which seeks cover during the day, while actively moves and forages during night (Koehn et al. 1994, Kahn et al. 2004, Koster and Crook 2007). Koehn et al. (1994) also proposed that use of wood cover could reflect the avoidance of predators. In this study, and as observed in Bond and Lake (2003a, 2005), the effect of sediment largely meant there were few alternative shelter options for *G. marmoratus*. Because the presence of *G. marmoratus* adults and juveniles related to the amount of both small and large woody debris, the addition of woody debris may be useful to support higher densities of *G. marmoratus*, which is a species targeted by anglers.
Runs with more LWD were also associated with the highest abundances of *G. olidus* and *N. variegata*. *N. variegata* has been reported in faster flowing reaches around macrophytes (Saddlier and Hammer 2010), accumulated flood debris, or logs (Allen et al. 2002), perhaps for cover but this little is known of this species ecology. Overhanging woody vegetation (Cadwallader et al. 1980, Closs 1994), organic debris (Koehn and O'Connor 1990a, Bond and Lake 2005) and additions of LWD (Bond and Lake 2005) have also been associated with the distribution of *G. olidus* in central Victorian rivers. Similarly, higher *G. olidus* abundance was associated with more woody debris, particularly the addition of SWD to high LWD locations. This is interesting, as most studies of fish responses to wood additions have focussed on locations where existing pieces are few and jam complexity is low. Considering that different sized pieces of woody debris are spatially aggregated by floods or other natural processes (e.g. beavers) forming complex jams, by only replacing large pieces that alter channel morphology, our understanding of fish assemblage responses to wood addition may have been limited. It is surprising that this study is among few that have manipulated wood size and amount to assess the response of riverine fish. Although increased quantities of woody debris and associated changes to channel morphology may be expected to result in higher fish abundances, further work is needed to establish how additions of wood influence fish assemblage structure independently of changes to channel morphology.

4.7. Conclusion

This study shows that fish assemblages may respond to woody debris irrespective of significant changes to channel morphology, suggesting that wood complexity itself is important under these circumstances. It also shows that addition of SWD may be important to some fish species, and additions of SWD to sites containing low quantities of LWD may lead to increased fish abundance. The relative ease of supply (*i.e.* renewable) and lower cost of handling of SWD, the increasing popularity of ‘soft’ versus ‘hard’ engineering techniques and the possibility of local community and voluntary groups undertaking debris replacement programs using this material means that SWD may provide a useful restoration method where increasing fish
abundances is the aim. In addition, SWD will be contributed to rivers from restored riparian vegetation before LWD becomes available from the same source. Effects of SWD addition should be largest where existing levels of LWD are low, but may also benefit some fish species (especially small species or juvenile fish) where existing LWD quantities are high. Therefore, the addition of SWD should be considered when restoring river reaches for the benefit of fish assemblages, especially where limited SWD enters rivers from riparian vegetation.
Chapter Five: Patch-specific spawning is linked to restoration of a sediment-disturbed lowland river, south-eastern Australia

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5.1. Abstract

Landscape-scale, terrestrial modifications of catchments can increase river sediment loads. In some rivers, the development of ‘sand-slugs’ (i.e. discrete slugs of travelling sand particles) subsequently alters habitat structure with links to declines in regional fish diversity. Increasingly, river channel restoration is being used to conserve biodiversity in sediment-disturbed rivers, but there are few examples to guide restoration efforts. In particular, few studies examine the effect of restoration on ecological processes such as spawning. We report on a trial restoration procedure, consisting of sediment extraction and woody debris replacement undertaken in two 1500 m reaches of the Glenelg River, south-eastern Australia. We aimed to examine the association between reach-scale restoration and fish spawning, predicting that reconstructed channel types (pools and runs) would be used more frequently than corresponding un-modified channel types for spawning. Artificial (polyvinyl chloride (PVC) tubes) and natural (small woody debris) spawning substrata were used to examine the association of fish spawning with reach and channel type. Restoration increased wood volume, but only increased average run depth at one reach. Species including Gadopsis marmoratus, Philypnodon grandiceps, Hypseleotris spp., Nannoperca variegata and Cherax destructor were observed within spawning substrates, but only P. grandiceps frequently spawned on PVC tubes and sparsely on small woody debris substrata. Spawning frequency varied between reach and channel types, with pools in both restored and un-manipulated reaches used more frequently than runs. Restored pools were less frequently used than un-manipulated pools, but restored runs were used up to six times more frequently than un-manipulated runs, indicating that
restoration of the shallowest parts of the channel increased spawning opportunities for *P. grandiceps*. This type of channel restoration may facilitate ecological processes that underpin the persistence of riverine fish populations.

Keywords: Sand-slugs, sediment, restoration, fish spawning, lowland, channel, fish habitat
5.2. Author Contributions

Travis Howson:
- Design the field experiment and conducted fieldwork
- Conducted all statistical analyses
- Wrote the manuscript

Belinda Robson:
- Assisted in interpreting results
- Provided significant comments on draft manuscripts
- Assisted in extensive editing the manuscript for publication

Brad Mitchell:
- Provided comments on draft manuscripts
- Assisted in gaining project animal ethics and sampling permits
5.3. Introduction

Terrestrial modification of catchment landscapes around the world and the subsequent, inflated export of sediment to river channels results in the development of slow-moving, discrete slugs of sand-size particles known as ‘sand-slugs’ (Lind et al. 2009). Sand-slugs are most apparent in low-gradient or lowland reaches where their physical impact is clear—burial of habitat-patches (e.g. pools) and components (e.g. woody debris) maintaining habitat-patches, by up to several meters of sediment. These changes to channel structure can influence habitat function and the roles of habitat-patches for biota (Bond and Lake 2005), especially in streams flowing across Mediterranean or semi-arid landscapes where water is scarce and abstraction or diversion common. Low summer water levels mean that shallow channels with sand-slugs are exposed to further disturbance, including increased risk of desiccation (O’Connor and Lake 1994, Bond and Lake 2005) and higher summer stream temperatures (Alexander and Hansen 1986). Declines in biodiversity, notably the absence of species from fish assemblages, has corresponded with significant deposits of sediment and subsequent changes to habitat structure (Berkman and Rabeni 1987, Waters 1995). Sediment transport and the burial of substrates by sand and silt-size particles, are widely reported to be detrimental to benthic specialists (Cordone and Kelly 1961, Ryan 1991, O’Connor and Lake 1994, Wood and Armitage 1997, O’Connor and Zampatti 2006). This is especially true for families with life-history traits that increase their vulnerability to sediment deposition such as the use of benthic substrates for oviposition or embryonic development (Cordone and Kelly 1961, Berkman and Rabeni 1987, Ryan 1991, Jones et al. 1999, Burkhead and Jelks 2001, Sutherland et al. 2002, Walters et al. 2003, Gillenwater et al. 2006, O’Connor and Zampatti 2006). Fish reproductive processes, in particular, are closely tied to the physical environment (Matthews 1998) and sensitive to sustained changes to habitat structure, such as those arising from press-type disturbances (e.g. altered flow, thermal or sediment regimes) (Regetz 2003, Aarts et al. 2004, Gillenwater et al. 2006). Habitat resources used for key reproductive purposes such as oviposition, including channel patches (e.g. pool or riffle) or components (e.g. large woody debris, gravel, depth, undercut banks)
correspond to the distribution and number of spawning events (Magee et al. 1996, Knapp et al. 1998, Dauble and Geist 2000, Merz 2001). The susceptibility of these resources to burial could mean that spawning is affected in reaches with sand-slugs. For example, Gamradt and Kats (1997) reported that the burial of rock substrates and channel patches (pools and runs) by sediment after recent wildfire disturbance, did not influence abundances of a newt, but the use of channel patches for oviposition decreased by up to 66%. The use of reaches affected by sand-slugs for spawning remains unclear, but the presence of new recruits (age 0+ juveniles) signifies some successful reproduction or at least that the dispersal of juveniles occurs there (Bond and Lake 2005).

Channel restoration offers a practical solution to sand-slugs in rivers by redistributing lost resources to recreate functional habitat-patches for biota (e.g. disturbance refuge, Bond and Lake 2005), but matching effective restoration programs with the impacts of multiple anthropogenic disturbances, over different spatial and temporal scales, is challenging (Lewis et al. 1996, George and Zack 2001). Also, the application of restoration techniques developed in other ecosystems may be inappropriate (‘the cookbook myth’, see Hilderbrand et al. 2005) when restoration is intended for specific purposes, such as to improve local spawning potential. Therefore, restoration methods currently applied to rivers affected by sand-slugs require evaluation, in order to establish whether the use of particular techniques produces the desired outcome.

Restoration techniques that could be beneficial to fishes inhabiting reaches with sand-slugs such as the replacement of large woody debris pieces have effectively recreated channel structure, including scour pools (Larson et al. 2001, Shields et al. 2004, Shields et al. 2006). Woody debris provides potential spawning benefits because it is used as a substrate to assist embryonic incubation (Jackson 1978b, Nash et al. 1999, Storey et al. 2006) and provides cover while nesting, mate finding or waiting for conditions to promote spawning (Bjornn and Reiser 1991, Merz 2001, Wills et al. 2004). The production of scour pools, however, depends on the interaction of woody debris with strong currents so in systems with more variable
hydrology (Puckridge et al. 1998) these natural processes could take some time (see Bond and Lake 2005). Alternatively, other techniques like sediment extraction can ‘instantly’ create deeper water, while excavated ‘sediment traps’ placed above restored reaches can manage the on-going downstream sediment movement (Alexander and Hansen 1983, Avery 1996). The successful use of sediment extraction to derive ecological benefits over larger reach scales (thousands of metres) is uncertain, as the employment of sediment extraction for in-stream mining purposes is considered damaging to aquatic biota (Harvey and Lisle 1998, Meador and Layher 1998).

Although habitat components or resources associated with fish reproduction have been identified, little is known about the association of spawning fish with variation in channel patches (i.e. ‘channel type’) or the use of modified channel for reproduction in rivers affected by sand-slugs. This information is needed, not only to guide current attempts at restoring reaches with sand-slugs, but also to increase understanding of the wider implications of extended sediment deposition on fish populations (Waters 1995). Therefore, the aim of this study was to investigate if spawning use, defined as the number of spawning events detected, varied spatially according to channel and reach (restored or un-manipulated) types. If sedimentation affects the use of channel types (pool or run) for spawning, and restoration is beneficial, then spawning use should be positively associated with restored channel. We also hypothesized that replaced woody debris would be used for oviposition, with spawning use linked to the size, complexity and the placing of pieces among channel types.

5.4. Materials and methods

5.4.1. Study sites

The Glenelg River catchment is located in far Western Victoria, Australia (37° 30’ S, 143° 30’ E, Figure 5.1), covering an area of 12,700 km². Two thirds of the catchment vegetation is cleared, with stock grazing and broad acre cropping dominating the land-use (Department of Water Resources 1989). Land clearing has been implicated
with severe erosion and sedimentation of tributaries and main-stem reaches (Erskine 1994, Lind 2004) with an estimated 10,000–50,000 m³ of sand-size particles stored per kilometer of channel (Lind 2004). Sediment has transformed channel morphology; runs dominate most reaches, varying between 5 and 20 m in width, 10’s and 100’s of meters long and usually between 0.3 and 1.2 m deep. Deep pools (up to 8 m) have been in-filled along with the partial or complete burial other important habitat components (e.g. woody debris, undercut banks, T. Howson personal observation). Remaining pools provide some refuge for flora and fauna, but some are affected by saline groundwater intrusion and contain high water conductivities (10,000 μS cm⁻¹) in bottom waters (Lind 2004, Turner and Erskine 2005). Sediment and salinity issues are compounded further by water extraction and diversion in the upper catchment, though an environmental flow allocated to the river below Rocklands Reservoir during drier months (October to May) is provided (Lind et al. 2006, Coates and Mondon 2009).

Fourteen fish species are common to pools and runs in the mid to upper catchment (Howson et al. 2009) with seven species spawning over spring and summer periods (October to February). River blackfish (Gadopsis marmoratus Richardson, northern form), flathead gudgeon (Philypnodon grandiceps, Krefft) and carp gudgeon (Hypseleotris spp.) may be used to assess reproductive responses to environmental changes because they are benthic, use wood during spawning (Jackson 1978b, Merrick and Schmida 1984), eggs and larvae are distinguishable (Koehn and O’Connor 1990a) and may adopt parental care behaviours during embryo incubation (males of G. marmoratus and P. grandiceps guard eggs).
Figure 5.1 The Glenelg River catchment, Victoria, Australia. Note: HCP1 and HCP2, Harrow Control Pool 1 and Pool 2; HCR1 and HCR2, Harrow Control Run 1 and Run 2; HRP1 and HRP2, Harrow Restored Pool 1 and Pool 2; HRR1 and HRR2, Harrow Restored Run 1 and Run 2; CCP1 and CCP2, Casterton Control Pool 1 and Pool 2; CCR1 and CCR2, Casterton Control Run 1 and Run 2; CRP1 and CRP2, Casterton Restored Pool 1 and Pool 2.
5.4.2. Rehabilitation procedure

Two reach-scale rehabilitation procedures were completed in different areas of the Glenelg River, one near the town of Harrow (ca. 50 km downstream of Rocklands Reservoir) during February 2003 (described in Howson et al., 2009), the other near the town of Casterton (ca. 60 km downstream of Harrow) in 2004 (Figure 5.1). Each rehabilitated reach consisted of an ca. 1500 m length of river, re-modelled using sediment extraction to lower bed height in runs, create and enlarge pools and construct sediment trap hole above the rehabilitated reach to stop sediment further migrating downstream into newly modified reaches (Figure 5.2). Large pieces of woody debris (LWD > 0.1 m in diameter) were also inserted (secured with hardwood piles) into each restored reach, but in different configurations. At Harrow, rack-member (‘engineered’) wood jams and single red gum (Eucalyptus camaldulensis) logs were placed in strategic positions in pools and runs. At the Casterton restored reach, large red gum tree stumps with attached roots were placed with single red gum logs along the edge of pools and throughout runs (Figure 5.2). Estimated loadings of LWD added were 0.011 m$^3$ m$^{-2}$ and 0.013 m$^3$ m$^{-2}$ of channel for Harrow and Casterton, respectively. These values are similar to loadings in other Victorian rivers (Lester et al. 2006, Howson et al. 2009).

5.4.3. Study design

Restoration altered the existing channel by creating defined pools and runs: two pools and two runs were randomly selected in each restored reach. Two additional, un-manipulated reaches were selected randomly as ‘controls’ to compare spawning use of channel types in restored reaches with use in un-manipulated reaches still containing significant sediment deposits. Again, two pools and two runs were selected. To characterise the distribution of spawning within channel types, two depth strata were selected. Shallow zones were defined as water less than 1.2 m deep and deep zones were defined as water greater than
Figure 5.2 Harrow and Casterton restored reaches. Note: Photographs A-G, correspond to labelled points on the diagram: A, dry, sediment-filled river channel prior to restoration; B, sediment extraction and shaping of channel; C, placed single LWD logs; D, placed racked-member, engineered LWD jam; E, placed trunk with attached root-ball; F, sediment trap; G, modified vehicle crossing. Photograph’s H and I are examples of SWD bundle and branch respectively. Photograph J: *Philypnodon grandiceps* oocytes attached to a SWD branch piece.
2.5 m deep. Shallow zones were common to both pools and runs, thus enabling a valid comparison of the spawning use of shallow water among different patches. The area of shallow zone sampled was also standardised by keeping the area of runs (run length multiplied by average width) similar to the pool shallow zone areas (pool perimeter multiplied by the average shallow zone width). Deep zones were studied to determine if spawning frequency differed among deep and shallow zones in pools.

5.4.4. Detecting fish spawning - use of spawning substrates

We used spawning substrates, consisting of PVC tube and small woody debris (SWD) to estimate spawning frequency. Such substrata have been successfully use to detect fish spawning in rivers (Knaepkens et al. 2004, Firehammer and Scarnecchia 2006, O’Connor and Zampatti 2006) and lakes (Hunt and Annett 2002, Mangan et al. 2005) and are useful when other methods (e.g. adult or nest counts, spawning substrate searches) are limited (i.e. spawning in deep or turbid water; inside of logs, Jackson 1978b). They represent features of natural substrata (e.g. logs, rocks) that can be readily replicated across locations. Importantly, restored and unrestored sites may have differed in the frequency of suitable spawning substrata, but we could not accurately measure this. By installing equal numbers of spawning substrata at each site, frequency of use could be accurately estimated. This method means that we have assessed the larger scale effects of restoration on spawning frequency. Actual spawning frequency at these sites, in the absence of our artificial substrata, may be limited by the occurrence of suitable substrata.

Spawning substrata consisted of two sizes of PVC tube (Large: 400 x 50 mm diameter & Small: 200 x 25 mm diameter) and two configurations of small woody debris: branches and bundles. Large PVC tubes were kept to the dimensions specified by Leevers et al. (2003) and O’Connor and Zampatti (2006) for Gadopsis spp., however, small PVC tubes were also tested to determine if small species (e.g. Nannoperca variegata) would use artificial substrata. Additionally, fly-screen (aluminium mesh) lined the inside of the tubes, providing a textured surface and
enabling eggs to be removed for identification. Branches were single pieces of SWD, while bundles were constructed of multiple pieces of SWD approximately 400 mm length with wood volume in each bundle kept similar to that of branches (ANOVA, $F_{1,58} = 2.496, P = 0.120, n = 30$).

In each run, 12 PVC tubes (6 large and 6 small) and 10 SWD pieces (5 single pieces, 5 bundles) were added to randomly chosen positions (Appendix Four). Similarly, within each pool, the two depth zones (shallow < 1.2 m and deep > 2.5 m) received 12 PVC tubes (as above), but SWD (10 pieces as above) was only added to the shallow zone. PVC tubes were set 10 cm off the bottom to avoid collecting sediment, while SWD pieces tended to rest above the substrate enabling gaps to occur under pieces. Once spawning was detected, the aluminium mesh was carefully removed and eggs were identified to species level, based on morphometric features identified in Koehn and O’Connor (1990a). Aluminium mesh with attached embryos was transferred to individual ‘incubators’ consisting of a 50 mm PVC tube with the ends enclosed (250 $\mu$m mesh). From larvae that hatched, a small sample of five individuals was collected, preserved in 70% ethanol and identified using a dissecting microscope and the nomenclature of Serafini and Humphries (2004) and Neira et al. (1998). Hatched larvae complemented the identification of egg species.

Sampling of spawning substrata was determined primarily by temperature as it is a primary cue for fishes spawning in southern Australia (Humphries et al. 1999). Initiation of spawning in river blackfish has been previously associated with temperatures of 16 °C or greater (Koehn and O’Connor 1990a), while spawning in flathead gudgeon and carp gudgeon has been associated with temperatures above 21 °C in Victoria (Koehn and O’Connor 1990a) In the Glenelg River, long term (28 years, gauge #238224) surface temperature data indicates 16 °C is reached during late October to early November, with higher temperatures rising above 21 °C in December. Checks of spawning tubes (5 times) and SWD substrates (twice) for oocytes were conducted over the summer period from November 2004 to February 2005. Average distance between spawning tubes and cover in the shallow zone
comprised a few meters (Appendix Five). Beneath most tubes, sand was the predominant substrate, although clay and silt substrate were also common (Appendix Five).

5.4.5. Habitat: structure and monitoring

Habitat components consisting of temperature, dissolved oxygen, pH, salinity, LWD, aquatic macrophytes, depth and substrate type were quantified within each pool and run. Temperature loggers (IBTag G, Thermodata Pty. Ltd., Melbourne) were placed at each pool and run location, suspended under a float in the shallow and deep water zones at depths of 0.6 and 3 m, respectively. Water height was recorded to the nearest mm using a constructed gauge (1 m fibreglass ruler attached to wooden stake), placed at one point in each site. Water quality variables, consisting of dissolved oxygen, pH and salinity were measured (YEO-KAL model 611, YEO-KAL Electronics, Sydney) at 0.5 m depth within each location and additional pool samples were taken at 0.5 m increments from the surface to maximum depth. The number, size (basal diameter, length) and type (existing or replaced, single or log jam) of LWD pieces were recorded within the shallow zone of pools and runs. We did not survey woody debris in the deep areas of pools (> 1.2 m). LWD volume was calculated, using basal diameter and piece length. Basal diameter was recorded 0.5 m from the largest end of the piece, while length was measured to the nearest 0.5 m (0.1 m for pieces < 5 m).

The proportion of the shallow zone covered by macrophytes was estimated within a 1 m² quadrat, replicated at 40 random points in each shallow zone of pool or run sites. The depth of runs was recorded at 50 random points. At each PVC spawning substrate, we also determined the distance to the nearest LWD piece (Existing-Single, Existing-Jam, Placed-Single and Placed-Jam), nearest macrophyte bed, nearest bank, depth of substrate and bed type. The environmental features surrounding PVC tubes were described to determine whether the use of spawning substrates over larger spatial scales related to the small-scale distribution of various habitat components.
5.4.6. Statistical analysis

Analysis of variance (SYSTAT Software Inc.) was used to compare the distribution of number of LWD pieces, volume of LWD, proportion of macrophyte cover and depth across channel types, treatment reaches and areas. Three-way, crossed models were used incorporating the factors: area (2 levels: Harrow and Casterton, fixed), treatment reach (2 levels: restored and control, fixed) and channel type (2 levels: pool and run, fixed). Prior to analysis, the number of LWD pieces and volume of LWD was standardised by the area of shallow zone surveyed. Residual plots were used to assess normality and homogeneity of variances: volume of LWD was transformed to a natural logarithm scale ($\log_e(x)$). After checking for co-linearity among variables, Principal Components Analysis (PCA, PRIMER® Version 5) was conducted on a reduced set of variables (average temperature, temperature variance, mean dissolved oxygen, mean pH, mean conductivity, mean water height, variance in water height, mean macrophyte cover, number of LWD pieces per m$^2$) to identify environmental variables contributing to variation between the positions of tube, channel and reaches within the catchment. All variables concerning the position of tubes (except water quality, macrophyte cover and LWD number) were transformed using a natural logarithm, $\log_e(x + 0.001)$ to improve normality and standardised (normalised, Clarke and Warwick 1994) prior to matrix construction (Euclidean-distance). Variables loading (correlating) greater than 0.4 on principal components were considered for interpretation of principal component axes.

Spawning frequency data (counts of substrates with eggs) was arranged into a four-way contingency table, with the factors area (2 levels; Harrow and Casterton), treatment reach (2 levels; restored and control), channel type (2 levels; pool and run) and trip (5 levels; trip 1 to trip 5). Replicate channel types were pooled (separately for pools and runs) within each treatment type and reach, to increase the number of observations per cell. Ratio of the variance to mean for the observed frequencies was less than two, suggesting the observed count data was not seriously over-dispersed and assumed to be Poisson distributed (Richards 2008). Log-linear modelling was used to identify the most parsimonious combination of
factors that best estimated the observed frequencies (saturated model) and to test null hypotheses of no association between channel type (pool and run), reach location (Area; restored and control) and trip.

5.5. Results

5.5.1. Temperature and hydrological conditions

Spawning occurred under warm temperatures and low flow conditions (Figure 5.3). Water temperature fluctuated mostly between 17 and 25 °C during sampling, with a steady rise in shallow zone temperature during November, as conditions shifted from cool (averaging 14 °C when spawning substrates placed) to warm (average 22 °C by 1st of December). Mean temperature in pool deep zones, ranged between 2 and 3 °C lower on average than the shallow zone (Table 5.1), but sufficient for river blackfish spawning (16 °C)(Table 5.1, Figure 5.3). In addition, the shallow zone mean temperature was warm enough for flathead gudgeon (21 °C) and carp gudgeon (21 °C) (Figure 5.3). River discharge was low (average 36.7 ML day⁻¹) and falling throughout most of the study period, except for two spates that resulted in short-term mixing of the water column (Figure 5.3). Annual summer water column stratification persisted with reduced flow, shown by low dissolved oxygen concentrations in the deep zone (Table 5.1).

5.5.2. Patch structure: environmental variation among reach, channel and tube locations

Restoration works significantly changed existing channel structure. Runs in the Harrow restored reach were more than 50% deeper than all other runs (area × treatment, $F_{1, 4} = 12.693, P < 0.05$, Figure 5.4a). The volume of LWD added to restored reaches increased the existing LWD volume (per m² of shallow zone) between 167% and 1123% for runs and 77% and 779% for pools. However, the average number of pieces and LWD volume (per m² of shallow zone) was not significantly different among channel types, reaches or areas (Figure 5.4b and 5.4c). The average (± sd) length of pools modified or constructed in restored reaches (94 ±
72 m) was not as long as control pools (183 ± 89 m), but the average (± sd) width of the shallow zone was wider in restored (4.2 ± 2.7 m) compared to control pools (1.6 ± 0.8 m). Macrophyte cover in restored reaches did not differ to control reaches, however, macrophyte cover was significantly greater ($F_{1, 8} = 19.984, P < 0.01$) at Harrow than Casterton (Figure 5.4d).

**Figure 5.3** Temperature and discharge profile over the duration of sampling, November 2004 to February 2005. Average site temperature, per 20 minute increments, measured 0.6 m (shallow zone: grey line) and 3 m (deep zone: black unbroken line) from the surface. Average daily discharge (broken black line) recorded at Dergholm gauging station (#238211). Note: vertical broken line represents the time of adding spawning substrates. Arrows indicate times of sampling, solid-line arrows indicate times of sampling PVC substrates, broken-line arrows represent times of sampling SWD substrates.
### Table 5.1

Mean (± standard deviation) values of temperature, dissolved oxygen, pH and salinity of shallow and deep zones across restored and un-manipulated (control) pools. Note: * temperature samples recorded using a water quality meter for trips 3, 5 & 6 (** data for trips 3 and 5 only).

<table>
<thead>
<tr>
<th>Location</th>
<th>Temperature (°C)</th>
<th>Dissolved oxygen (mg L⁻¹)</th>
<th>pH</th>
<th>Salinity (g L⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harrow C P1 (shallow zone)</td>
<td>20.7 ± 2.0</td>
<td>8.2 ± 4.6</td>
<td>7.2 ± 0.1</td>
<td>1.5 ± 0.2</td>
</tr>
<tr>
<td>Harrow C P1 (deep zone)</td>
<td>18.7 ± 1.9</td>
<td>1.7 ± 1.4</td>
<td>7.1 ± 0.1</td>
<td>1.6 ± 0.3</td>
</tr>
<tr>
<td>Harrow C P2 (shallow zone)</td>
<td>21.9 ± 2.2</td>
<td>9.5 ± 4.0</td>
<td>7.3 ± 0.1</td>
<td>1.4 ± 0.1</td>
</tr>
<tr>
<td>Harrow C P2 (deep zone)</td>
<td>17.6 ± 1.3</td>
<td>0.0 ± 0.1</td>
<td>7.1 ± 0.1</td>
<td>1.5 ± 0.2</td>
</tr>
<tr>
<td>Harrow R P1 (shallow zone)</td>
<td>19.4 ± 2.5</td>
<td>8.1 ± 5.1</td>
<td>7.3 ± 0.1</td>
<td>1.7 ± 0.1</td>
</tr>
<tr>
<td>Harrow R P1 (deep zone)</td>
<td>16.5 ± 2.0</td>
<td>0.5 ± 0.9</td>
<td>7.0 ± 0.2</td>
<td>3.8 ± 2.2</td>
</tr>
<tr>
<td>Harrow R P2 (shallow zone)</td>
<td>19.6 ± 2.5</td>
<td>6.9 ± 2.6</td>
<td>7.2 ± 0.1</td>
<td>1.7 ± 0.1</td>
</tr>
<tr>
<td>Harrow R P2 (deep zone)</td>
<td>17.5 ± 1.1*</td>
<td>0.2 ± 0.3</td>
<td>7.0 ± 0.1</td>
<td>2.4 ± 0.6</td>
</tr>
<tr>
<td>Casterton C P1 (shallow zone)</td>
<td>20.2 ± 2.6</td>
<td>9.6 ± 3.4</td>
<td>7.6 ± 0.1</td>
<td>1.6 ± 0.4</td>
</tr>
<tr>
<td>Casterton C P1 (deep zone)</td>
<td>16.9 ± 2.1</td>
<td>4.1 ± 2.4</td>
<td>7.4 ± 0.1</td>
<td>1.6 ± 0.4</td>
</tr>
<tr>
<td>Casterton C P2 (shallow zone)</td>
<td>20.8 ± 2.3</td>
<td>9.0 ± 3.9</td>
<td>7.8 ± 0.3</td>
<td>1.6 ± 0.5</td>
</tr>
<tr>
<td>Casterton R P1 (shallow zone)</td>
<td>20.5 ± 3.0</td>
<td>9.2 ± 4.6</td>
<td>7.9 ± 0.1</td>
<td>1.8 ± 0.2</td>
</tr>
<tr>
<td>Casterton R P1 (deep zone)</td>
<td>22.9 ± 0.6**</td>
<td>9.0 ± 6.4**</td>
<td>7.8 ± 0.0**</td>
<td>1.8 ± 0.3**</td>
</tr>
<tr>
<td>Casterton R P2 (shallow zone)</td>
<td>21.1 ± 2.5</td>
<td>7.3 ± 3.6</td>
<td>7.8 ± 0.2</td>
<td>1.7 ± 0.2</td>
</tr>
</tbody>
</table>

Descriptive statistics of environmental components indicated differences between reaches were detectable, but were generally small (Table 5.1 and Appendix Five). Casterton reaches contained a marginally lower water height, higher mean pH, salinity, greater temperature range and the lower cover of macrophytes compared to Harrow (Table 5.1, Appendix Five, Figure 5.4). Greater differences in physicochemical variables were observed within pools, between the deep and shallow zones (Table 5.1). Saline stratification of the water column was only observed at the Harrow restored reach. Deep zone dissolved oxygen concentrations were much lower than the shallow zone, often less than 3mg L⁻¹, particularly for Harrow reaches (Table 5.1). No relationships were apparent between tube position and depth, distance to bank and LWD across channel types and reaches (Appendix Five).
Figure 5.4 Spatial variation of four habitat components, across pools and runs in control (white) and restored (black) reaches, within Harrow and Casterton Areas, after the application of a restoration procedure: (a) depth of runs ($n = 8$), (b) site LWD volume, standardised according to site shallow zone area, (c) number of LWD pieces per site, standardised according to area of site shallow zone, (d) proportion of macrophyte cover per site.

Multivariate analysis of the shallow zone environmental variables showed samples clustered according to treatment and area, with little difference between runs and pools within each treatment reach (Figure 5.5). Five principal components with eigenvalues over 1 were extracted, explaining 74.1% of the data variability, the first two principal components accounting for 45.3% of the variability (Appendix Six). The variables mean macrophyte cover ($r = 0.42$), mean water height ($r = 0.44$), pH ($r = -0.45$), salinity ($r = -0.35$) and temperature variance ($r = -0.40$) all loaded moderately on the first principal component axis with mean temperature ($r = 0.60$)
and water height variance \((r = -0.57)\) loading higher on the second principal component axis (Figure 5.5). The first principal component axis resembled an underlying hydrological gradient among locations as mean water height was inversely related to mean pH, mean salinity and temperature variation, and each correlated with the upstream–downstream position of reaches. The second principal component axis showed water height variance was inversely related to mean temperature and the Harrow restored reach was characterised by a greater fluctuation in water height and lower mean temperature. Therefore, larger scale hydrological influences corresponded to environmental differences between reaches, rather than smaller-scale, within-site differences in tube position.

![Figure 5.5 Principal Component Analysis plot of measured habitat variables surrounding the shallow zone PVC spawning substrates. Grey symbols represent Casterton reaches, black symbols represent Harrow reaches, triangles = control pools, inverted triangles = control runs, squares = restored pools, diamonds = restored runs. Note: each point represents a spawning tube.](image)
5.5.3. Spawning use of substrata across different patches

Only two species were recorded spawning on PVC tube substrates, *P. grandiceps* and *Hypseleotris* spp. (one occasion). Although adult *G. marmoratus* were observed residing in tubes, no eggs were collected. Flathead gudgeon spawning always occurred inside the tube and particularly on the sides. Smaller tubes (69) were favoured ($\chi^2 = 12.706, P < 0.01$) over larger tubes (33). Log-linear modelling of the spawning frequency data from PVC tube substrates in the shallow zone only, showed that a model containing all main effects and one two-way interaction (treatment × channel type) best fit the data ($\chi^2 = 36.2, P = 0.23, \text{BIC} = -107.2$).

Harrow reaches contained a higher number of spawning events (65) than Casterton reaches (37) but this reflected spawning responses at the restored reaches (Harrow: 39, Casterton: 15), as the number of spawning events at control reaches (Harrow: 26, Casterton: 22) was similar. The treatment effect varied with channel type, because restored runs had many more spawning events (24) than control runs (4), but restored pools had fewer spawning events (30) than control pools (44) (Figure 5.6). The number of spawning events in control pools (44) was 11 times higher than observed in control runs (4) (Figure 5.6). This was not the case for restored reaches where the number of spawning events was found to be more similar between pools (30) and runs (24). No spawning tubes in pool deep zones contained any evidence of spawning nor were they colonised by other biota.

Added SWD was used as a spawning substrate only by *P. grandiceps* and few events were recorded (5): three in control pools and two in rehabilitated pools. Four egg masses were recorded on the more complex SWD bundle; only on one occasion did *P. grandiceps* use a SWD branch as a spawning substrate. Lower numbers of spawning events on SWD pieces may reflect the lower frequency and number of SWD pieces sampled. Added SWD substrates were colonised by a range of algal, macroinvertebrate and fish taxa, including: Amphipoda, Odonata (Epiproctophora), Trichoptera, Decapoda (*Paratya australiensis* and *Cherax albidus, Cherax destructor*) and Gastropoda. Fish captured residing within SWD bundles and included the variegated pygmy perch (*N. variegata*) and adult flathead gudgeon (*P. grandiceps*).
Figure 5.6 Distribution of recorded spawning events among treatment reaches (restored and control) and channel types (pools and runs). Black fill represents Harrow reaches, white fill represents Casterton reaches.

5.6. Discussion

5.6.1. Use of artificial substrata to detect spawning

The limited use of artificial substrata by species other than *P. grandiceps*, despite their presence at these locations (Howson et al. 2009), highlights the inherent uncertainties in assessing the spawning use of restored channels by fish assemblages. The selected sampling period of late spring–summer, or choice of particular types of artificial substrata, may have prevented detection of spawning in some species (e.g. *Galaxias olidus, Nannoperca australis*). However, the lack of response from species known to be present, spawning and to use artificial substrata was surprising. Frequent checking of artificial substrata may have disturbed *G. marmoratus* occupying tubes, preventing spawning, but it was not possible to employ continuous, non-contact monitoring on a large number of substrates spread across multiple locations. Spawning response by *P. grandiceps* only, may also relate
to the dominance of this species in the Glenelg River fish assemblage (Howson et al. 2009). A high abundance may be advantageous, because *P. grandiceps* may be the first to reach, occupy and perhaps defend substrates from other fish. Dominance of *P. grandiceps* in the Harrow restored reach fish assemblage may also reflect possible tolerance to conditions following the disturbance of sediment extraction, which may confer a recruitment advantage, if resources for larval and juvenile development are not limited by sediment extraction. Ultimately, tracking fish movement or individual identification (e.g. tagging) during spawning periods would assist in understanding the distribution of spawning events and response to restoration, particularly over small spatial scales (Pope and Willis 1997).

It is also possible that sufficient suitable natural substrates were present, reducing the use of artificial substrata for spawning (Hunt and Annett 2002, Wills et al. 2004). Selection of particular substrates was apparent: the opening diameter and length of the spawning tube were related to spawning frequency, with twice as many spawning events found on smaller tubes. Interestingly, no eggs were deposited on the outside of tubes at any stage, indicating that internal tube characteristics were more important. We suspect that given the opportunity, adult *P. grandiceps* selected smaller holes for spawning perhaps to avoid larger competitors or predators, who may have found PVC tubes to provide excellent concealment. The utility of artificial substrata to detect spawning has great potential for monitoring, although refinement of the technique, including use of various types of substrates by different species is needed.

5.6.2. Environmental variation, low river discharge and spawning

Minimum temperatures previously associated with the onset of spawning in *G. marmoratus*, *P. grandiceps*, and *Hypseleotris* spp. corresponded with the beginning of sampling in this study (Jackson 1978b, Koehn and O'Connor 1990a) and the timing of spawning events was similar between restored and un-manipulated reaches, indicating that the initiation of spawning occurred over a large-scale, perhaps by an environmental cue (e.g. temperature, discharge) or biological activity
Variables sensitive to discharge variability (temperature, dissolved oxygen, salinity, pH and water height) that could potentially affect the quality of nesting locations in shallow areas clearly differed among reaches, but did not correspond to reach or patch spawning use. This was probably because *P. grandiceps* was tolerant of this range of physicochemical conditions (Koehn and O'Connor 1990a, Larson and Hoese 1996, Allen et al. 2002, Humphries et al. 2002).

### 5.6.3. Channel restoration and spawning event distribution

Substrates in un-manipulated pools were most frequently used for spawning, refuting the hypothesis of a higher use of restored reaches at Harrow and Casterton. Despite sand-slug impacts, un-manipulated pools still contained features attractive to fish, such as woody debris and macrophytes. Restoration of runs and pools may increase connectivity to groundwater (Kasahara and Hill 2008) which may increase the input of saline or low-oxygen waters in some reaches of the Glenelg River. Shallow regions of pools (< 1.2 m) were warm, high in oxygen and contained an abundance of ‘hard substrates,’ which were readily used by spawning *P. grandiceps*. In contrast, the absence of spawning in the deep zone (> 2.5 m) corresponded with persistent, low concentrations of dissolved oxygen (< 3 mg L⁻¹). Dissolved oxygen is critical to embryonic development (Balon 1975, Keckeis et al. 1996, Greig et al. 2007) and hypoxic concentrations (< 1 mg L⁻¹) can increase embryo mortality (Bjornn and Reiser 1991, Keckeis et al. 1996). Adults may avoid low concentrations, as additional biological demands can affect the amount of parental care given to eggs (Jones and Reynolds 1999).

Spawning in existing pools may reflect favourable conditions associated with the value of deep water in reaches with sand slugs (Bond and Lake 2005). However, in the presence of annual summer stratification (Lind 2004) use of deep waters is probably limited. Therefore, re-creating deep pools (> 2.5 m) may not provide any extra biological benefit for spawning and efforts targeting more frequent, rather than deeper pools may be more useful.
Higher spawning use of restored runs supported the hypothesis that spawning use is positively associated with the occurrence of large increases in large wood volume and depth. Furthermore, the consistency of higher spawning use in restored reaches from different areas was striking. Runs, like shallow zones of pools, contained woody debris, macrophytes and higher temperatures and dissolved oxygen concentrations, all components contributing to suitable conditions for spawning. It appears the combination of increased depth, along with suitable substrates and or cover, facilitated the spawning of *P. grandiceps* in both runs and pools.

Increases in large woody debris volume were expected to provide additional ‘hard substrates’ for oviposition (Merrick and Schmida 1984) or at the very least, a source of cover (Bjornn and Reiser 1991, Merz 2001, Zimmer and Power 2006). However, there is still uncertainty about how *P. grandiceps* perceived individual restoration treatments. It is probable that increased use of spawning substrates further from placed LWD in Harrow restored runs, either reflected a negative response (avoidance) to the large woody debris treatment, or positive response to the use of substrates placed closer to other cover (e.g. submerged macrophyte beds). Further manipulations confirmed our expectations that even small pieces of woody debris, in different configurations, can be used for spawning when added to provide additional substrate. This also suggests that the functional role of smaller sized pieces of woody debris for fish warrants further study.

5.6.4. Spawning, detection and implications for restoration ecology

While it is often asserted that restoration practices will benefit fishes that move into new patches from surrounding areas (Schlosser 1995), the recovery pathways are seldom considered (Lake et al. 2007). Movement is assumed to drive colonisation of restored reaches, particularly after a significant disturbance (Palmer et al. 1997, Bond and Lake 2005) and other studies show this clearly (House and Boehne 1985, Gowan and Fausch 1996). Our results suggest that local spawning may also contribute to the development of fish assemblages in restored reaches. Local
spawning and recruitment may also offer a partial explanation of the observed dominance of some species and the slower response of other species within some restored reaches (Howson et al. 2009), that otherwise could be expected to respond quickly via active movement or through passive larval drift.

5.7. Conclusions

Restoration of river channel involving the removal of sand-slugs and reinstatement of woody debris was associated with a higher frequency of fish spawning. Considering the limitations of sampling over a short temporal period, prolonged drought and spawning response by only one species on artificial substrata, these results may not represent the overall effectiveness of the restoration procedure to assist spawning fish. Nevertheless, these results do show that restoration of channels with sand-slugs may be positively associated with the distribution of spawning events. Artificial substrata proved useful to examine spawning potential both within and across different channel patches. Clearly, river channel restoration can influence ecological processes, such as spawning, that are vital to sustain animal populations.
6. Chapter Six: General Discussion

6.1. River rehabilitation of a sediment-disturbed lowland river: habitat changes and response of fish assemblages

There has been a scarcity of published empirical evidence that shows a positive response by fish assemblages to river rehabilitation in Australia, and little knowledge exists on successful rehabilitation techniques for sediment-disturbed rivers in general. Therefore, this thesis focussed on fish assemblage responses to experimental river restoration techniques applied to an Australian sediment disturbed lowland river. The prevailing guiding principle in river rehabilitation management is the ‘field of dreams hypothesis’ (sensu Palmer et al. 1997), which recognises the importance providing more or different habitat resources or conditions in order to facilitate greater biodiversity. Sediment impacted lowland rivers offer a unique opportunity to examine the ‘field of dreams hypothesis,’ because it is not unreasonable to assume that even simple alterations of increasing depth and woody debris replacement could lead to large changes in ‘habitat diversity’ and therefore biodiversity, considering woody debris itself attracts a large number of species in lowland rivers (Benke et al. 1985, O’Connor 1991, Lehtinen et al. 1997).

The first study of this thesis examined the role of sediment extraction and LWD replacement as mechanisms for creating greater fish assemblage diversity. It was expected that constructed pools and runs lined with woody debris pieces could provide suitable places for fish, particularly if rehabilitated reaches assist fish for the purposes of refuge or reproduction (Chapter One). However, the modification of a 1500 m degraded reach of the Glenelg River using sediment extraction and large woody debris replacement did not positively alter fish assemblage structure compared to un-manipulated reaches between the before and any after period sampled (Chapter Three, Table 6.1). After three years, no physical structural damage of the rehabilitated reach was observed (e.g. burial of replaced LWD pieces), unlike previous studies that highlight the problems of channel restoration in rivers with high sediment loads (Frissell and Nawa 1992, Lintermans 2002). The
absence of a positive response by the fish assemblage after two years of post-works monitoring was unexpected and contrary to significant, rapid changes in assemblages observed elsewhere (Shields et al. 1998, Zika and Peter 2002).

Large temporal changes in assemblage composition corresponded with low flows and reduced water quality arising from a longer term drought disturbance. Spatial variation in fish assemblage composition only weakly correlated with water quality, suggesting it was only partly responsible for differences between locations during the period of monitoring. High electrical conductivity readings reflecting elevated salinity within the rehabilitated reach and were noted to be at levels that are detrimental to some species (e.g. *Gadopsis marmoratus*) and particularly for more sensitive early life history stages (Clunie et al. 2002).

Over the summer of 2002-2003, many runs dried out in the mid-upper catchment leaving increasingly saline pools as the only refuge. The low river flow during this time period enabled earthworks machinery to readily access these runs and pools. The subsequent impact of channel modification, particularly within or adjacent to pools that served as refuges is unknown, and was beyond the scope of this thesis to determine. Although the types of techniques used in the construction phase (pile driving) are known elsewhere to locally disturb fish and potentially increase mortality (Popper 2005). The disturbances of drying combined with the disturbance of channel modification may have contributed to a delayed response to restoration, beyond the length of the present study.

Bond and Lake (2005) conducted another similar restoration experiment in a sediment-disturbed stream in a central Victorian catchment. However, their experiment differed from the present study because the restoration works did not include sediment extraction and fewer large wood pieces were added (1 or 4 per 100 m as opposed to an average of 8 pieces per 100 m in the present study). Bond and Lake (2005) observed that small scour pools created from placed wood structures were used by numbers of *G. marmoratus*, *G. olidus* and *N. australis*, although abundances declined as the pools dried up during a supra-seasonal
drought. In response to their results, Bond and Lake (2005) stated that they were not aware of any other studies that had specified drought as an explanation for restoration failure, concluding, that refuges should be incorporated into planning to ensure the survival of populations in response to on-going drought.

A significant period of reduced discharge attributed to drought, also appeared to influence the response of fish assemblages in the present study, however, expanding critical pool refuge by using sediment extraction was expected to have counteracted this disturbance. Fishes still declined in rehabilitated and control pool refuges despite large pools of water, although salinity increased significantly with extreme low flow conditions. This highlights that refuge quality is perhaps as important as the presence of the refuge itself, which is often not considered when establishing wildlife refuges as a strategy to combat disturbance. It also suggests that a strategy to create higher quality pools for native fish of the Glenelg River, should focus on excavating more shallower pools, which have reduced saline water intrusion or increased mixing of the water column, rather than construction of fewer, deeper pools.

Delays in the potential assemblage responses to restoration arising from the influence of larger, landscape-scale disturbance overwhelming smaller, site-scale patches, is not just an attribute of sediment-disturbed or drought-affected rivers, but is being increasingly reported in other recent studies of habitat restoration overseas (Larson et al. 2001, Pretty et al. 2003, Harrison et al. 2004, Palmer 2009, Sundermann et al. 2011) as well as in Australia (Becker and Robson, 2009). Its not surprising given that restoration is being undertaken in the most disturbed systems, and after improvements to channel structure are made, other factors such as degraded water quality may still persist (Larson et al. 2001, Harrison et al. 2004). Undertaking restoration in the most disturbed areas may not be ideal, as there are likely to be a number of different barriers that present challenges to recovery (Robson et al. 2011). Alternatively, focussing on long-term restoration at larger catchment scales, in order to assist structural and functional improvements (e.g. revegetation to assist water quality, woody debris dynamics over larger areas) is
perhaps another solution, rather than undertaking micro-management of degradation and disturbance on fish assemblages (Fausch et al. 2002).

Limited differences between fish assemblages in the control, restored and reference reaches described in Chapter Three could also indicate that the widespread disturbance of sand-slugs may not have as great, or as consistent an effect on biotic assemblages as initially thought (Downes et al. 2006, Lind et al. 2009). Removing the potential influence of any restoration works disturbance by focussing on un-manipulated sediment-disturbed, shallow runs, and, controlling woody debris complexity in a way that would unlikely change channel structure, could provide further insight into assemblage responses to types of habitat change, or at least, lack of habitat change. In absence of disturbances associated with channel restoration, positive assemblage-level responses to increased woody debris complexity was noted within the period of one year (Chapter Four). The positive response of adult and juvenile fish, despite sediment remaining at the sites and only minor changes to channel morphology, illustrates the importance of woody debris itself as a driver of assemblage change. Furthermore, the intrusion of saline water in deep pools suggests that channel modification alone is unlikely to have a substantial influence on fish assemblages because useable space within these pools is likely to be limited to surface waters only.

6.2. River rehabilitation: the response of fish to woody debris complexity

The distribution of woody debris, particularly spatial variation in the amount of large woody debris (LWD), was related to fish assemblage structure (composition, richness, total abundance; Chapter Four). Shallow runs with more LWD contained the highest species richness and higher total abundance, supporting earlier findings of Angermeier and Karr (1984), and Wright and Flecker (2004), that the amount of woody debris is an important contributor of influencing fish assemblage patterns. At the assemblage level, the use of shallow water with more woody debris, perhaps provides many species with a variety of resources (e.g. invertebrate rich patches)
and conditions (e.g. cover, low velocity refuge) (Chapter One). Furthermore, the use of woody debris as cover leads to segregation, which subsequently may lower biotic interactions among individuals (Sunbaum & Naslund, 1998, Crook and Robertson, 1999, Allouche 2002). Shallow waters that contain abundant woody debris could also provide an additional advantage to small fish species, by segregating resource rich patches from predators. Very few larger piscivorous fish (e.g. *Perca fluviatilis*) were observed in shallow runs to deeper pools, which may reflect avoidance of shallow water as they are vulnerable to increased risk from avian or terrestrial predator attacks (bigger-deeper hypothesis, Power 1987, Harvey and Stewart 1991). Providing suitable cover is available, small species that are vulnerable to attack from these larger predatory fish may seek refuge in shallow waters (Everett and Ruiz 1993).

For *Gadopsis marmoratus*, the largest native fish encountered in shallow runs, adult abundances were greatest where more LWD was present, supporting the importance of large pieces, most likely used for cover (Jackson 1978a, Koehn 1986, Davies 1989, Koehn et al. 1994). However, by manipulating smaller wood pieces, the study reported in Chapter Four was the first to show that juveniles may also respond to the addition of woody debris, where less LWD is present. Although the addition of small pieces of woody debris did not influence the abundance of larger individuals, neither did the addition of larger woody debris pieces in Chapter Three, despite adults being more tolerant of salinity than early life-history stages (Clunie et al. 2002). These results may suggest that the response of *G. marmoratus* to woody debris additions is perhaps more contingent upon the movement of juveniles, the life-history stage when dispersal is most likely to occur (Koster and Crook, 2007).

The role of added woody debris for juvenile *G. marmoratus* is unclear given the variety of functions woody debris provides for fish (Chapter One). It’s possible that SWD provided additional cover for these small individuals where less LWD occurred. The use of areas with woody debris by both juvenile and adult *G. marmoratus* is similar to other predatory native fish strongly affiliated with woody debris (e.g. Murray cod, Koehn 2006), suggesting that woody debris may offer structural
separation of adults and juveniles, considering, adults are known to be carnivorous and territorial (Koehn and O’Connor, 1990a). The absence of a response of larger Gadopsis marmoratus to SWD additions may indicate larger fish had selected or defended larger wood pieces first (resource preference), or, smaller pieces are unsuitable for larger fish and thus taken advantage of by smaller individuals (resource partitioning). From these results, there is little to suggest that the size-specific association with wood piece size represents an ontogenetic shift in habitat use, but it is evident that both large and small pieces are useful to G. marmoratus.

The results from Chapter Four suggest that the return of woody debris to sediment-disturbed river reaches is influential on fish assemblage composition (Table 6.1), however, further research is needed to identify how the specific characteristics or arrangement of woody debris are useful. In general, little is known about the influence of smaller woody debris pieces on riverine fish, relative to LWD (Culp et al. 1996) and this thesis is the first to investigate the relationship between woody debris size and quantity on an Australian lowland river fish assemblage. In the presence of LWD, SWD is important for some native fish species at high and low levels, and for Philypnodon grandiceps, the influence of SWD does not necessarily depend on the background amount of LWD. Although taxon richness nor total abundance was affected by the SWD treatment, the response of fish to SWD was largely species-specific with results demonstrating use by both adults and juvenile native fish.

In two configurations, added SWD was used by adult Philypnodon grandiceps as part of nesting, although, the low number of events recorded made it difficult to distinguish if structural complexity of SWD was important. Detecting the use of replaced LWD for oviposition was attempted using additional methods of an underwater camera and snorkeling (Appendix Seven). However, a trial of these techniques failed to detect the presence of oocytes placed on LWD (underwater camera), and it was not possible to access locations where oviposition was expected to occur (i.e. in tight gaps, underneath logs or inside hollows) during the corresponding trial using artificial substrata. Experimentation with other artificial
substrata, simulating different characteristics of natural woody debris (e.g. hollows), was used to compare the spawning potential between different channel and reach types, and also among different types of substrate used (Table 6.1). Artificial substrata with small openings were favoured twice as much as large openings. Several fish species including *G. marmoratus* and *N. variegata* used artificial substrata, but only two species: *Philypnodon grandiceps* and *Hypseleotris* sp. spawned. Furthermore, *Philypnodon grandiceps* regularly used artificial substrata in shallow water (< 1.2 m), but never in deep water (> 2.5 m) across all reaches (Chapter Five). Pools contained the highest frequency of spawning events, indicating that pools are important habitat patches in sediment-disturbed rivers, particularly at vulnerable times (e.g. low flow periods, Chapter One). A substantially higher number of spawning events in rehabilitated runs compared to unmanipulated, shallow runs was quite surprising and encouraging for environmental managers, suggesting that benthic ovipositing fishes with parental care may take advantage of the provision of spawning substrata and stable water depth from rehabilitation of sediment-disturbed reaches (Chapter Five).

Naturally, there are many types and uses of woody debris valued by fish (Chapter One). This thesis supports the addition of woody debris as a useful tool to address habitat degradation in sediment-disturbed rivers; however, fish responses to restoration may depend upon where, how and when woody debris is added to rivers. In expectation that fish will respond positively to additions of woody debris in rivers, small-scale (e.g. wood arrangement, disturbance attributed to restoration procedures) as well as large-scale (e.g. water quality, climate conditions) factors require further consideration. To understand how environmental modifications could be more beneficial to fish, it is important to identify how woody debris is used by fish and factors precluding its use.

6.3. Implications for future river rehabilitation projects – project assessment
Traditionally, our understanding of fish-habitat relationships in river ecosystems has largely been based on surveys examining the distribution of fish and associated habitat components, predominantly over small scales (Fausch et al. 2002). However, these studies do not reveal anything about the processes that lead to the colonisation of fish and their association with changes to habitat structure. Manipulative field experiments are particularly useful for gaining insight into the mechanics of aquatic ecosystems (Kingsford et al. 1998), but they have not been extensively used to understand functional relationships between freshwater fish and habitat structure in Australian rivers. This is perhaps due to great difficulties and costs associated with manipulating important fish habitat components, for example, the transport and re-introduction of large woody debris over the potential range of spatial scales that fish may move.

River rehabilitation works offer the opportunity to do such experiments, albeit, often with limited replication of the ‘impact’ location. Some fish responses to the addition of instream structures (abundance increase of 1.7 to 5 times or more, see Table 1.1) may be great enough to permit the detection of statistical differences among experimental groups, even with low replication. The inclusion of a greater number of rehabilitated locations was not possible for the present study because of the logistics and costs involved. Project costs often limit the amount of restoration undertaken, and where different techniques are employed across reaches to address the various types of degradation that are present, restoration may not be truly replicated. Therefore, understanding the influence of river rehabilitation may rely on the analysis of projects on a case-by-case basis. Ongoing monitoring of individual projects also becomes critical for gauging the impact of experimental techniques and guiding future projects in the same region. In these situations, increasing the number of control locations relative to treatment locations (c.f. asymmetrical design) could be helpful (Glasby 1997, Underwood 1997).

Fish assemblage responses to restoration were expected within the time frame of this Ph.D. study, considering most species in the Glenelg River were small, short-lived (few years) and widely distributed, and a range of other fish species are
reported to respond within a short time (1 year, Table 1.1). One winter season within one year was available for the before-period, which provided some description of fish assemblages in the year prior to restoration and to the effects of drought. Although the period of monitoring in Chapter Three was similar to the average before-period of fish monitoring in the literature (1.2 ± 1.1 years [1 s.d.], Table 1.1), at least multiple seasons within multiple years are recommended in future studies. Similarly, two years of monitoring after restoration works encompassing winter and summer seasons, is comparable to the average after-period time (2.4 ± 1.6 [1 s.d.] years) of monitoring for other published studies in Table 1.1. In the presence of possible on-going disturbance such as droughts, it is advised that the period of monitoring should be extended, because patterns of colonisation and exclusion in response to environmental conditions and other biotic interactions may only become apparent after several years (Closs and Lake 1996).

6.4. Implications for future river rehabilitation projects – scaling effects

By definition, river rehabilitation projects are usually conducted in degraded reaches that are surrounded by larger disturbances as a result of catchment development. The consequences of conducting smaller-scale rehabilitation programs in larger disturbed landscapes has been identified as a hurdle (Larson et al. 2001, Pretty et al. 2003, Harrison et al. 2004, Bond and Lake 2005, Palmer 2009, Sundermann et al. 2011). In the Glenelg River catchment, multiple disturbances have been identified that extend over different temporal and spatial scales, which can further compound existing problems (e.g. flow reductions to a channel full of sediment) (Lind et al. 2009, Robson and Mitchell 2010). Recognition of ‘barriers’ to achieving restoration goals is important and may indeed require a more systematic approach when conducting restoration works for restoration to become effective (Recovery cascade model sensu Robson et al. 2011).

Temporal changes in assemblage structure reported in Chapter Four indicated significant movement of fishes into shallow runs. Fish movement is a common
means of explaining the rapid response of fish to rehabilitated locations (Gowan and Fausch 1996, Antón et al. 2011). Scales of movement in Australian native fish are poorly known, especially for small sized individuals (e.g. juveniles) and species (Koehn and O’Connor 1990a, Crook et al. 2001, Baumgartner 2005). Some species (e.g. *Gadopsis marmoratus*) display high site fidelity (Koehn 1986, Kahn et al. 2004, Koster and Crook 2007) and therefore movements into new locations may not be common. In contrast, Gowan and Fausch (1996) studying trout (*Salvelinus fontinalis, Salmo trutta*) in high Rocky Mountain streams, USA, found that large scale movements of adults contributed to observed increases in abundance and biomass, whereas differences were not observed for juvenile recruitment, survival or growth. They also observed large variations in fish abundances within and among streams suggesting that larger regional scale effects influenced fish over larger scales.

Knowledge on rates of fish movements, particularly, abundance fluctuations amongst locations attributed to movement is critical to predicting recovery trajectories in river restoration, especially if reaches are lacking fish prior to restoration. Movement patterns are also important to determine as it may provide clues to assemblage stability at rehabilitated locations and ultimately help to address the question: will the restoration effect last? For taxon that are well known to move (i.e. salmonids), a recent study has demonstrated that the persistence of restored pools sustained higher adult trout abundance two decades on (White et al. 2011). Other studies (e.g. Brooks et al. 2006) have indicated responses to restoration persisted for a short time (< 2 years), although species driving assemblage composition in Brooks et al. (2006) were also known to migrate as juveniles (e.g. *Retropinna semoni, Gobiomorphus coxii*). If migration is an important process driving the response to restoration, then, interruptions to the migration or variation in the recruitment process, independent of habitat restoration, may explain why only short-term effects were observed (i.e. supply-side ecology, Underwood and Fairweather, 1989). Short-term and long-term effects of restoration could also reflect the temporary function or specific use of habitat patches, which may vary between different age-cohorts or sizes among species (Schlosser 1991, Koehn et al. 1994, Solazzi et al. 2000, Crook and Gillanders 2006).
Identifying the links between patches, which fish move to and away from is likely to yield important insight on the use of patches across river landscapes and is an avenue for further research (Schlosser 1991, Fausch et al. 2002, Crook and Gillanders 2006).

Understanding reasons for movement can be just as important as determining the extent of movement. Adult movements into restored locations may reflect other processes such as migration (i.e. detection at restored sites on the way to somewhere else), or, requirements such as reproduction, where spawning may represent an important ecological end point of movement. Apart from salmonids, fish use of restored habitat locations for reproduction is rarely examined, although, as shown in the present study, local spawning and recruitment may offer another pathway for the recolonisation of restored locations, particularly if isolation of restored reaches exist because of barriers such as reduced flow (e.g. drying of shallow areas, larvae not travelling far in drift, Robson et al. 2011). Identifying the function of restored locations for fish across river landscapes is imperative, although the reasons for the purpose of fish reaching (by choice or through displacement) these locations is not clear for many current studies of restoration. This certainly remains an important area for future research.

There is some concern that restoration techniques developed and implemented across river landscapes may only benefit particular taxon (e.g. Salmonids, Roni et al. 2005). The results of this study (Chapter Four) also suggest that habitat rehabilitation techniques may not benefit all species in an assemblage. Therefore, it is important to consider the limits of rehabilitation for species that are less abundant or rare. Rare species might have higher conservation value than other species, which may initiate habitat restoration specifically to assist these species. Therefore, a strategy consisting of a suite of complementary rehabilitation techniques (i.e. captive breeding programs, assisting dispersal by removing barriers) in addition to sediment extraction and woody debris replacement may be needed in order to elicit a positive response by rare species. Rare species have little influence on patterns of fish assemblage structure (Capone and Kushlan 1991), so
determining the response of rare species to rehabilitation programs is likely to demand more resources, especially if other potential effects are of interest (e.g. predator influences). Another limitation of focusing on rare species is that more effort will be required to obtain sufficient sample sizes of rare species in order to make valid comparisons across locations or through time.

Current knowledge of the outcomes of habitat restoration projects shows benefits to populations of species already well established within the system. In Chapters Three, Four and Five, species that were locally abundant prior to rehabilitation responded best to the structures in the short term, most likely due to individuals colonising from adjoining reaches or nearby environments (Gowan and Fausch 1996, Antón et al. 2011). Rehabilitation might benefit dominant species in an assemblage because they successfully tolerate local environmental conditions (i.e. they are better adapted to other forms of disturbance and therefore have characteristics that allow them to respond to improved habitat better). For some introduced species (e.g. *Cyprinus carpio*) that dominate fish assemblages, the possibility of a positive response to rehabilitation procedures is of concern (Nicol et al. 2004). Habitat rehabilitation may be questionable if procedures only benefit existing, abundant species that are at the detriment of other native species. Therefore, habitat alterations in conjunction with other management strategies may be required to achieve river management or conservation objectives regarding the whole fish assemblage.

6.5. Conclusion

To date, there are no examples of rehabilitation procedures demonstrating complete biotic recovery in a river impacted by sand-slugs. In agreement with Rutherfurd and Budahazy (1996), the results of this thesis support the claim that this is partly due to the type of disturbance. For example, processes of disturbance, such as land clearing, have occurred over several decades and have led to one or more ‘press disturbances’. The continual downstream movement of sand is one example that can continuously impact the river as it moves downstream over long
time scales. Under press disturbances such as sedimentation, habitat improvement has been suggested to take years to decades, yet without intervention it could take longer (Allan and Flecker 1993). Understanding the ecological impacts and their recovery from the result of sand-slugs in Australian lowland river systems, is further hampered by a general deficiency of ecological research and monitoring of restoration efforts (Downes et al. 2006). This thesis has contributed to this area of research by providing a greater understanding of restoration efforts in sediment disturbed rivers (significant outcomes are outlined below) by using a combination of spatial comparisons of fish assemblages and manipulative field experiments.

Many restoration strategies, including those used in sand-slug rivers are based on those applied in other types of systems or are modelled on historical accounts of river structure. The type of restoration procedure studied in this thesis was based on reconstructing a reach that reflected existing knowledge of historic river structure and places where native fish are known to reside (i.e. large pools and LWD) with additional modifications performed to increase fish accessibility to the reach (modification of river crossings and incorporation of a fishway). Other identifiable components of the river environment (SWD) were also investigated and found to influence fish assemblage structure. This study has demonstrated that the use of sediment extraction and small woody debris replacement can be a useful technique, associated with positive responses from fish in a sediment disturbed lowland river. Given the short period of monitoring and limited number of restored reaches, the results suggest:

- That restoration works are having a minimal short-term or longer term negative impacts on fish assemblages. Numerous fish species were observed within the rehabilitated reaches within a relatively short period of time (i.e. 6 months of completed works).

- That rehabilitated reaches did not lead to longer-term changes in fish assemblages or to increases in species richness or total abundances compared to un-manipulated reaches within two years of completed works.
• Large-scale disturbances (i.e. drought) affect water quality and quantity, which corresponded with decreased abundances of several species, potentially delaying any positive response to rehabilitation efforts.

• Saline water intrusion limits usefulness of deep pools. Larger efforts from other habitat rehabilitation procedures are needed to increase mixing (e.g. environmental flows, Coates and Mondon 2009).

• Fish assemblages in a lowland river do respond to woody debris complexity (i.e. size and amount) and that small woody debris (SWD) can influence fish assemblages in reaches with less LWD.

• Effects of woody debris replacement can be subtle. Different species and different size classes within a species may respond differently to manipulations of woody debris.

• In addition to providing habitat structure and cover, reaches of increasing depth, together with the addition of woody debris in sediment disturbed rivers, may assist fish spawning. The detection of fish spawning and movement into rehabilitated locations indicates that different processes may alter fish assemblage structure over time.

• SWD is useful alternative/additional consideration for restoration programs in sediment-disturbed rivers. Benefits include reduced cost and greater ease of installation. Trade-offs equal less permanent fixtures that may need to be replaced on a regular basis if adjacent riparian vegetation cannot supply the necessary demands.

Typically, habitat enhancement type projects alter only parts of the physical and/or the chemical environment to improve locations for biota. Conditions as well as resources need to be considered as part of habitat restoration, given that habitat
can change under uncontrolled (natural & some anthropogenic disturbances) as well as controlled (rehabilitation) circumstances. Understanding how habitat influences biota over a range of spatial and temporal scales is required to best predict organism responses to rehabilitation.

Community willingness to combat river degradation will ensure the continuing popularity of habitat rehabilitation programs in Australian rivers. Many systems are still experiencing stress even after problems have long been identified (Karr and Chu 1999), meaning, many rehabilitation efforts are being trialled and hence, judged under conditions of continuing environmental degradation. Careful planning and consideration of a range of factors is needed when confronted by the challenge of undertaking a restoration project. Otherwise, it is unrealistic to expect current rehabilitation strategies to address systems with multiple disturbances, especially in the short-term (Ziemer, 1999).

In conclusion, while this thesis has contributed to a more nuanced understanding of rehabilitation processes and has facilitated a greater understanding of issues and complexities associated with these processes in an Australian context, it must be recognised that this is an under researched area, and the studies conducted as part of this thesis have raised many more questions that require further research and understanding. Overall, there is a need to continue to build a solid evidence base by using monitoring and experimental techniques that can be used as a cost-effective, outcome driven resource for managers of rivers to ensure the best use of available resources to achieve positive outcomes for degraded river systems.
Table 6.1 Thesis research questions and null hypotheses examined concerning tests of rehabilitation.

<table>
<thead>
<tr>
<th>Question</th>
<th>Null hypotheses</th>
<th>Outcome</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Does the implementation of a reach scale river rehabilitation program alter fish assemblage structure?</td>
<td>$H_0$: The rehabilitated reach does not contain a different assemblage composition compared to un-manipulated locations, before to after the completion of restoration works. $H_0$: Species richness, individual or group and total taxa abundances are not greater in the rehabilitated reach compared to un-manipulated locations after the completion of works. $H_0$: Water quality parameters do not vary between locations or relate to assemblage composition.</td>
<td>$H_0$: accepted. The rehabilitated reach did not contain an assemblage that was different to control reaches. $H_0$: accepted (species richness, individual and total taxa abundances). Species richness, individual and total taxa abundances did not differ between rehabilitated and control reaches through time. $H_0$: rejected. Water quality variables did differ between reference and other reaches. No difference in water quality between rehabilitated and control reaches. $H_0$: rejected. Water quality related to assemblage composition among rehabilitated and control reaches through time.</td>
</tr>
<tr>
<td>2. Does the quantity and size of woody debris influence fish assemblage structure?</td>
<td>$H_0$: In channels impacted by sediment, the amount of LWD present has no effect on assemblage structure (composition, richness, total and taxon abundance). $H_0$: In channels impacted by sediment with differing amounts of LWD, adding SWD has no effect on increasing richness, total, taxon, juvenile or adult abundance or produce a different assemblage composition. $H_0$: Other habitat components do not differ among LWD and SWD treatment groups, including any changes attributed through time.</td>
<td>$H_0$: rejected (composition, species richness, total abundance and for $G. marmoratus$ abundance). Fish assemblage structure differed with amount of LWD present. $H_0$: accepted (species richness, total abundance, adult $G. marmoratus$ and $N. variegata$ abundance). There was no effect of SWD addition. $H_0$: rejected (juvenile $G. marmoratus$ abundance and composition in Low LWD; $G. olidus$ abundance in High LWD; $P. grandiceps$ and adult $P. grandiceps$ abundance both High and Low LWD). Assemblage composition, species and size abundances differed with addition of SWD. $H_0$: rejected (submerged and emergent macrophytes, depth, width). Low LWD had more submerged macrophytes than High LWD sites. Low LWD SWD control had more emergent macrophytes than other groups. Low LWD SWD treatment group locations were shallower and wider than...</td>
</tr>
</tbody>
</table>
other groups. Stream width varied between SWD control and treatment locations in Before-period but not in After-period.

3. Does rehabilitation consisting of woody debris replacement and sediment extraction influence fish spawning?

<table>
<thead>
<tr>
<th>Hypothesis</th>
<th>Conclusion</th>
</tr>
</thead>
<tbody>
<tr>
<td>$H_0$: Spawning frequency on artificial substrata does not differ among channel types (pools and runs) and reach types (rehabilitated and un-manipulated).</td>
<td>$H_0$: rejected (for PVC substrata). Spawning frequency for <em>P. grandiceps</em> did spatially differ across reach types and channel types.</td>
</tr>
<tr>
<td>$H_0$: In pools, spawning frequency on artificial substrata does not vary between deep and shallow zones or across rehabilitated and un-manipulated reaches.</td>
<td>$H_0$: rejected. Evidence of oviposition recorded for both SWD branches and bundle arrangements. Null hypothesis unable to be tested for replaced LWD</td>
</tr>
<tr>
<td>$H_0$: Replaced woody debris does not function as a spawning substrate</td>
<td>$H_0$: rejected. Small hole (25 mm) substrata used more frequently than large hole (50 mm) by <em>P. grandiceps</em> for oviposition. Null hypothesis for SWD substrates unable to be tested</td>
</tr>
<tr>
<td>$H_0$: Artificial substrate complexity does not influence the frequency of spawning events</td>
<td>$H_0$: rejected (run depth, macrophyte cover) Harrow rehabilitated runs deeper than other reaches. Macrophyte cover greater in Harrow than Casterton reaches</td>
</tr>
<tr>
<td>$H_0$: Habitat characteristics including physicochemical parameters do not differ among reaches</td>
<td>$H_0$: rejected (PVC substrata) Spawning frequency for <em>P. grandiceps</em> did spatially differ across reach types and channel types.</td>
</tr>
</tbody>
</table>

**Null hypothesis unable to be tested**
References


Green, R. H. (1979) *Sampling design and statistical methods for environmental biologists*. John Wiley, New York, USA.


Appendix One: SWD treatment characteristics

Summary of the characteristics of 10 individual SWD treatment branches placed in SWD treatment sites during November 2003, after the initial before-period fish samples were collected. Note: values represent means ± 1 standard deviation.

<table>
<thead>
<tr>
<th>Treatment site</th>
<th>Number of pieces</th>
<th>Diameter (cm)</th>
<th>Perimeter (m)</th>
<th>No. branches</th>
<th>Number of individuals with leaves</th>
<th>Number of individuals with bark</th>
<th>Distance to Bank (m)</th>
<th>Distance to SWD (m)</th>
<th>Distance to LWD (m)</th>
<th>Length of Trunk (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gorge High</td>
<td>10</td>
<td>6.0 ± 1.1</td>
<td>4.8 ± 2.2</td>
<td>13 ± 21.9</td>
<td>2</td>
<td>6</td>
<td>1.5 ± 1.6</td>
<td>0.5 ± 0.8</td>
<td>0.5 ± 0.9</td>
<td>2.2 ± 1.0</td>
</tr>
<tr>
<td>Gorge Low</td>
<td>10</td>
<td>5.7 ± 2.0</td>
<td>5.3 ± 1.4</td>
<td>13 ± 16.4</td>
<td>0</td>
<td>7</td>
<td>2.3 ± 1.1</td>
<td>0.9 ± 0.5</td>
<td>1.8 ± 0.9</td>
<td>2.4 ± 0.7</td>
</tr>
<tr>
<td>Balmoral High 1</td>
<td>10</td>
<td>4.1 ± 1.5</td>
<td>5.2 ± 1.6</td>
<td>15 ± 11.7</td>
<td>2</td>
<td>7</td>
<td>0.7 ± 3.3</td>
<td>0.4 ± 0.4</td>
<td>0.6 ± 1.5</td>
<td>2.2 ± 0.9</td>
</tr>
<tr>
<td>Five Mile Low 1</td>
<td>10</td>
<td>4.4 ± 2.7</td>
<td>5.5 ± 1.0</td>
<td>13 ± 6.1</td>
<td>1</td>
<td>7</td>
<td>3.5 ± 1.8</td>
<td>0.7 ± 0.9</td>
<td>1.8 ± 1.8</td>
<td>2.5 ± 0.6</td>
</tr>
<tr>
<td>Fulhams Bridge Low</td>
<td>10</td>
<td>5.6 ± 2.0</td>
<td>5.2 ± 1.7</td>
<td>11 ± 31.1</td>
<td>1</td>
<td>9</td>
<td>2.7 ± 2.5</td>
<td>0.8 ± 1.0</td>
<td>2.3 ± 1.7</td>
<td>2.3 ± 0.7</td>
</tr>
<tr>
<td>Harrow High 2</td>
<td>10</td>
<td>4.8 ± 2.3</td>
<td>5.5 ± 1.7</td>
<td>28 ± 30.2</td>
<td>0</td>
<td>9</td>
<td>4.4 ± 2.5</td>
<td>0.9 ± 1.0</td>
<td>3.5 ± 1.7</td>
<td>2.3 ± 0.7</td>
</tr>
</tbody>
</table>
Appendix Two: Water quality across twelve, sediment-disturbed locations in the Glenelg River from June 2003 to July 2004

Mean (± 1 standard deviation) temperature, dissolved oxygen, pH, salinity, conductivity and turbidity values collected at twelve run sites during trips (1, 4, 5, 7 & 8) during the before and after time periods.

<table>
<thead>
<tr>
<th>Site</th>
<th>Temperature (°C)</th>
<th>Dissolved Oxygen (mg L⁻¹)</th>
<th>Dissolved Oxygen saturation (%)</th>
<th>pH</th>
<th>Salinity (g L⁻¹)</th>
<th>Conductivity (μS @ 25 °C)</th>
<th>Turbidity (NTU)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Balmoral High 1</td>
<td>11.1 ± 4.6</td>
<td>9.1 ± 4.3</td>
<td>81.6 ± 32.2</td>
<td>7.7 ± 0.4</td>
<td>1.7 ± 0.4</td>
<td>3377.5 ± 901.8</td>
<td>4.9 ± 6.2</td>
</tr>
<tr>
<td>Balmoral High 2</td>
<td>12.1 ± 6.5</td>
<td>8.2 ± 2.3</td>
<td>75.9 ± 11.0</td>
<td>7.5 ± 0.2</td>
<td>2.2 ± 0.1</td>
<td>3796.8 ± 164.7</td>
<td>3.4 ± 3.3</td>
</tr>
<tr>
<td>Balmoral Low</td>
<td>13.1 ± 6.0</td>
<td>9.4 ± 4.5</td>
<td>88.6 ± 24.5</td>
<td>7.9 ± 0.4</td>
<td>1.8 ± 0.5</td>
<td>3482.5 ± 875.0</td>
<td>1.8 ± 2.5</td>
</tr>
<tr>
<td>Five Mile High</td>
<td>13.2 ± 4.2</td>
<td>8.9 ± 2.5</td>
<td>82.9 ± 14.1</td>
<td>7.6 ± 0.3</td>
<td>1.6 ± 0.5</td>
<td>3132.8 ± 958.6</td>
<td>6.8 ± 7.5</td>
</tr>
<tr>
<td>Five Mile Low 1</td>
<td>10.0 ± 0.5</td>
<td>10.2 ± 1.1</td>
<td>88.0 ± 5.7</td>
<td>7.7 ± 0.1</td>
<td>2.1 ± 0.2</td>
<td>3908.0 ± 382.0</td>
<td>9.4 ± 0.6</td>
</tr>
<tr>
<td>Five Mile Low 2</td>
<td>14.1 ± 6.3</td>
<td>9.2 ± 2.1</td>
<td>87.0 ± 9.8</td>
<td>7.6 ± 0.2</td>
<td>2.1 ± 0.1</td>
<td>3840.6 ± 248.7</td>
<td>7.0 ± 1.5</td>
</tr>
<tr>
<td>Fulham’s Bridge Low</td>
<td>14.3 ± 6.0</td>
<td>9.2 ± 1.8</td>
<td>89.1 ± 4.0</td>
<td>7.7 ± 0.1</td>
<td>1.6 ± 0.6</td>
<td>2978.2 ± 907.0</td>
<td>2.6 ± 3.3</td>
</tr>
<tr>
<td>Gorge High</td>
<td>13.1 ± 5.1</td>
<td>8.5 ± 3.2</td>
<td>78.1 ± 21.0</td>
<td>7.7 ± 0.2</td>
<td>1.7 ± 0.5</td>
<td>3039.4 ± 770.1</td>
<td>2.2 ± 2.1</td>
</tr>
<tr>
<td>Gorge Low</td>
<td>13.4 ± 4.7</td>
<td>8.3 ± 2.8</td>
<td>78.0 ± 17.5</td>
<td>7.6 ± 0.3</td>
<td>1.9 ± 0.5</td>
<td>3357.6 ± 622.9</td>
<td>4.7 ± 5.3</td>
</tr>
<tr>
<td>Harrow High 1</td>
<td>11.1 ± 6.0</td>
<td>12.0 ± 0.6</td>
<td>101.5 ± 3.0</td>
<td>8.0 ± 0.2</td>
<td>2.1 ± 0.4</td>
<td>3790.8 ± 481.4</td>
<td>6.8 ± 1.2</td>
</tr>
<tr>
<td>Harrow High 2</td>
<td>10.7 ± 4.2</td>
<td>11.9 ± 1.8</td>
<td>101.5 ± 11.2</td>
<td>8.0 ± 0.1</td>
<td>2.0 ± 0.5</td>
<td>3712.0 ± 625.2</td>
<td>6.3 ± 1.7</td>
</tr>
<tr>
<td>Harrow Low</td>
<td>11.0 ± 5.4</td>
<td>10.5 ± 1.0</td>
<td>89.6 ± 4.0</td>
<td>8.0 ± 0.2</td>
<td>2.0 ± 0.4</td>
<td>3841.0 ± 603.0</td>
<td>4.9 ± 4.7</td>
</tr>
</tbody>
</table>
Appendix Three: Habitat characteristics of sediment-disturbed channels containing differing LWD amounts

Habitat characteristics of twelve, sediment-disturbed runs containing differing amounts of LWD: a) mean depth (n = 50 per site) across LWD and SWD groups, b) mean depth (n = 10 per site, per trip) across LWD and SWD groups among trips. Mean stream depth among SWD treatment and control groups, c) between High and Low LWD amounts, d) between before and after-periods. Proportion of emergent macrophytes distributed among High and Low LWD, SWD treatment and control locations (e). Proportion of submerged macrophytes distributed across low and high LWD locations (f).
Appendix Four: Examples of Artificial substrata used to detect fish spawning

Examples of spawning substrata types used to determine fish spawning distributions in the Glenelg River: a) SWD (Bundle form), b) PVC tube (Large), and c) searching for artificial substrata
Appendix Five: Habitat characteristics of pools and runs survey for spawning within rehabilitated and unmanipulated reaches in the Glenelg River

Measures of environmental attributes surrounding spawning tube substrates in restored (R) and control (C) reaches of the Glenelg River. Values for distance to components, river height and depth of PVC substrate represent means (± standard deviation). Note: † denotes distance > 10 m for all tubes, ES, Existing Single LWD piece; PS, Placed Single LWD piece.

<table>
<thead>
<tr>
<th>Location</th>
<th>Shallow zone area (m²)</th>
<th>Mean dist. to LWD (m)</th>
<th>Dominant LWD Type</th>
<th>Mean dist. to emergent macrophytes (m)</th>
<th>Mean dist. to submerged macrophytes (m)</th>
<th>River height (m)</th>
<th>Mean distance of PVC substrate to bank (m)</th>
<th>Dominant benthic substrate</th>
<th>Mean depth of PVC substrate (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harrow C P1</td>
<td>564</td>
<td>3.6 ± 3.1</td>
<td>ES</td>
<td>0.6 ± 0.6</td>
<td>0.1 ± 0.1</td>
<td>1.03 ± 0.02</td>
<td>1.2 ± 0.5</td>
<td>CLAY &amp; SILT</td>
<td>0.6 ± 0.3</td>
</tr>
<tr>
<td>Harrow C P2</td>
<td>434</td>
<td>1.6 ± 0.9</td>
<td>ES</td>
<td>0.4 ± 0.2</td>
<td>0.3 ± 0.3</td>
<td>1.06 ± 0.04</td>
<td>3.2 ± 1.8</td>
<td>SILT</td>
<td>0.6 ± 0.3</td>
</tr>
<tr>
<td>Harrow C R1</td>
<td>480</td>
<td>1.3 ± 1.2</td>
<td>ES</td>
<td>0.7 ± 0.6</td>
<td>0.3 ± 0.3</td>
<td>1.11 ± 0.07</td>
<td>1.9 ± 1.0</td>
<td>SAND</td>
<td>0.5 ± 0.3</td>
</tr>
<tr>
<td>Harrow C R2</td>
<td>480</td>
<td>1.2 ± 0.8</td>
<td>ES</td>
<td>0.3 ± 0.4</td>
<td>0.7 ± 0.9</td>
<td>1.12 ± 0.07</td>
<td>1.7 ± 1.0</td>
<td>SAND</td>
<td>0.3 ± 0.1</td>
</tr>
<tr>
<td>Harrow R P1</td>
<td>539</td>
<td>1.4 ± 1.2</td>
<td>ES</td>
<td>0.5 ± 0.3</td>
<td>3.4 ± 3.3</td>
<td>1.02 ± 0.11</td>
<td>1.4 ± 1.1</td>
<td>SAND</td>
<td>0.6 ± 0.2</td>
</tr>
<tr>
<td>Harrow R P2</td>
<td>460</td>
<td>3.1 ± 2.7</td>
<td>ES</td>
<td>0.4 ± 0.7</td>
<td>0.9 ± 1.0</td>
<td>0.97 ± 0.14</td>
<td>3.4 ± 3.0</td>
<td>SAND</td>
<td>0.5 ± 0.2</td>
</tr>
<tr>
<td>Harrow R R1</td>
<td>540</td>
<td>1.8 ± 1.3</td>
<td>PS</td>
<td>1.7 ± 1.4</td>
<td>2.4 ± 2.2</td>
<td>1.01 ± 0.11</td>
<td>4.8 ± 2.5</td>
<td>SAND</td>
<td>0.7 ± 0.2</td>
</tr>
<tr>
<td>Harrow R R2</td>
<td>420</td>
<td>2.2 ± 1.8</td>
<td>PS</td>
<td>0.3 ± 0.3</td>
<td>0.2 ± 0.4</td>
<td>1.01 ± 0.09</td>
<td>6.9 ± 2.7</td>
<td>SAND</td>
<td>0.7 ± 0.2</td>
</tr>
<tr>
<td>Casterton C P1</td>
<td>560</td>
<td>3.8 ± 3.1</td>
<td>ES</td>
<td>2.4 ± 2.9</td>
<td>0.1 ± 0.2</td>
<td>0.89 ± 0.11</td>
<td>3.4 ± 3.1</td>
<td>SILT</td>
<td>0.6 ± 0.2</td>
</tr>
<tr>
<td>Casterton C P2</td>
<td>552</td>
<td>2.1 ± 1.4</td>
<td>ES</td>
<td>2.6 ± 1.5</td>
<td>0.3 ± 0.4</td>
<td>0.86 ± 0.06</td>
<td>1.9 ± 1.4</td>
<td>SAND/SILT</td>
<td>0.5 ± 0.3</td>
</tr>
<tr>
<td>Casterton C R1</td>
<td>455</td>
<td>3.3 ± 1.8</td>
<td>ES</td>
<td>0.4 ± 0.3</td>
<td>0.7 ± 1.4</td>
<td>0.83 ± 0.09</td>
<td>3.3 ± 1.8</td>
<td>SAND</td>
<td>0.4 ± 0.2</td>
</tr>
<tr>
<td>Casterton C R2</td>
<td>540</td>
<td>2.8 ± 2.3</td>
<td>ES</td>
<td>1.7 ± 0.9</td>
<td>2.2 ± 1.9</td>
<td>0.91 ± 0.05</td>
<td>2.1 ± 0.8</td>
<td>SAND</td>
<td>0.5 ± 0.2</td>
</tr>
<tr>
<td>Casterton R P1</td>
<td>462</td>
<td>4.4 ± 3.5</td>
<td>ES</td>
<td>1.6 ± 2.4</td>
<td>1.0 ± 1.6</td>
<td>0.84 ± 0.08</td>
<td>2.0 ± 2.4</td>
<td>SAND</td>
<td>0.6 ± 0.2</td>
</tr>
<tr>
<td>Casterton R P2</td>
<td>576</td>
<td>5.8 ± 4.0</td>
<td>ES</td>
<td>2.3 ± 2.2</td>
<td>†</td>
<td>0.84 ± 0.07</td>
<td>2.8 ± 1.9</td>
<td>SAND</td>
<td>0.5 ± 0.2</td>
</tr>
<tr>
<td>Casterton R R1</td>
<td>420</td>
<td>1.6 ± 1.4</td>
<td>PS</td>
<td>2.4 ± 2.1</td>
<td>5.6 ± 3.7</td>
<td>0.84 ± 0.08</td>
<td>2.6 ± 1.7</td>
<td>SAND</td>
<td>0.4 ± 0.2</td>
</tr>
<tr>
<td>Casterton R R2</td>
<td>420</td>
<td>1.9 ± 1.6</td>
<td>PS</td>
<td>1.1 ± 1.0</td>
<td>†</td>
<td>0.84 ± 0.07</td>
<td>3.0 ± 2.0</td>
<td>SAND</td>
<td>0.4 ± 0.1</td>
</tr>
</tbody>
</table>
Appendix Six: Explained variance of principal components and associated correlations with habitat variables across rehabilitated and un-manipulated reaches in the Glenelg River

Principal Component Analysis identifying relationships between environmental variables characterising the position of PVC spawning substrates and location. Note: correlations between variables and principal components larger than 0.4 are underlined.

<table>
<thead>
<tr>
<th>Variable</th>
<th>PC 1</th>
<th>PC 2</th>
<th>PC 3</th>
<th>PC 4</th>
<th>PC 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>LWD pieces per area</td>
<td>0.15</td>
<td>0.06</td>
<td>-0.31</td>
<td>0.54</td>
<td>0.20</td>
</tr>
<tr>
<td>Proportion of macrophyte cover</td>
<td>0.42</td>
<td>-0.14</td>
<td>-0.16</td>
<td>0.06</td>
<td>-0.03</td>
</tr>
<tr>
<td>Depth of substrate</td>
<td>0.05</td>
<td>-0.25</td>
<td>-0.30</td>
<td>-0.44</td>
<td>0.31</td>
</tr>
<tr>
<td>Distance to nearest emergent macrophyte</td>
<td>-0.17</td>
<td>0.13</td>
<td>-0.35</td>
<td>-0.22</td>
<td>0.33</td>
</tr>
<tr>
<td>Distance to nearest submerged macrophyte</td>
<td>-0.26</td>
<td>0.05</td>
<td>0.44</td>
<td>-0.06</td>
<td>0.38</td>
</tr>
<tr>
<td>Distance to nearest bank</td>
<td>0.02</td>
<td>-0.20</td>
<td>-0.19</td>
<td>-0.52</td>
<td>0.12</td>
</tr>
<tr>
<td>Distance to nearest LWD piece</td>
<td>-0.04</td>
<td>-0.01</td>
<td>0.04</td>
<td>-0.31</td>
<td>-0.71</td>
</tr>
<tr>
<td>Mean river temperature</td>
<td>-0.01</td>
<td>0.60</td>
<td>-0.01</td>
<td>-0.09</td>
<td>0.01</td>
</tr>
<tr>
<td>Variance in river temperature</td>
<td>-0.40</td>
<td>-0.24</td>
<td>-0.11</td>
<td>0.11</td>
<td>-0.03</td>
</tr>
<tr>
<td>Mean river height</td>
<td>0.44</td>
<td>0.04</td>
<td>0.10</td>
<td>-0.02</td>
<td>0.18</td>
</tr>
<tr>
<td>Variance in river height</td>
<td>-0.01</td>
<td>-0.57</td>
<td>0.00</td>
<td>0.20</td>
<td>-0.11</td>
</tr>
<tr>
<td>Mean dissolved oxygen concentration</td>
<td>-0.14</td>
<td>0.14</td>
<td>-0.62</td>
<td>0.15</td>
<td>-0.17</td>
</tr>
<tr>
<td>Mean pH</td>
<td>-0.45</td>
<td>0.18</td>
<td>-0.10</td>
<td>0.03</td>
<td>-0.06</td>
</tr>
<tr>
<td>Mean salinity</td>
<td>-0.35</td>
<td>-0.25</td>
<td>0.10</td>
<td>0.13</td>
<td>0.14</td>
</tr>
</tbody>
</table>
Appendix Seven: Trial of underwater visual techniques for determining the presence of oocytes on replaced LWD

Objective

The use of underwater visual techniques, including underwater videography and snorkelling to identify the presence of fish oocytes and species type

Methods and materials

Environmental conditions

Surveys were conducted during low flows on the 27 and 28th of January 2005. Weather conditions were sunny, with few clouds and little wind. Water clarity was high, a secci-disc could be seen all the way to the bottom in 1 m of water.

Oocyte collection

Artificial substrata (PVC pipe, 25 & 50 mm diameter, 200 & 400 mm long) containing a fine flyscreen mesh were used to collect oocytes of Philypnodon grandiceps (Krefft), a species known to use hard substrates for oviposition (Larson and Hoese 1996) and regularly used artificial substrata in restored reaches (Chapter Five). P. grandiceps oocytes were considered useful, as they are highly recognisable (pointed at one end, blunt, wide and round at the other) and thus could be used to determine if visual techniques can at least, identify egg shape. Oocytes were deposited on flyscreen mesh within artificial PVC substrates. The flyscreen mesh was removed from artificial substrates and attached to replaced LWD pieces using cable ties, approximately 0.4 m from the surface.
Underwater camera setup

An inexpensive, underwater video camera (Swann SW-L-UWC, Swann Communications Pty. Ltd. Australia) was trialled with an AKAI colour monitor (ATF-M4270, AKAI Electric Co. Ltd. Japan) (Plate 7.1). The monitor was mounted in a small aluminium case which enabled a suitable hands-free use, while also providing a structure to which a cloth could be draped over the monitor to allow shielding of sunlight and viewing of the monitor.

The camera was mounted to pole consisting of a 2 m length of 25 mm electrical conduit, in order to use the camera at distance from the LWD, without suspending sediment and increasing turbidity near pieces of LWD. The pole also served as a mount for an underwater light source, which was useful for peering into dark hollows of LWD pieces.

An underwater light source consisting of a 12 volt, 50 W halogen globe connected to a large deep cycle battery provided illumination. A VHS video recorder (TEAC, Japan), powered by a 150 W DC-AC inverter (Digitor, Dick Smith Electronics Australia) was also used to capture video of oocyte surveys. Connected to the battery was a small portable generator (Honda EX350, Honda, Japan), which periodically charged the battery to ensure enough current was supplied to the light source and video recorder.
Plate 3.1 Underwater camera equipment set up

Underwater camera survey of replaced LWD pieces

Two operators were required to undertake videography, one to operate the camera in the water and one to operate the electronics on shore (Plate 7.2). Setting up of equipment and conducting the video survey took approximately 1 hour to complete. Five pieces of LWD located in shallow water (< 1 m) were selected, primarily because light intensity was maximised.
Plate 3.2 Operation of the underwater camera equipment

Snorkelling

Snorkelling was conducted as an alternative to using an underwater camera, but it also served as a useful confirmatory tool for the discovery of any potential oocytes. Snorkelling was conducted once the underwater camera examined LWD pieces as bodily movements around pieces stirred up sediment and decreased water clarity. Each LWD piece was carefully examined for a period of at least 15 minutes.

Results and Discussion

Several problems were discovered with attempting to detect the presence of known oocytes, or positions on the logs, where oocytes are likely to be present. Consequently, this meant these methods and equipment employed were unreliable and thus abandoned without further testing. The underwater camera failed to find
the presence of placed oocytes on the outside of LWD pieces. It became apparent during trials and inspection of video afterwards, that discriminating the presence of oocytes from adjacent patches of algae (Plate 7.3). This is likely due to combination of factors including oocyte size, image and lense quality of the camera. A camera with a higher quality lens and greater focal range may have more clearly discriminated the presence of oocytes.

Plate 3.3 An example image of placed oocytes attached to the outside of a replaced LWD piece.

Snorkelling, although, being able to be employed to search most of the outside of the pieces effectively, did not reveal the presence of oocyte clusters on the outside of the five surveyed, replaced LWD pieces. Snorkelling also revealed difficulties in accessing locations. For example, it was near impossible to examine small gaps on
underside of large LWD pieces and especially hollows, which were often smaller than the size of a human head. Searching LWD pieces also proved challenging with snorkelling as large amounts of silt were disturbed upon movement around LWD pieces, immediately clouding the water and prohibiting visual searches of oocyte clusters.

Conclusion

Different methods attempting to reveal the presence of oocytes on replace LWD pieces were found to problematic in a number of ways. These problems require overcoming before these methods can be used reliably to detect the presence of oocytes within sites.

From the evidence of spawning on artificial PVC substrates within the same location as replaced LWD pieces, it appears internal characteristics were more important for oviposition and as oviposition never occurred on the outside of artificial PVC substrates, it appears small gaps are important, making both methods trialed here extremely difficult to detect the presence of oocytes. Further evaluation of equipment, particular, camera quality is likely to yield the necessary requirements to undertake oocyte surveys.