Journal of Sustainability Science and Management
Volume 8 *Number* 1, June 2013: 43-52

ISSN: 1823-8556

THE IMPACT OF MANAGEMENT PRACTICES OF EXOTIC WILLOWS (*SALIX* SPP.) ON AQUATIC INVERTEBRATE COMMUNITIES IN SOUTH AUSTRALIAN FRESHWATER STREAMS

WAHIZATUL AFZAN AZMI[1*] AND JOHN JENNINGS[2]

[1]*Department of Biological Sciences, Faculty of Science and Technology, Universiti Malaysia Terengganu, 21030 Kuala Terengganu, Terengganu, Malaysia.* [2]*School of Earth and Environmental Sciences, The University of Adelaide, South Australia 5005, Australia.*

**Corresponding author: wahizatul@umt.edu.my*

Abstract: The impact of willows, their removal and subsequent revegetation on aquatic invertebrate communities were examined in two freshwater streams in the Mount Lofty Ranges, South Australia. We hypothesized that lower abundance, species diversity and changes in functional feeding groups would occur where willows were present and have been removed. Unexpectedly, invertebrate abundance was significantly higher when willows were present in both streams. The introduced hydrobiid snail (*Potamopyrgus antipodarum*) was the most dominant taxon overall and was significantly more abundant under willows in both streams. More than half of total abundance under willows was contributed by scrapers (mostly *P. antipodarum*) as willow roots are presumed to provide a more stable habitat from high currents and have increased food availability compared with other vegetation. Where willows were removed and not revegetated, there were lower invertebrate species numbers and diversity in both streams. The removal of willows influenced not only loss of habitat, but also an increase in light intensity, decline in water quality and food availability. Our findings conclude that the presence of willows also reduces species numbers and diversity. Large scale willows removal may need special management considerations in order to reduce the impact on aquatic invertebrate communities.

KEYWORDS: Aquatic invertebrate communities, willows, abundance, revegetation

Introduction

Exotic willows (*Salix* spp.) are a naturalized component of the flora of Australia and have invaded thousands of kilometers of watercourses (Read and Barmuta, 1999). They were widely introduced by European settlers during the early 19th century mainly for erosion control, bank stabilization and navigation (Ladson *et al.,* 1997). Their high invasion rate has raised concerns because they are presumed to be responsible for a decline in native biodiversity and are generally seen as a serious threat to Australian freshwater ecosystems. Various ecologists have speculated that changes between native Australian vegetation and exotic willow species will affect channel morphology and erosion, water quality, primary and secondary production and aquatic fauna densities (Glova and Sagar, 1994; McKie and Cranston, 2001; Jayawardana *et al.,* 2006).

The impact of willows on aquatic invertebrate communities has been researched previously, but there are inconclusive and/or contradictory findings. In streams in Central Otago, New Zealand, Lester *et al.* (1994) found lower densities of benthic macroinvertebrates in reaches lined by *S. fragilis.* In the Murray River at Blanchetown, South Australia, Schulze and Walker (1997) found only minor differences in the composition of littoral macroinvertebrates between sites lined with willow (*S. babylonica*) and river red gum (*Eucalyptus camaldulensis*). They concluded that willows may influence stream invertebrates through altering food supplies (through decreasing sunlight penetration and fallen leaves during autumn) or habitat (by reducing the size of substrate particles and flow rates). Jayawardana *et al.* (2006) found no consistent differences in species richness and abundance in willows and *Phragmites australis* habitats in Birch Creek, Victoria. In contrast, some studies have suggested that willows may support high numbers of aquatic invertebrates by providing a food supply or habitat for some species to exploit. Glova and Sagar (1994) found greater species richness and diversity in benthic invertebrate communities in reaches lined by willows than in reaches with bare banks in three New Zealand streams. Interestingly, Yeates and Barmuta (1999) reported in forest streams in Hobart, Tasmania, that three macroinvertebrates (*Notolina* sp., Trichoptera: Leptoceridae; *Koornonga* sp., Ephemeroptera: Leptophlebiidae; *Physastra gibbosa,* Mollusca: Planorbidae) strongly preferred to feed on green willow leaves over senescent willow, green and senescent eucalypt leaves.

Most studies have only compared communities of aquatic invertebrates in willow and non-willow/ native vegetation sites or reaches with bare banks. No study has been carried out to investigate the community response to potential effects of the removal of willows and subsequent revegetation on aquatic invertebrates. Specifically, our research questions were: are there any difference in species richness and abundance when willows are present, removed or revegetated and how do they compare with original vegetation? Outcomes of these hypotheses would give an insight as to whether willows should be retained, removed or controlled, and whether revegetation is warranted.

Materials and Methods

Study Sites

Sixth Creek is a freshwater stream located at 34°52‶N and 138°45‶E (Figure 1). It flows north to meet the Torrens River at Castambul, NE of Adelaide in the Mount Lofty Ranges, South Australia. It is a fifth order stream with a total length of ~18 km and drains an area of ~46 km2. It ranges from 1−4 m in width, 0.5−4.0 m in depth and the water velocity is slow to fast. The upper reaches of the stream are heterogeneous with a coarse bed substrate, fine roots, leaf packs, woody debris, and overhanging vegetation. Some reaches are totally exposed to sunlight with fast flowing water (mainly in the winter and spring), while others are canopied with large trees and shrubs and have slow moving water. The middle and lower parts of the stream are relatively deep (2.5−4.0 m) and mostly covered by forest canopy, with grasses and shrubs along the stream edge.

Deep Creek is also located in the Mount Lofty Ranges NE of Adelaide at 34°56‶N and 138°46‶E (Figure 1). This small fourth order stream is approximately 8.5 km long, is 2.0−4.5 m in width, 0.5−2.5 m in depth, is slow to sometimes fast flowing and the catchment area is ~13 km2. The stream bed is almost debris free, mainly comprising sand, pebbles, cobbles and small boulders. Generally, the upper reaches of the stream are exposed to direct sunlight where the water is deep and slow moving and the bed mostly sandy. The lower reaches are partly cleared and dominated by willows. The climate in the Mount Lofty Ranges catchment area is hot Mediterranean. The average annual rainfall in the catchment areas is approximately 900 mm, with average maxima of 25^{0}C (soaring into the 40s) during dry summers and with maximum temperatures averaging 12^0 to 15^{0}C during cool, wet winters. Both study catchments include a mixture of rural residential, horticulture, orchards and grazing land.

Each study stream had four different treatments:

(1) willows present (WP) – ~60% of the riparian area dominated by willows and ~40% covered by ash, herbs and shrubs at both Sixth Creek (WP1) and Deep Creek (WP2).

(2) willows removed (WR) – banks more or less bare, except for grasses at Sixth Creek (WR1); at Deep Creek, much of the ground was bare and ~20% were covered by grasses and small shrubs (WR2).

(3) revegetation (RV) – at Sixth Creek (RV1), willows were removed in 1997 and revegetated mainly with *Juncus* spp. (rush), *Carex* spp. (tussock sedge), *Rorrippa* spp.

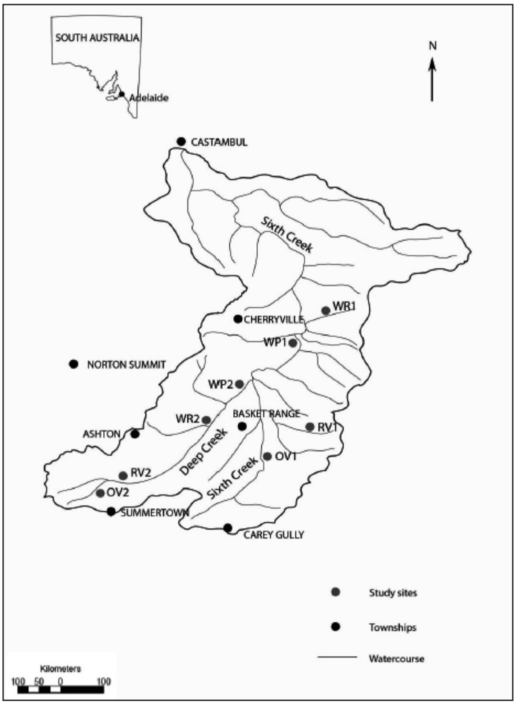

Figure 1: Location of study sites in the Mount Lofty Ranges catchment area, South Australia (Notes: WP = Willows present; WR = Willows removed; RV = Revegetation; OV = Original vegetation; 1 = Sixth Creek; 2 = Deep Creek).

(watercress) and *E. viminalis* (white gum or manna gum); at Deep Creek (RV2), willows were removed in 2004 and revegetated mainly with *Acacia melanoxylon*, *A. retinoides,* and *E. viminalis*.

(4) original vegetation (OV) – ~50% of the canopy comprising mainly *E. viminalis, E. obliqua* (messmate stringybark)*, A. retinoides*, *A. melanoxylon* and ~50% various shrubs and herbs, many of which are introduced weeds (periwinkle, soursob, blackberries and bamboo) at Sixth Creek (OV1); at Deep Creek (OV2), ~60% were dominated by native species included *E. viminalis, E. obliqua, A. melanoxylon, Juncus* spp., *Carex* spp. and the rest are much denser, species-rich ground carpet of exotic grasses and small shrubs.

Sampling Methodology

The sampling of the aquatic invertebrate communities was carried out every two months from September 2007 to November 2008 to examine any difference in diversity and patterns of seasonal abundance at each site and across treatments. The sampling for each treatment covered ~100 m^2 in area and within each treatment, three different habitats (pools, riffles and edges) were surveyed. Each habitat was replicated 10 times to obtain reasonable estimates of population density. Aquatic invertebrates were collected using an aquatic net with a 40 x 40 cm frame, 60 cm long net of 250 mm mesh. Big stones in swift water were hand-lifted and checked for invertebrates. The invertebrates and the content of each sample (net) was transferred into properly labeled plastic containers, preserved in 80% ethanol and taken back to the laboratory for analysis. Samples were washed in white trays and screened through 1 mm and 250 mm sieves. The invertebrates were initially sorted into functional feeding groups and then identified as far as possible (Merritt and Cummins, 1996; Wats, 1998; Gooderham and Tsyrlin, 2002; Dean *et al.,* 2004; Theischinger and Hawking, 2006).

Data Analysis

Two-way ANOVAs were used to evaluate differences in invertebrate distribution at different sites, treatments, seasons and habitats. The data were log (x+1) transformed to ensure normality in calculations of means and ANOVAs. Where there were significant differences, posterior pairwise comparisons (t-test) were employed to describe which means were most alike (or different) and to test the equality of means for each pair of variables. We used the same model of two-way ANOVAs to estimate differences in the abundance of functional feeding groups among sites, treatments, seasons and habitats. These analyses were performed using the Prism 5.00 statistical program (PRISM, 2007). The degree of species composition or species diversity (H') for each site was determined by using the Shannon Wiener Index. The higher value of

H', the greater the diversity and supposedly the cleaner the environment (Ludwig and Reynolds, 1988; Metcalfe, 1989). Richness Index that has been used was Margalef's Index (R). This index indicates the number of species in a sample or the abundance of the species per unit area (Ludwig and Reynolds, 1988; Metcalfe, 1989). Homogeneity or pattern of distribution of species in relation to other species in a sampled per unit area was calculated using Evenness Index (E) (Ludwig and Reynolds, 1988; Metcalfe, 1989). Principal component analysis (PCA) was used to determine differences among species assemblages and to illustrate the pattern of variation for the most highly species associated ($P < 0.05$) with the effects of treatments and habitats using the statistical program of PC-ORD version 5.13 developed by McCune and Mefford (2006).

Results and Discussion

A total of 76,683 individuals from 51 taxa in 42 families of aquatic invertebrates were collected from Sixth and Deep Creeks from September 2007 to November 2008. Significantly more aquatic invertebrates were collected in Sixth Creek (40,864 individuals) compared with Deep Creek (35,819 individuals) ($P < 0.001$) (Table 1). The effect of treatments on invertebrate total abundance varied significantly among habitats in each site and during different seasons. The total abundance was significantly greater in sites with willows present than in other treatments for both streams (29,741 individuals). Where willows were removed, slightly lower numbers of aquatic invertebrates (13,050 individuals) were recorded. In revegetation and original vegetation sites, fewer numbers of aquatic invertebrates were found (15,753 individuals and 18,139 individuals). Further analysis using the *t*-test showed that the total number of individuals in the willows present treatment in Sixth Creek was significantly greater ($P < 0.001$) compared with other treatments for both streams. However, no significant differences in total abundance were detected for other treatments. In terms of taxon richness, in both streams there was a slightly more diverse and richer fauna was recorded in original vegetation (51 taxa) compared with both revegetation (49 taxa) and sites with willows removed (47 taxa), but was lower in sites with willows present (39 taxa) (Figure 2).

Snails (Gastropoda) were the most dominant invertebrate, comprising almost half of the total abundance at each site (48.1%). They were numerous in all treatments in both streams. Plecoptera were the second most abundant invertebrate comprising 18.2% in Sixth Creek and 17.3% in Deep Creek, and Diptera were the third highest in both streams, although more were found in Deep Creek (14.6%) compared with Sixth Creek (10.6%). Similar trends in total abundance were observed in both streams for Trichoptera, Ephemeroptera and Coleoptera. However, fewer Odonata and Hemiptera were recorded in Sixth Creek than Deep Creek, but slightly more bivalves were found in Sixth Creek (1.2%) than in Deep Creek (0.8%). A number of minor groups, Oligochaeta, Amphipoda, Arachnida, Ostracoda, Hirudinea and Decapoda, were represented by less than 1.0% of the total abundance at each site. Collembola and Isopoda were only found in Sixth Creek.

The PCA case score plots indicate that the invertebrate community assemblages were divided into three main groups (Figure 1). Revegetation (RV1 and RV2) and original vegetation (OV1 and OV2) samples from both sites were grouped together as Group 1. Group 1 was characterized by high relative abundances of physid snails (*Physa acuta*), planorbiid snails (*Isidorella* sp., *Glytophysa* sp.), lymnaied snails and gripopterygid stonefly nymphs (*Dinotoperla evansi*). Group 2 comprised mainly samples of willows present (WP1 and WP2) and willows removed (WR1 and WR2) from both sites. The most abundant species found in these treatments were hydrobiid snail (*Potamopyrgus antipodarum*), leptophlebiid mayfly nymphs (*Illiesoperla mayii*), caenid mayfly nymphs (*Tasmanocoenis tillyardi*), and coenagrionid damselfly nymphs. Interestingly, riffle habitat samples WP1, WP2 and WR2 were clearly separated from the other groups (Group 3). This

Table 1: Results of two-way ANOVAs on total abundance and species richness of aquatic invertebrates collected from Sixth and Deep Creeks (d.f. = degree of freedom; MS = mean squares; *** $P < 0.0001$; ** $P < 0.001$; *$P < 0.01$).

Source	d.f.	Species richness			Total abundance		
		MS	*F*	*P*	MS	*F*	*P*
Treatment (Tre)	3	19.53	2.551	0.092	6850.00	1998.000	<0.0001***
Site (Si)	1	6.25	0.024	0.880	609.50	177.800	<0.0001***
Season (Se)	3	20.61	2.693	0.081	6583.00	110.900	<0.0001***
Habitat (Ha)	2	1074.00	7.388	0.008*	644.90	19.500	0.0002 **
Tre x Si	3	40.74	0.108	0.953	223.80	65.260	<0.0001***
Tre x Se	9	9.75	1.274	0.322	4134.00	0.491	0.8600 **
Tre x Ha	6	37.80	0.125	0.991	744.70	22.520	<0.0001***
Se x Si	3	1840.00	7.179	0.012*	1239.00	20.880	0.0004 **
Se x Ha	6	386.60	2.658	0.071	1685.00	81.970	<0.0001***

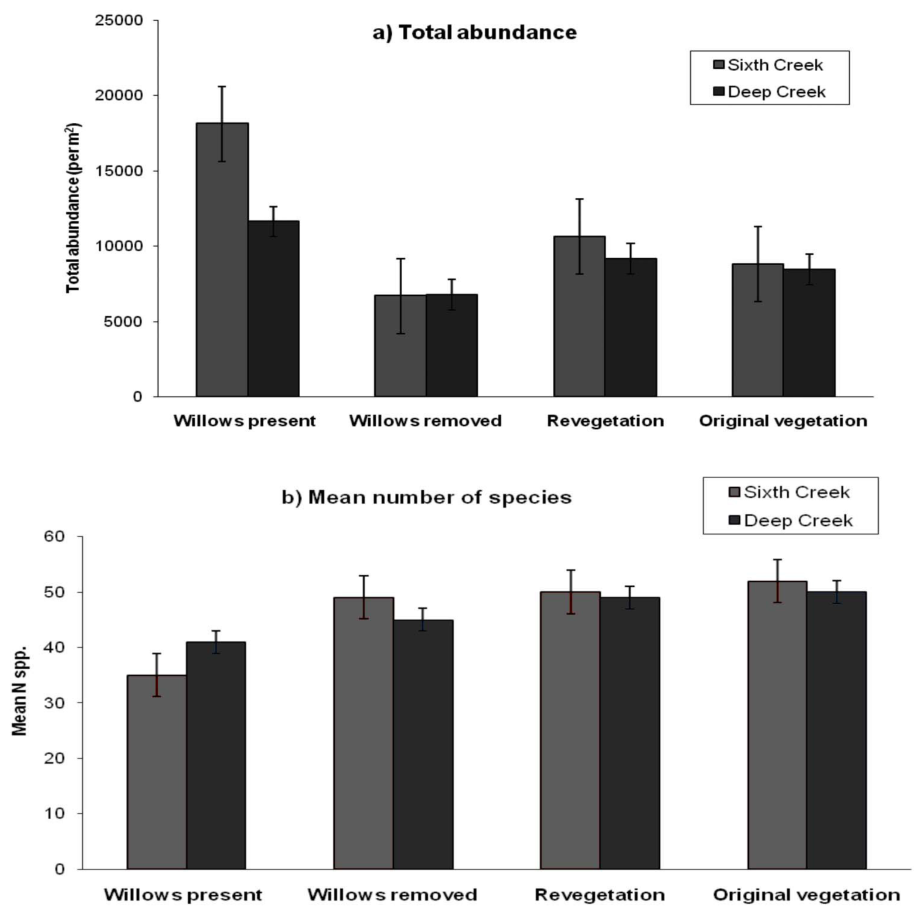

Figure 1: Total abundance (a) and mean number of species (b) of aquatic invertebrate communities in four different treatments in Sixth and Deep Creeks.

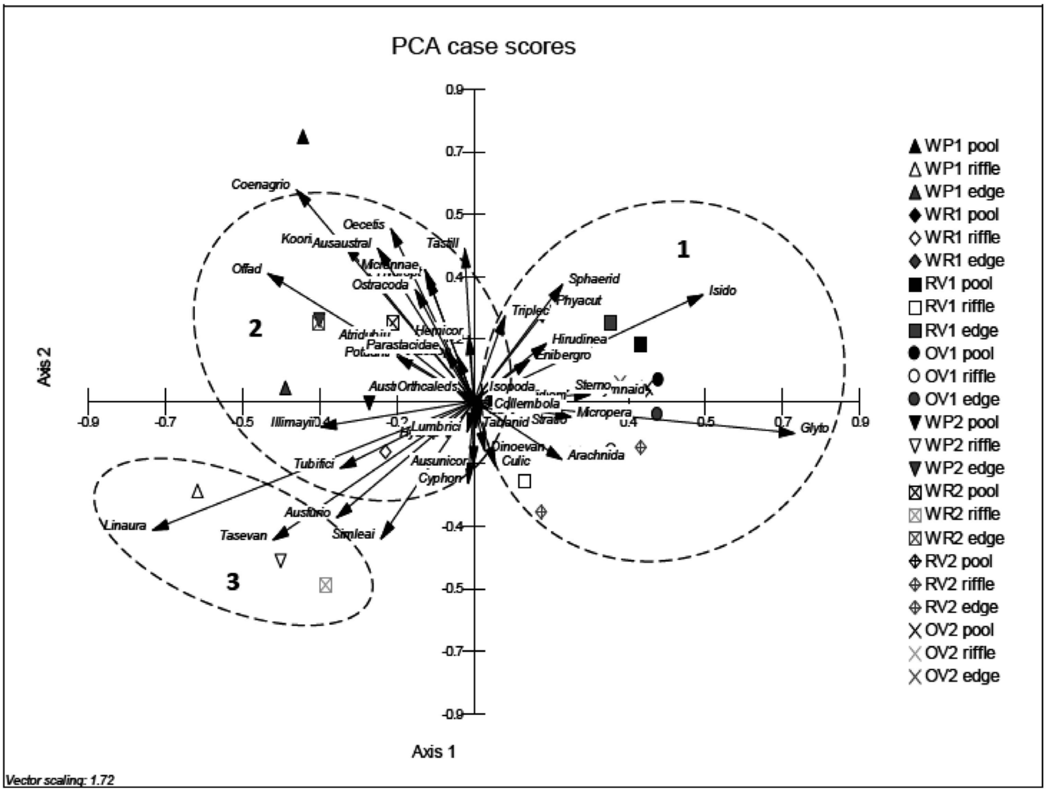

Figure 2: The principal component analysis (PCA) case scores showing species assemblages associated with treatments in each habitat. (Note: the three dotted circles represent three main groups which indicate the degree of association between invertebrates, treatments and habitats).

group was characterized by high abundance of elmid beetle larvae (*Simsonia leai*), conoesucid caddisfly nymphs (*Lingora aurata*) and hydrobiosid caddisfly nymphs (*Taschorema evansi*).

The introduced New Zealand hydrobiid snail (*P. antipodarum*) was the most dominant taxon overall and was significantly more abundant under willows in both streams (26,433 individuals). Other commonly collected taxa included the introduced physid snails (*Physa acuta*), gripopterygid stonefly larvae (*D. evansi*), midge larvae (Chironomidae), mayfly larvae (*T. tillyardi*) and water bugs (*Enithares bergrothi*) which were mostly abundant under willows treatments in both streams, but this pattern was not statistically significant. Planorbiid snails (*Isidorella* sp.), baetid mayfly larvae (*Offadens* sp.), elmid beetle larvae (*Simsonia leai*), hydrobiosid caddisfly larvae (*Taschorema evansi*) and damselfly nymphs (Coenagrionidae) were significantly more abundant under willows in Sixth Creek. The abundance of blackfly larvae (*Austrosimilium furiosum*) and amphipods (*Austrochiltonia australis*) was significantly greater when willows were present in Deep Creek. The freshwater snails *Glyptophysa* sp. and Lymnaiedae, and *Koornonga insconspicua* (Leptophlebiidae) were significantly more abundant in sites lacking willows.

Calculated biological indices indicated that the richness, of the species varied slightly among the four treatments. The highest diversity was recorded in original vegetation (H' = 2.80) followed by revegetation (H'= 2.78) and willows present (H' = 2.64). Willows removed sites were the poorest among these three sites (H' = 1.86). Original vegetation sites demonstrated the highest value of richness index (R) which was 5.97 followed by revegetation (R = 4.59), willows present (R = 4.14) and willows removed (R = 4.05). The evenness index was higher in willows present sites compared with other sites.

Overall, our results indicated that the presence of willows along the riparian zones of Sixth and Deep Creeks had a negative impact on the taxon diversity of the aquatic invertebrate communities. Initially, we hypothesized that lower abundance, taxon diversity and fewer functional feeding groups would occur in those treatments where willows were present and when they had been removed. Unexpectedly, invertebrates were significantly more abundant in both sites where willows were present, whereas, lower abundance and taxon diversity were recorded in the willows removed treatment at both sites. More invertebrates were collected in Sixth Creek than in Deep Creek. This may be due to water availability as the former creek is permanent and flows all year round, whereas the latter is more ephemeral and dries up in some years. Water balance, temperature tolerances, respiratory system and mode of movement of aquatic invertebrates are all adaptations for living underwater (Merritt and Cummins, 1996). When Deep Creek dries up, the aquatic invertebrate communities face the challenge to survive and this would presumably contribute to lower taxon abundance and richness in Deep Creek.

However, taxon richness of the aquatic invertebrate communities in Sixth and Deep Creeks were similar and no consistent differences were detected. Interestingly, treatments had a significant effect on the total abundance, taxon richness and invertebrate community assemblages across different habitats, seasons and sites. This suggests that there may be some physical or chemical mechanism(s) that might explain these differences in invertebrate assemblages in the different treatments. We suspected that willows may provide more habitat heterogeneity, especially for the scrapers (mostly gastropods), than other treatments. The distribution and abundance of aquatic invertebrates has long been known to be associated with the heterogeneity of habitats (McKie and Cranston, 2001; Rios and Bailey, 2006; Walsh *et al.,* 2007). High heterogeneity refers to physically complex substrate types which consist of leaves, gravel or cobble, macrophytes, moss and wood; which generally support more taxa than structurally simple substrates such as sand and bedrock.

We found the introduced hydrobiid snails were the most dominant taxon with about 50% of the total abundance under willows in both streams. Some New Zealand studies have shown that willows may have positive effects on invertebrates by providing a stable habitat among tree roots for large numbers of *P. antipodarum* (Winterbourn, 1970) or trichopteran shredders (*Pycnocentria forcipata*) (Linklater and Winterbourn, 1997). However, *P. antipodarum* has become a pest species in many parts of the world and has rapidly invaded Australian freshwater systems. Several authors speculated that the establishment of this species may disrupt the physical characteristics of invaded ecosystems (Zaranko *et al.,* 1997). A study in the western United States reported that the invasion of this snail may impact the food chain of native trout and concluded that *P. antipodarum* is a potential competitor with native aquatic species of the streams (Richards *et al.,* 2004). However, to date, little research has documented decreases in native invertebrate populations in Australian streams where *P. antipodarum* has invaded. Thus, this study is important to understand the possible indirect impact of willow presence on the reduction of native invertebrates mediated by a strong competition for habitat or food between *P. antipodarum* with native invertebrates.

In terms of taxon richness, a reduction in taxon number and diversity in both streams was clearly associated with the presence of willows. A number of factors which result in a lower diversity of aquatic invertebrates being supported by willows than native vegetation, include decreased primary production due to shading, increased sedimentation, reduced water flow, and leaching of inhibitory chemicals from leaves, barks or roots. Detritus from willow leaves may also contribute to the lower densities, as willow detritus only provide a food source for a relatively short period of time and may not be fully utilized by aquatic invertebrates (Lester *et al.,* 1994). Jayawardana and Westbrooke (2010) found greater abundance of shredders in willow habitats in three Victorian streams. They suggested that the vegetation changes can bring about changes in material cycle and energy flow within these streams.

Higher invertebrate diversity was observed in both original vegetation and revegetation sites in both streams. Taxon diversity and richness in these sites were the highest compared with other treatments, indicating that original vegetation and revegetation treatments provide suitable habitat for a diverse community of aquatic invertebrates. The richer fauna in the original vegetation and revegetation sites is probably due to a sparse, open canopy which permits higher primary productivity and favours a more diverse growth of aquatic plants. Some native snails (*Glyptophysa* sp. and Lymnaiedae) were significantly more abundant in sites lacking willows. River red gum (*E. camaldulensis*) and white gum (*E. viminalis*) dominate these sites, and the presence of these snails may be due to the presence of diatoms and microflora on decaying leaves of these eucalypts as found by Schulze and Walker (1997). Most predator taxa (e.g. dragonfly nymphs) had higher densities in these treatments. More diverse growth of riparian/aquatic vegetation and more complex substrate types (gravel/cobbles, macrophytes) are thought to provide additional sources of food and shelter in original vegetation and revegetation treatments. Increases in habitat heterogeneity increase taxon richness and abundance of invertebrates and may provide suitable habitat for reproduction, protection from predators, and food supply for many taxa. Furthermore, leaves of native species (especially *Eucalyptus*) are often only consumed by invertebrates after a period of conditioning in the stream.

In contrast, slightly lower numbers of invertebrates were recorded in sites where willows had been removed in both streams. The sudden removal of willows without subsequent revegetation and little plant growth, apart from a few weeds, led to a reduction in the abundance and diversity of aquatic invertebrate communities, and may be due to loss of habitat and also to decreasing substrate size. Degradation of the substrate can be devastating to invertebrate communities, as decreased substrate size is generally associated with a decrease in invertebrate abundance (Jayawardana *et al.,* 2006). Each habitat sampled in willows removed treatments included areas of slow moving water with patches of fine sand and silt. Most pollution tolerant invertebrates such as oligochaetes, snails and midge larvae, can survive in these areas. The high abundance of these taxa indicates strongly that they are tolerant of or have adapted to live in stressed environments. This findings support that removal of willows without revegetation has had a detrimental impact on taxon diversity. This could also result in accelerating bank erosion allowing more organic pollution-tolerant taxa to be established.

Even though, slightly lower numbers of invertebrates were observed in sites where willows were removed, the number of species was higher than sites in willows present (Figure 2). Changing in channel morphology and habitat preferences are some factors that could probably influenced the aquatic fauna species richness in willows sites (Glova and Sagar 1994; Jayawardana *et al.,* 2006). Those factors probably have a direct link with food supply in the aquatic food chain that leads to numerical increases in some prey taxa such as chironomid, Ephemeroptera and mosquito larvae specifically in willows removed site. Therefore, it shows that certain invertebrates prefer open riffle and edge habitats under willows removed site as their nymphs require high oxygen levels, which often associated with fast running water (Merritt and Cummins, 1996). Besides, higher light intensity can increase primary production of algal growth which in turn may increase secondary production and change invertebrate communities (Lester *et al.,* 1994; Schulze and Walker, 1997). In contrast, under willows treatment, lower numbers of species were found because most of the species show a preference for still and slow moving water. Dense shade does not allow the light required for the production of high quality food for invertebrates (Bunn *et al.,* 1999) and probably due to that reason, the number of species was lower compared with willows removed sites.

Conclusion

In conclusion, willows may support high numbers of aquatic invertebrates by providing a more stable and complex habitat and also providing a suitable food source (e.g., by producing greater surface for colonisation by biofilms on willow leaves and roots). However, removal of willows without any subsequent revegetation resulted in lower species richness and abundance of aquatic invertebrates, suggesting that taxon richness and abundance will only recover when the riparian canopy is reinstated by suitable revegetation efforts. This present study has provided information on the changes in aquatic invertebrate communities that might take place when the original riparian vegetation changes to willow; and when willows were removed without revegetation. It is also inferred that large scale willows removal may require special management considerations particularly in small and shallow streams to reduce the impact on aquatic invertebrates. These would include of site specific willow management strategies which consider variety of factors such as the ecological characteristics of the water body (e.g., invertebrates and fish assemblages, primary productivity, etc.) and the physico-chemical conditions of the water body (e.g., bank stability, water temperature, flows, depth and size, etc.).

Acknowledgements

We thank Jose Facelli (University of Adelaide) for statistical advice, Chris Madden (Freshwater Macroinvertebrates) for assistance during field work, species identification and valuable comments on the manuscript, Peter Goonan (Environment Protection Authority) for valuable advice and critiques on the manuscript, Katrina Warner (Sixth Creek Catchment Group) for helping with sites and field work, the Government of Malaysia for a scholarship to Wahizatul Afzan Azmi and the Nature Conservation Society of SA Inc. for a Conservation Biology Grant.

References

Bunn, S.E., Davies, P.M. and Mosisch, T.D. (1999). Ecosystem Measures of River Health and their Response to Riparian and Catchment Clearing. *Freshwater Biology,* 41: 333-346.

Dean, J.C., St Clair, R.M. and Cartwright, D.I. (2004). *Identification Keys to Australian Families and Genera of Caddisfly Larvae (Trichoptera).* Thurgoona, New South Wales: Cooperative Research Centre for Freshwater Ecology.

Glova, G.J. and Sagar, P.M. (1994). Comparison of Fish and Macroinvertebrate Standing Stocks in Relation to Riparian Willows (Salix spp.) in Three New-Zealand Streams. *New Zealand Journal of Marine and Freshwater Research*, 28: 255-266.

Gooderham, J. and Tsyrlin, E. (2002). *The Waterbug Book: A Guide to the Freshwater Macroinvertebrates of Temperate Australia.* Collingwood, Victoria: CSIRO Publishing.

Jayawardana, J.M.C.K. and Westbrooke, M. (2010). Potential Effects of Riparian Vegetation Changes on Functional Organisation of Macroinvertebrates in Central Victorian Streams. *Victorian Naturalist,* 127(2): 36-48.

Jayawardana, J.M.C.K., Westbrooke, M., Wilson, M. and Hurst, C. (2006). Macroinvertebrate Communities in Willow (Salix spp.) and Reed Beds (Phragmites australis) in Central Victorian Streams in Australia. *Marine and Freshwater Research*, 57: 429-439.

Ladson, A.R., Gerrish, G., Carr, G. and Thexton, E. (1997). *Willows along Victorian Waterways*. Melbourne: Department of Natural Resources and Environment.

Lester, P.J., Mitchell, S.F. and Scott, D. (1994). Effects of Riparian Willow Trees (Salix fragilis) on Macroinvertebrate Densities in Two Small Central Otago, New Zealand Streams. *New Zealand Journal of Marine and Freshwater Research,* 28: 267-276.

Linklater, W. and Winterbourn, M.J. (1993). Life Histories and Production of Two Trichopteran Shredders in New Zealand streams with Different Riparian Vegetation. *New Zealand Journal of Marine and Freshwater Research,* 27: 61-70.

Ludwig, J.A. and Reynolds, J.F. (1988). *Statistical Ecology: A Primer on Methods and Computing*. New York: John Willey and Sons.

McCune, B. and Mefford, M.J. (2006). PC-ORD. Multivariate Analysis of Ecological Data. Version 5.13. MjM Software, Gleneden Beach, Oregon, USA.

McKie, B. and Cranston, P.S. (2001). Colonisation of Experimentally Immersed Wood in South Eastern Australia: Responses of Feeding Groups to Changes in Riparian Vegetation, *Hydrobiologia*, 452: 1-14.

Merritt, R.W. and Cummins, K.W. (1996). *An Introduction to the Aquatic Insects of North America. (3rd ed.*) Dubuque, Iowa: Kendall-Hunt Publishing Company.

Metcalfe, J.L. (1989). Biological Water Quality Assessment of Running Waters Based Onmicroinvertebrates Communities: History and Present Status in Europe. *Environmental Pollution,* 60: 101-139.

PRISM. (2007). Graphpad Prism 5.00. Graphpad Software Inc. San Diego, California USA, 2007.

Read, M. and Barmuta, L. (1999). Comparisons of Benthic Communities Adjacent to Riparian Native Eucalypt and Introduced Willow Vegetation. *Freshwater Biology*. 42: 359-374.

Richards, D.C., O"Connell, P. and Shinn, D.C. (2004). Simple Control Method to Limit the Spread of the New Zealand Mudsnail *Potamopyrgus antipodarum*. *North American Journal of Fisheries Management,* 24: 114-117.

Rios, S.L., and Bailey, R.C. (2006). Relationship between Riparian Vegetation and Stream Benthic Communities at Three Spatial Scales. *Hydrobiologia,* 553: 153-160.

Schulze, D.J. and Walker, K.F. (1997). Riparian Eucalypts and Willows and their Significance for Aquatic Invertebrates in the River Murray, South Australia. *Regulated Rivers Research & Management,* 13: 557-577.

Theischinger, G. and Hawking, J. (2006). *The Complete Field Guide to Dragonflies of Australia*. Collingwood, Victoria: CSIRO Publishing.

Walsh, C.J. Waller, K.A., Gehling, J. and Nally, R.M. (2007). Riverine Invertebrate Assemblages are Degraded More by Catchment Urbanisation than by Riparian Deforestation. *Freshwater Biology*, 52(3): 574-587.

Watts, C. (1998). *Preliminary Guide to the Identification of Adult and Larval Dytiscidae and Adult Aquatic Hydrophilidae (Insecta: Coleoptera)*. Thurgoona, New South Wales: Cooperative Research Centre for Freshwater Ecology.

Winterbourn, J. (1970). Population Studies on the New Zealand Freshwater Gastropod, Potamopyrgus antipodarum (Gray). *Journal of Molluscan Study*, 39: 139-149.

Yeates, L.V. and Barmuta, L.A. (1999). The Effects of Willow and Eucalypt Leaves on Feeding Preference and Growth of Some Australian Aquatic Macroinvertebrates. *Australian Journal of Ecology,* 24: 593-598.

Zaranko, D.T., Farara, D.G. and Thompson, F.G. (1997). Another Exotic Mollusc in the Laurentian Great Lakes: The New Zealand native *Potamopyrgus antipodarium* (Gray 1843) (Gastropoda, Hydrobiidae). *Canadian Journal of Fisheries and Aquatic Sciences,* 54: 809-814.