

**MEASURING AND MANAGING THE IMPACT OF SECONDARY
SALINISATION ON RIPARIAN FLORA**



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Cover photographs: two of the riparian plant species which are adversely affected by secondary salinisation in catchments of the Collie River in Western Australia; *Bossiaea eriocarpa* (left) and *Hypocalymma angustifolium* (right).

Abstract

Although secondary salinisation of streams has long been recognised as a major threat to biodiversity in Australia, there is little empirical evidence of the impacts of salinisation on riparian flora and fauna. We studied the effects of soil salinity on riparian plants and plant-dwelling invertebrates in two adjacent catchments of the Collie River in the south west of Western Australia. We found significant negative relationships between soil salinity and the diversity of riparian plants, both within and between catchments. Increasing salinity was also associated with a change in plant species composition, which appeared to be due to a greater impact of salinity on some riparian plant species than on others. In controlled laboratory experiments, increasing salinity significantly decreased germination rates in *Bossiaea eriocarpa*, *Hypocalymma angustifolium*, *Hakea lissocarpha*, *Hakea undulata* and *Hakea varia*, all of which were relatively more common in the field at low soil salinities, while there was no effect of salinity on germination rates in *Melaleuca viminea*, *Xanthorrhoea preissii* and *Corymbia calophylla*, which were all relatively more common at high soil salinities. We also found significant negative relationships between soil salinity and the diversity of plant-dwelling invertebrates within and between catchments, and a change in invertebrate species composition as salinity increased. This appeared to be due to a decrease in the number of species of predatory spiders at higher salinities. We hypothesise that increasing salinity reduced the diversity of riparian plant species, which in turn reduced the habitat complexity for spiders and led to a decrease in the diversity of spider species. This may have important consequences for the functioning of salinised riparian ecosystems.

Introduction

Salt is a natural component of landscapes throughout the world, and occurs at high levels in subsoils throughout most of Australia (Nielson *et al.* 2003; Pannell and Ewing 2006). Anthropogenic impacts, such as inadequate drainage from irrigation systems and replacing deep rooted perennial native vegetation with shallow rooted annual crops and pastures, has led to rising water tables, dissolving stored salt and increasing groundwater salinity (secondary salinisation). Secondary salinisation is a widely acknowledged environmental issue, affecting approximately 80 million hectares of land in the arid and semi arid regions of the world (Abrol *et al.* 1988; Dregne *et al.* 1991; Ghassemi *et al.* 1995; Beresford *et al.* 2001) and approximately 5.7 million hectares of agricultural land in Australia (National Land and Water Resources Audit 2000).

Dryland salinisation, caused by clearing native vegetation, is recognised as a pressing environmental issue in Australia and particularly in Western Australia (Cramer and Hobbs 2002; Hart *et al.* 2003). Although the impacts of dryland salinity on agriculture and rural communities in Australia have been reported extensively (Conacher and Murray 1973; Stirzaker *et al.* 2000; Doupé *et al.* 2003; Pannell and Ewing 2005), the impacts on biodiversity have only recently been considered (Hancock *et al.* 1996; Keighery 2000; Blacklow *et al.* 2003).

Low lying areas within the landscape, such as valley floors and aquatic systems (rivers, streams and wetlands) are usually the first to show the effects of secondary salinisation (Cramer and Hobbs 2002; Davis *et al.* 2003; Hart *et al.* 2003). Riparian zones are important components of these systems because they mediate ecological communication between terrestrial and aquatic habitats by maintaining bank stability, intercepting overland flow, trapping sediment and absorbing nutrients, as well as providing shade and exporting energy to

fuel aquatic food webs (Hancock *et al.* 1996; Naiman and Decamps 1997; Brock *et al.* 2005; Naiman *et al.* 2005). Riparian zones of south-west Western Australia are also home to a distinctive and diverse flora that supports an equally distinctive and diverse invertebrate community (Hancock *et al.* 1996; Myers 1990).

Unfortunately, we know very little about how riparian communities will be affected by secondary salinisation. Laboratory studies of salt sensitivity suggest that many species of riparian plants may be adversely impacted by stream salinities greater than 2000 mg L⁻¹ (Hart *et al.* 2003), but there has been little confirmation of such effects from field studies. The current lack of detailed biological information on riparian ecosystems in Australia means that baseline data on community structure are rarely available for sites already affected by secondary salinisation (Hart *et al.* 1990; Lymbery *et al.* 2003) and comparisons often involve salinised and non-salinised sites in very different geographic localities (e.g. Keighery *et al.* 2001).

Lymbery *et al.* (2003) and Doupé *et al.* (2006) compared the effects of stream salinisation on riparian vegetation in experimental catchments in the Collie River system in the south west of Western Australia. These catchments are geographically adjacent, and geologically and botanically similar, but differ in the extent to which they have been cleared and therefore in the salinity of their streams. Lymbery *et al.* (2003) found an increase in soil salinity, a decrease in plant species diversity and an altered plant species composition in a cleared catchment (downstream of the clearing treatment) compared to an uncleared catchment. Doupé *et al.* (2006) sampled along a salinity gradient within the cleared catchment and found a strong negative relationship between riparian plant species diversity and soil salinity, as well as a change in species composition with salinity. The inference from both of these studies was that clearing has led to a redistribution of salt via groundwater, increasing the salt load in both the stream and the adjacent riparian soil profile. Increased soil salinity in turn led to reduced species diversity and a change in species composition of the riparian vegetation.

The observational nature of these two previous studies, however, leaves doubt over a causative role for salinity; other environmental variables, correlated with salinity, may be responsible for changes in plant species diversity and composition in these catchments. In addition, we do not know the extent to which changes in plant species diversity and composition affect other aspects of ecosystem structure or function in the catchments. Such knowledge is essential if we are to properly preserve, repair, and manage the integrity of riparian ecosystems threatened by secondary salinisation (Hancock *et al.* 1996; Ridely and Pannel 2005).

In this paper we report the results of a new study of the riparian ecosystems in the catchments surveyed by Lymbery *et al.* (2003) and Doupé *et al.* (2006). We examine associations between soil salinity and the diversity and composition of both plant and associated invertebrate communities. We also test the effect of salinity on plant germination in a controlled experimental setting. We hypothesise that: (1) increased soil salinity will be associated with reduced riparian plant species diversity and different species composition within and between catchments, as previously found; (2) increased soil salinity will also be associated with reduced invertebrate species diversity and different species composition within and between catchments; and (3) riparian plant species with different relative abundances at high and low salinities will have different germination and growth rates in response to different salinity treatments.

Materials and Methods

Field studies

Study area

Lemon and Ernies catchments are situated within a 1000 ha area of state forest, in the 750 mm annual rainfall zone, 30 km east of the town of Collie in the south west of Western Australia. Both catchments have sand/sandy clay/loam and gravel soils overlying Archaean granite and are open forest areas of *Eucalyptus marginata*, *Corymbia calophylla*, *Eucalyptus wandoo* and *Xanthorrea pressii*. In 1976, 186 ha (54%) of Lemon catchment was cleared from ridgeline to streamline; Ernies catchment was left as an uncleared control. From 1976 to 1997, annual stream flow increased from ~10 mm to 1000 mm and stream salinity increased from ~100 mg l⁻¹ to 1600 mg l⁻¹ in Lemon catchment. Stream salinity decreases sharply as water flows from the cleared land through 7 km of forest to the Collie River. Annual stream flow and stream salinity remained at baseline levels in Ernies catchment (Ruprecht and Schofield 1991a,b; Lymbery *et al.* 2003).

Sampling

Sampling took place in spring (September to November) in 2006 and 2007. Ten quadrats, each 4 m² in area, were established adjacent to the stream over 1 km in Ernies catchment, and 20 quadrats were established over 6 km in Lemon catchment, starting approximately 1 km downstream of the clearing treatment.

Within each quadrat, apparent soil conductivity was measured in the 0-1.5 m range using a Geonics EM-38 electromagnetic induction meter, calibrated over a range of soil textures found within the area. Apparent conductivity (EC_a) readings are reported as EC_{1.5} values (in mS m⁻¹), by using the calibration formula $EC_{1.5} = 0.42(EC_a) - 1.74$ (Bennett and George 1995). All plant species rooted within the quadrat were identified and using a grid system, percentage cover was estimated. Plant species were further categorised into one of six different functional groups based on their life form: annual grass, annual herb, perennial grass, perennial herb, shrub and tree. In each quadrat, two 30 second vegetation sweeps were taken (one in the morning and one in the afternoon) using a sweep net 380 mm in diameter, with a mesh size of 1.4 mm². A pilot study found that the number of invertebrate species captured by sweeping was not significantly different between sweeps of 30 seconds, 60 seconds and 120 seconds duration (data not shown). Captured invertebrates were killed in ethyl acetate and preserved in 70% ethanol/5% glycerol. Samples from morning and afternoon sweeps from each quadrat were combined, then sorted to order and morphospecies, and all individuals in each morphospecies were counted. Morphospecies were further classified into functional groups based on dominant feeding habit, following Koricheva *et al.* (2000).

Analysis

We compared estimates of both plant and invertebrate diversity and plant and invertebrate composition between Ernies catchment and the first 3 km (i.e. the most saline end) of Lemon catchment (see Results), and also along the 6 km salinity gradient at Lemon catchment.

Plant and invertebrate species diversity for each quadrat was estimated by the number of species (S), Margalef's species richness index (d, the number of species corrected for differences in abundance; Margalef 1958) and the Shannon-Wiener index (H; Shannon 1948). Functional group diversity for each quadrat was estimated by the number of species in each life form category (for plants) or feeding habit category (for invertebrates). Differences in these parameters among catchments were tested by analysis of variance, while relationships of the parameters with soil salinity along the salinity gradient in Lemon catchment were tested by linear regression. All variables were tested for normality and transformed, if necessary, prior to analysis.

Similarity in species composition of plants and invertebrates among quadrats were estimated from square root transformed percentage cover or frequency data, by using the Bray-Curtis coefficient (Bray and Curtis 1957). Patterns of similarity were assessed with non-metric multidimensional scaling (Kruskal and Wish 1978) of pairwise similarity coefficients. Stress values, or the distortion between the original similarity rankings and similarity rankings in the ordination plot, were compared at different dimensionalities of ordination to determine the most suitable plot dimensions. A stress value of 0.2 or less was taken to indicate a reasonable representation of rank similarities by the ordination plot (Clarke 1993). Differences in species composition among catchments was tested by a permutation procedure applied to the pairwise similarity matrix (ANOSIM, implemented using the computer package PRIMER 6.0; Clarke and Gorley 2006). The contribution of individual species to the dissimilarity among catchments was assessed by averaging the Bray-Curtis similarity term for each species over all pairwise quadrat combinations, using the SIMPER procedure in PRIMER 6.0 (Clarke and Gorley 2006). Differences in species composition along the salinity gradient in Lemon catchment were tested by constructing a similarity matrix between all pairs of quadrats from Euclidean distance estimates of soil salinity, and correlating this matrix with the matrix of Bray-Curtis similarities using the RELATE procedure in PRIMER 6.0 (Clarke and Gorley 2006). The contribution of individual species to the dissimilarity among quadrats with different soil salinities was assessed by binning quadrats into soil salinity groups (low = $EC_{1.5} < 30 \text{ mSm}^{-1}$; high = $EC_{1.5} \geq 30 \text{ mSm}^{-1}$) and then averaging the Bray-Curtis similarity term for each species over all pairwise quadrat combinations, using the SIMPER procedure in PRIMER 6.0 (Clarke and Gorley 2006).

Laboratory studies

Seeds and seed collection

Eleven native riparian plant species found in the catchments were chosen for study: the tree *Corymbia calophylla*; the shrubs *Astartea fascicularis*, *Bossiaea eriocarpa*, *Hakea lissocarpha*, *Hakea prostrata*, *Hakea undulata*, *Hakea varia*, *Hypocalymma angustifolium*, *Melaleuca viminea* and *Xanthorrhoea preissii*; and the grass *Neurachne alopecuroidea*. These species were chosen based on their observed distribution in relation to soil salinity in the field, and on the availability of their seeds. Seeds were collected from within the Collie River provenance in 2007, by local native seed suppliers. The seeds were coated with Dryacide® grain protectant and stored until testing commenced. Prior to testing, a subsample of each seed batch was tested for viability by making a longitudinal cut; seeds which were firm and creamy-white in colour were assumed to be viable. The remaining seeds in each batch were weighed, sterilised using 1% sodium hypochlorite solution for ten minutes, then rinsed with distilled water and allowed to dry.

Germination trials

Germination response of the seeds was tested at various salinity treatments, prepared by dissolving NaCl in distilled water. Treatment levels are expressed as equivalent $EC_{1.5}$ soil conductivity levels in $mS\ m^{-1}$. Seeds of all plant species were tested at 0, 100, 120 and 140 $mS\ m^{-1}$. Eight replicates of each species were exposed to the treatment levels. Each replicate consisted of 10 seeds, placed on two sheets of Whatman No. 1 filter paper in a 9 cm Petri dish. The filter paper was pre-soaked in 5 mL of test solution, with an additional 2 ml added to the Petri dish, enabling the filter paper to be moist, without free water collecting in the container. The Petri dishes were covered with lids and randomly arranged in a dark incubator set at 15°C.

Dishes were examined daily and any seeds that had germinated were counted and removed. Germination was deemed to have occurred once the radicle was at least three times the length of the seed. The test was considered complete once seeds had germinated or until no further germination had occurred after two successive counts, after which any un-germinated seeds were cut in half to test for viability. Percentage germination was calculated for each replicate and compared among treatments and species by analysis of variance, after arcsine transformation.

Results

Soil salinity

There was a significant difference in soil salinity between catchments, with salinity being greater in Lemon catchment (Table 1; ANOVA, $F_{1,28} = 13.36$, $P = 0.001$). In Lemon catchment, salinity within quadrats was inversely related to distance downstream from the cleared portion of the catchment (Figure 1; $r^2=0.48$, $F_{1,18}$, $P = 0.001$). Comparisons of plant and invertebrate diversity between catchments therefore utilised the first 10 upstream quadrats at Lemon catchment, where soil salinity was usually greater than 30 $mS\ m^{-1}$ (see Figure 1).

Table 1. Mean, standard error and range of soil salinities ($EC_{1.5}$) measured in all quadrats in each catchment.

Catchment	Mean salinity ($mS\ m^{-1}$)	Standard error ($mS\ m^{-1}$)	Range ($mS\ m^{-1}$)
Ernies	11.4	1.8	5.6 – 18.6
Lemon	32.4	3.9	14.4 – 71.8

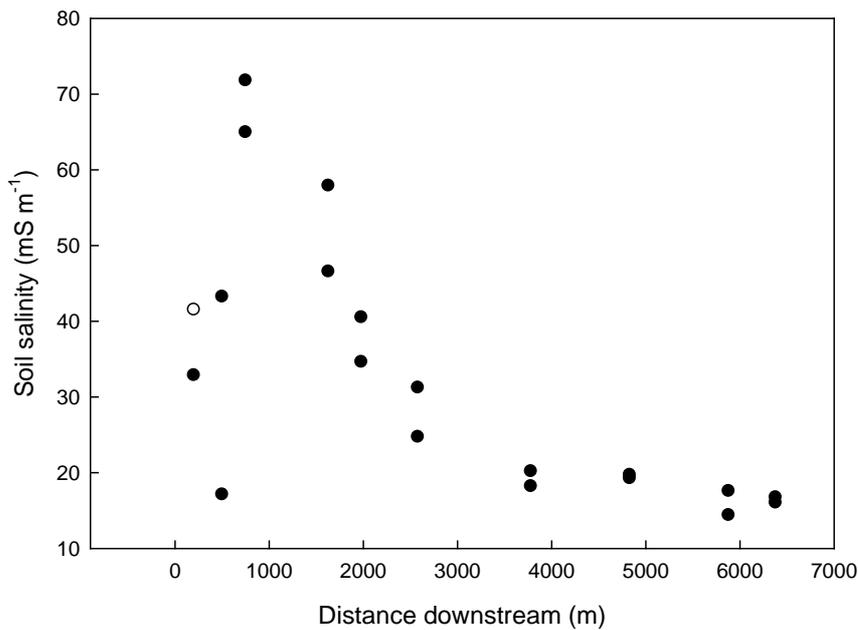


Figure 1. Relationship between soil salinity ($EC_{1.5}$, $mS\ m^{-1}$) and distance downstream (m) from the cleared area at Lemon catchment.

Plant species diversity and composition

All plant species diversity measures were greater in Ernies than in Lemon catchment, although this difference was significant only for d (Table 2; $F_{1,20} = 4.94$, $P = 0.04$). All plant life forms were present in both catchments (Figure 2), with shrubs being relatively more common in Ernies catchment ($F_{1,20} = 21.14$, $P = 0.002$) and annual grasses and herbs being relatively more common in Lemon catchment (for annual grasses, $F_{1,20} = 31.65$, $P < 0.0001$; for annual herbs $F_{1,20} = 5.32$, $P = 0.03$). There was a significant difference in plant species composition between catchments (Figure 3; ANOSIM, $\rho = 0.54$, $P = 0.001$). SIMPER analysis showed that major contributors to this dissimilarity between catchments were the shrubs *Hypocalymma angustifolium* and *Hakea prostrata*, which were relatively more common in Ernies catchment, and the shrubs, *Melaleuca viminea* and *Xanthorrhoea preissii*, which were relatively more common in Lemon catchment.

Table 2. Mean (with standard error in parentheses) number of plant species (S), Margalef's species richness index (d) and Shannon-Wiener species diversity index (H) in Ernies and Lemon catchments.

Catchment	S	d	H
Ernies	13.1 (1.4)	2.76 (0.29)	1.46 (0.20)
Lemon	10.2 (1.3)	1.89 (0.26)	1.05 (0.18)

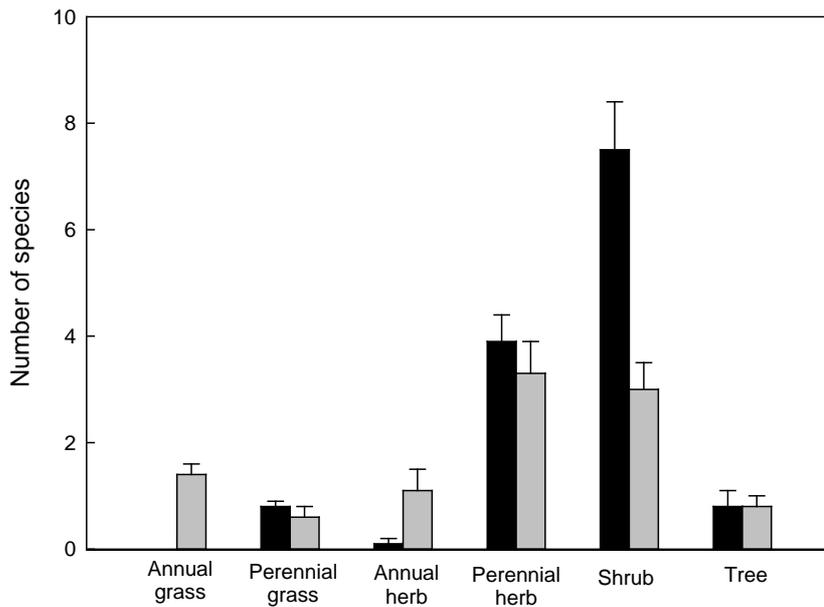


Figure 2. Mean number of species per quadrat (with standard error bars) of each major plant life-form in Ernies (black) and Lemon (grey) catchments.

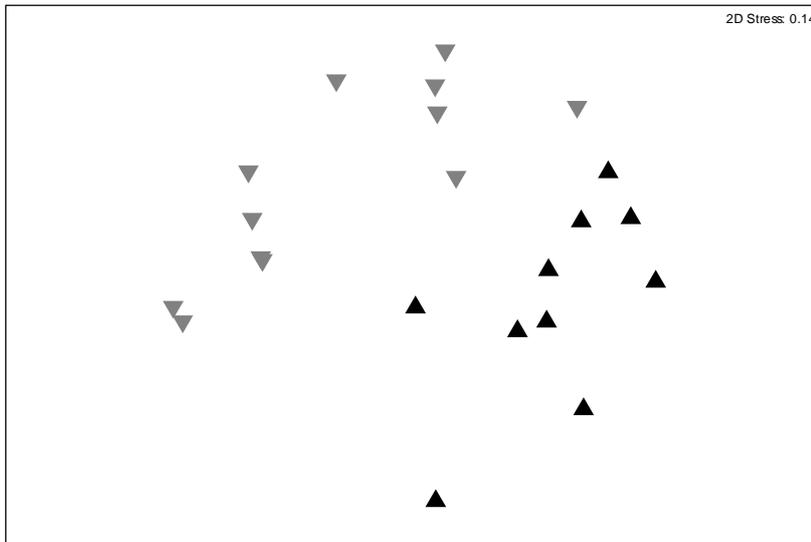


Figure 3. Multidimensional scaling plot of quadrat samples from Ernies (black) and Lemon (grey) catchments, grouped by Bray-Curtis similarity coefficients of plant species composition.

Within Lemon catchment, all plant species diversity measures were inversely related to soil salinity (Figure 4), although this relationship was significant only for S ($r^2 = 0.22$, $F_{1,19} = 5.13$, $P = 0.04$) and H ($r^2 = 0.41$, $F_{1,19} = 12.28$, $P = 0.002$). All plant life forms were found along the salinity gradient, with no significant relationships with soil salinity. There was some separation of plant species composition with soil salinity, although it was not quite significant (Figure 5; RELATE, $\rho = 0.18$, $P = 0.07$). SIMPER analysis showed that the shrubs *Acacia*

saligna, *Hypocalymma angustifolium*, *Hakea varia*, *Astartea fascicularis*, *Hakea prostrata*, *Loxocarya flexuosa* and *Bossiaea eriocarpa*, and the perennial grasses *Microlaena stipoides* and *Neurachne alopecuroidea* were relatively more common at lower soil salinities, while the annual herb *Cotula coronopifolia*, the tree *Corymbia calophylla* and the shrubs *Melaleuca viminea*, *Kunzea micrantha* and *Xanthorrhoea preissii* were relatively more common at higher soil salinities.

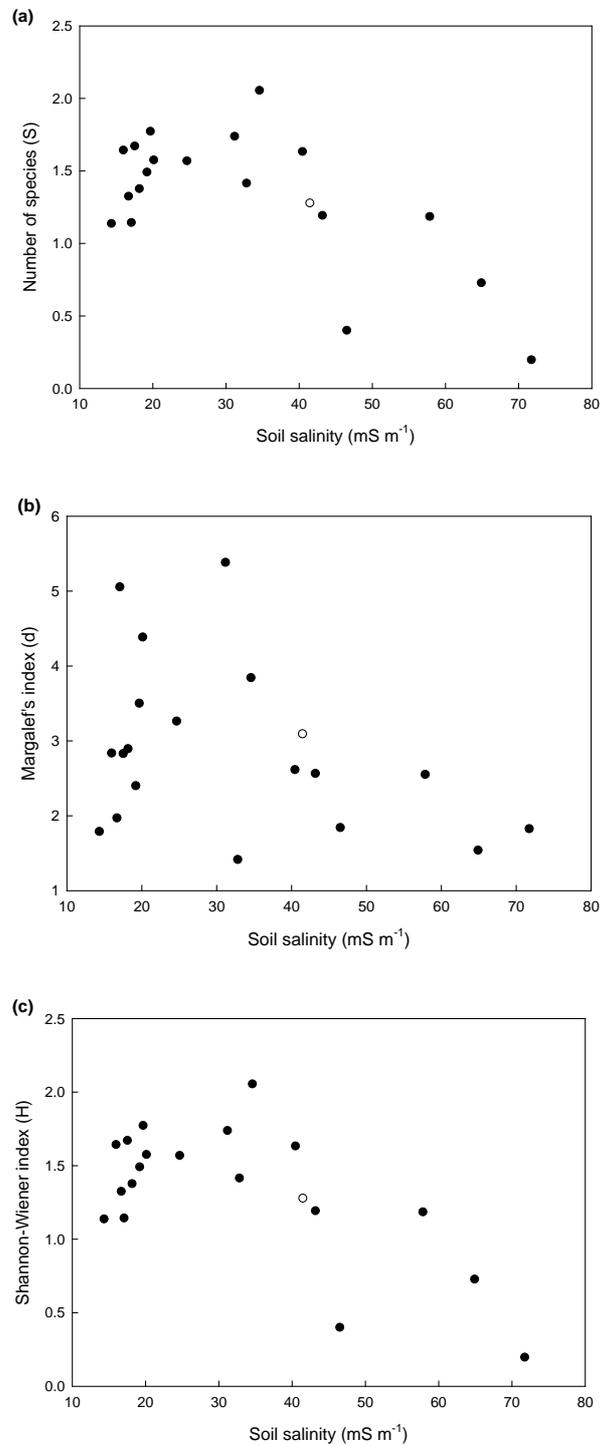


Figure 4. Relationship between soil salinity ($EC_{1.5}$, $mS\ m^{-1}$) and (a) number of plant species (S), (b) Margalef's species richness index (d) and (c) Shannon-Wiener species diversity index (H) in quadrat samples from Lemon catchment.

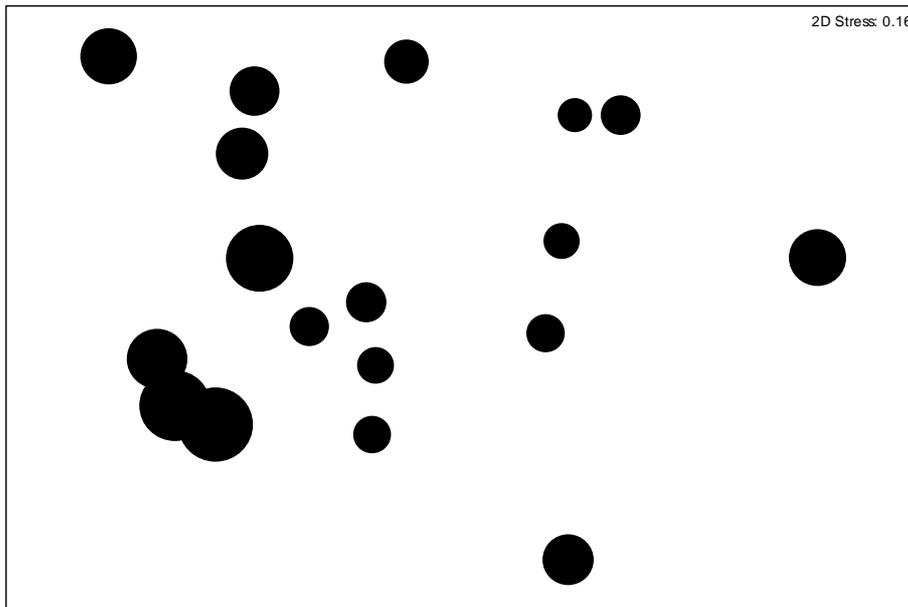


Figure 5. Multidimensional scaling plot of quadrat samples from Lemon catchment, grouped by Bray-Curtis similarity coefficients of plant species composition. The soil salinity value for each quadrat ($EC_{1.5}$, $mS\ m^{-1}$) is represented by the size of the superimposed bubble.

Invertebrate species diversity and composition

All invertebrate species diversity measures were greater in Ernies than in Lemon catchment, although the difference was significant only for H (Table 3; $F_{1,18} = 162.89$; $P < 0.0001$). All invertebrate feeding groups were relatively more common in Ernies than in Lemon catchment and this difference was significant for predators (Figure 6; $F_{1,18} = 4.10$, $P = 0.05$). There was a significant difference in invertebrate species composition between catchments (Figure 7; ANOSIM, $\rho = 0.20$, $P = 0.002$). This was due principally to a significantly greater number of species of Araneae in Ernies than in Lemon catchment (15.5 ± 3.9 compared to 5.9 ± 1.2 ; $F_{1,18} = 5.57$, $P = 0.03$) and a significantly greater number of species of Diptera in Lemon than in Ernies catchment (25.9 ± 6.5 compared to 4.8 ± 1.2 ; $F_{1,18} = 10.16$, $P = 0.005$).

Table 3. Mean (with standard error in parentheses) number of invertebrate species (S), Margalef's species richness index (d) and Shannon-Wiener species diversity index (H) in Ernies and Lemon catchments.

Catchment	S	d	H
Ernies	19.8 (2.6)	5.0 (0.4)	2.5 (0.1)
Lemon	16.5 (3.0)	4.7 (0.7)	0.9 (0.1)

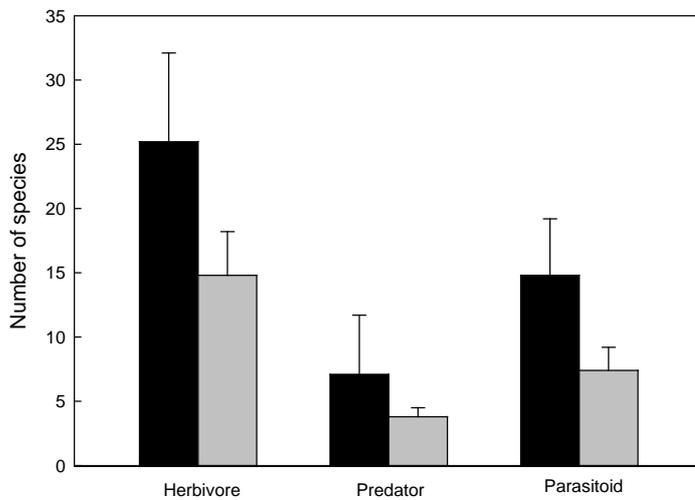


Figure 6. Mean number of species per quadrat (with standard error bars) of each major invertebrate feeding group in Ernies (black) and Lemon (grey) catchments.

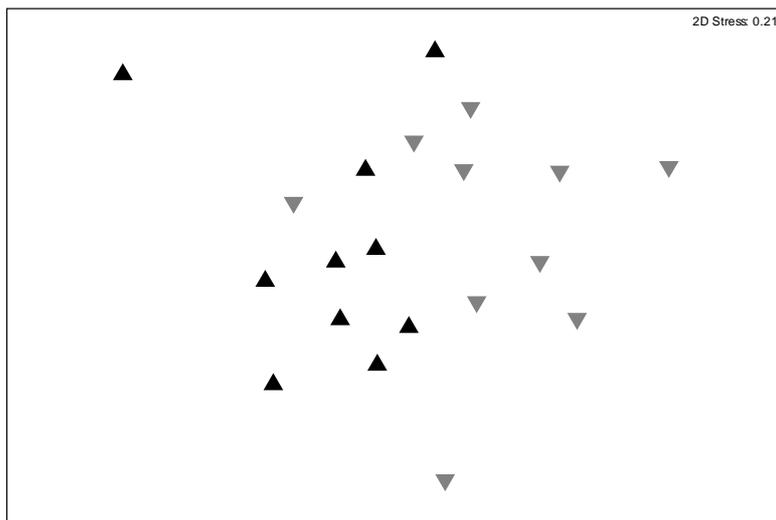


Figure 7. Multidimensional scaling plot of quadrat samples from Ernies (black) and Lemon (grey) catchments, grouped by Bray-Curtis similarity coefficients of invertebrate species composition.

Within Lemon catchment, all invertebrate species diversity measures were significantly inversely related to soil salinity (Figure 8; for S, $F_{1,18} = 7.02$, $P = 0.02$; for d, $F_{1,18} = 6.43$, $P = 0.02$; for H, $F_{1,18} = 6.44$, $p = 0.02$). Invertebrate species composition was also significantly related to soil salinity values (Figure 9; $\rho = 0.26$, $P = 0.02$). This was due principally to a significant decrease in the number of species of predators, particularly Aranaea, as soil salinity increased (Figure 10; for all predators, $r^2 = 0.23$, $F_{1,18} = 5.37$, $P = 0.03$; for Aranaea, $r^2 = 0.30$, $F_{1,18} = 7.89$, $P = 0.01$). There were also moderately strong negative relationships between soil salinity and the species richness of a number of other invertebrate orders, including Thysanoptera ($r^2 = 0.27$, $F_{1,18} = 6.73$, $P = 0.02$) and Collembola ($r^2 = 0.23$, $F_{1,18} = 5.28$, $P = 0.03$).

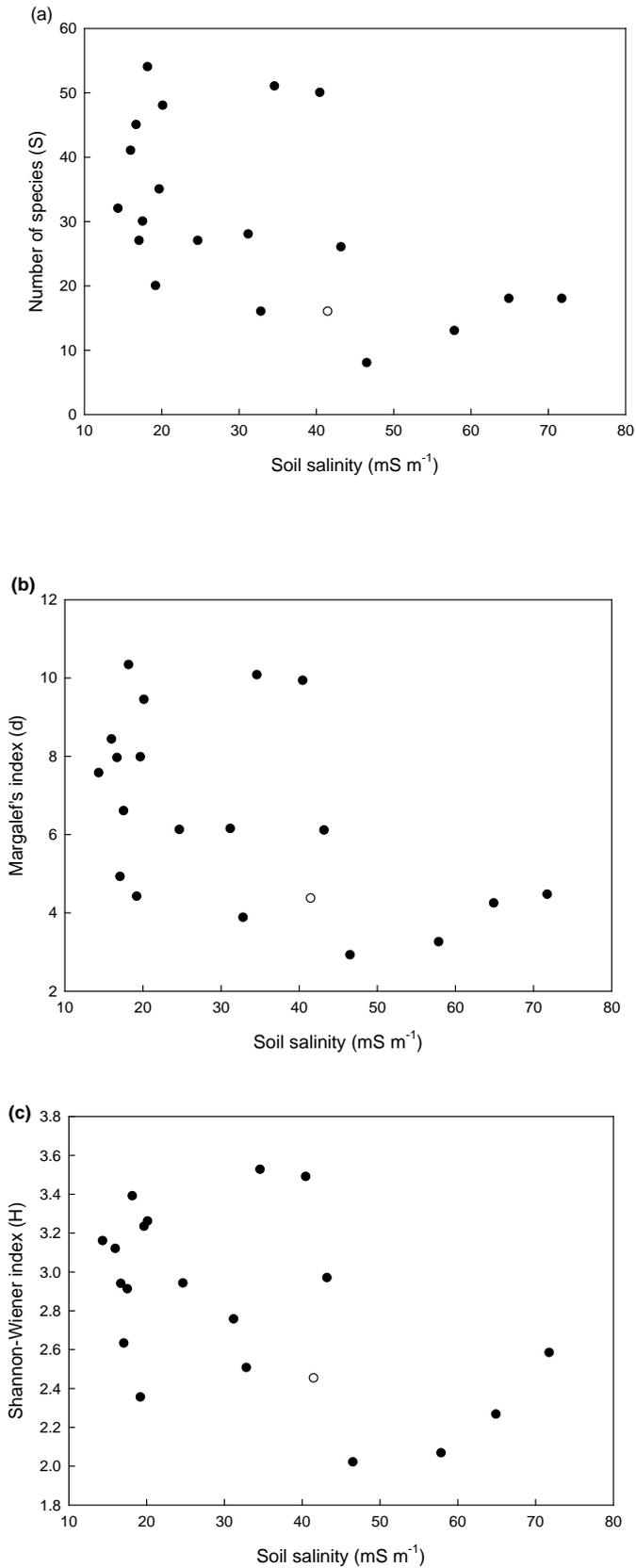


Figure 8. Relationship between soil salinity ($EC_{1.5}$, $mS\ m^{-1}$) and (a) number of invertebrate species (S), (b) Margalef's species richness index (d) and (c) Shannon-Wiener species diversity index (H) in quadrat samples from Lemon catchment.

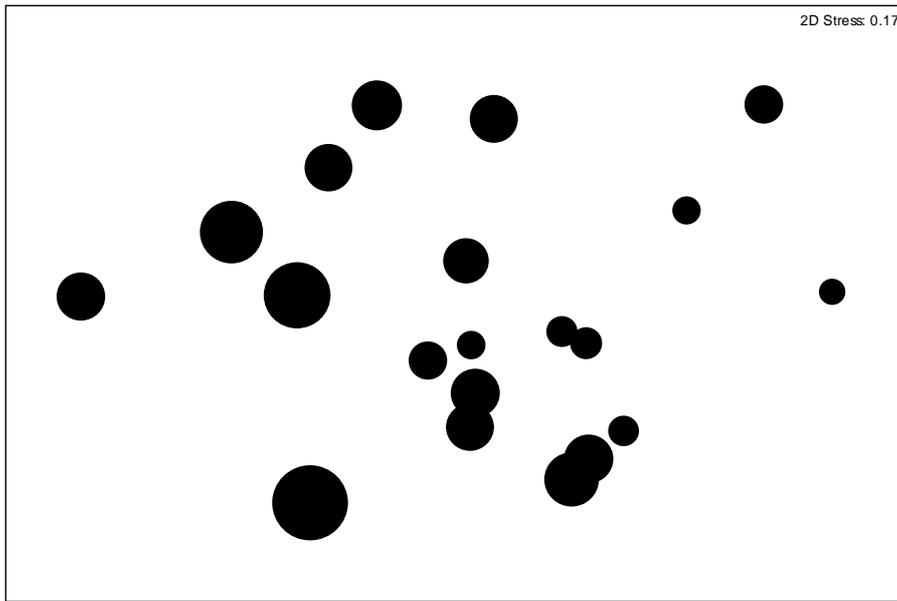


Figure 9. Multidimensional scaling plot of quadrat samples from Lemon catchment, grouped by Bray-Curtis similarity coefficients of invertebrate species composition. The soil salinity value for each quadrat ($EC_{1.5}$, $mS\ m^{-1}$) is represented by the size of the superimposed bubble.

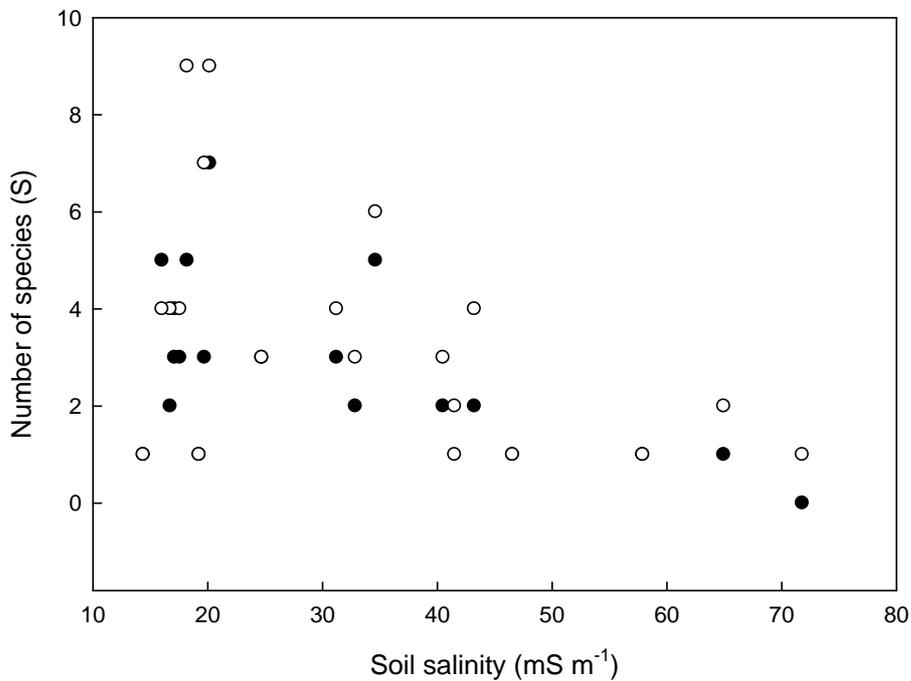


Figure 10. Relationship between soil salinity ($EC_{1.5}$, $mS\ m^{-1}$) and the number of species of all predators (white) and Araneae (black) in quadrat samples from Lemon catchment.

Trophic interactions

Over the salinity gradient in Lemon catchment, invertebrate species diversity measures were positively related to plant species diversity measures (Figure 11), significantly so for S ($r^2 = 0.19$, $F_{1,18} = 4.20$, $P = 0.05$) and H ($r^2 = 0.39$, $F_{1,18} = 11.57$, $P = 0.003$). When the number of species of each invertebrate order were compared to the number of plant species, significant positive relationships were found only for Aranaea ($r^2 = 0.47$, $F_{1,18} = 16.11$, $P = 0.0008$) and Hemiptera ($r^2 = 0.21$, $F_{1,18} = 4.92$, $P = 0.04$) (Figure 12).

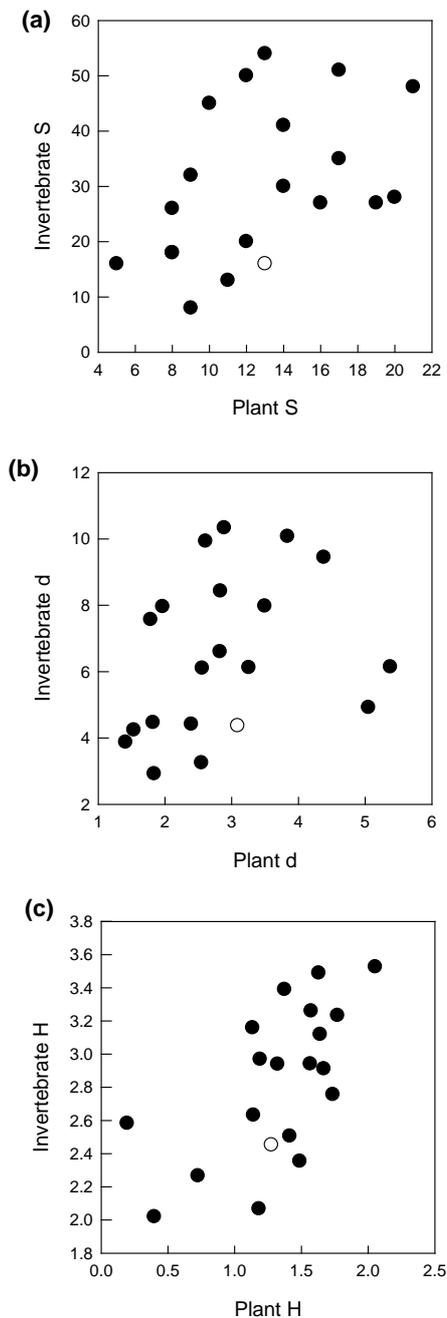


Figure 11. Relationship between plant and invertebrate species diversity measures over all quadrats within Lemon catchment: (a) number of species (S); (b) Margalef's species richness index (d); (c) Shannon-Wiener diversity index (H).

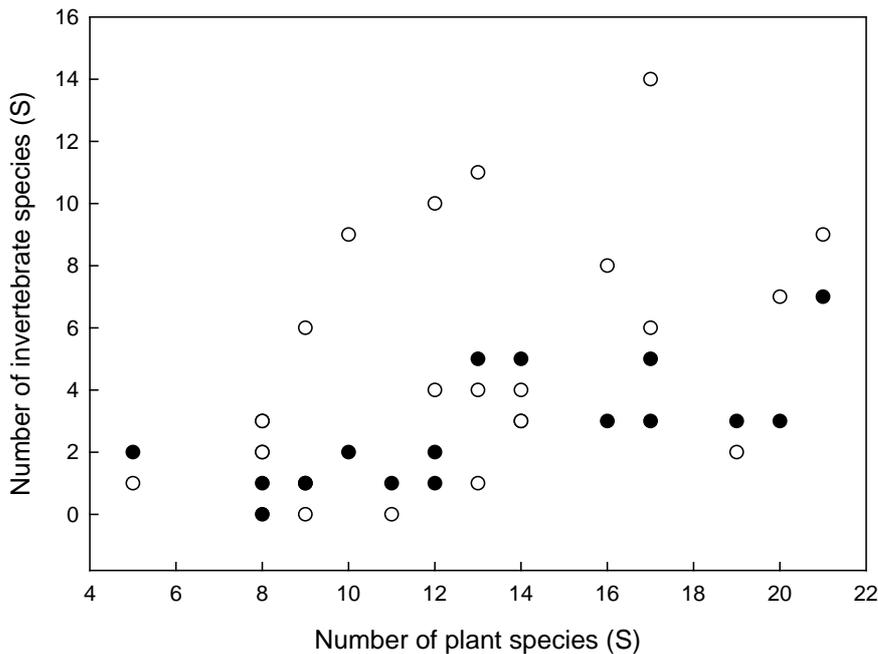


Figure 12. Relationship between the number of species of plants and the number of species of Araneae (black) and Hemiptera (white) over all quadrats in Lemon catchment.

Plant germination trials

Germination rate differed significantly among species (2-way ANOVA, $F_{10,308} = 156.46$, $P < 0.0001$), with least square means (LSM), averaged over salinity treatments, varying from 12.5% for *Astartea fascicularis* to 95.0% for *Hakea lissocarpha*. There was also a significant difference in germination rate among salinity treatments ($F_{3,308} = 40.66$, $P < 0.0001$), with germination being greater at 0 mS m^{-1} (LSM, averaged over species = 53.9%) and least at 140 mS m^{-1} (LSM = 32.4%). In addition, there was a significant interaction term ($F_{30,308} = 4.87$, $P < 0.0001$), indicating that species differed in their germination response at different salinities.

Single species analyses found significantly lower germination rates at higher salinity treatments for five of the 11 species tested: *Bossiaea eriocarpa* ($F_{3,28} = 12.44$, $P < 0.0001$); *Hypocalymma angustifolium* ($F_{3,28} = 4.14$, $P < 0.02$); *Hakea lissocarpha* ($F_{3,28} = 2.91$, $P = 0.05$); *Hakea undulata* ($F_{3,28} = 25.38$, $P < 0.0001$); and *Hakea varia* ($F_{3,28} = 11.91$, $P < 0.0001$). There was also a strong, but non-significant, negative trend between germination rates and salinity for *Hakea prostrata* ($F_{3,28} = 2.52$, $P = 0.08$), but no effects of salinity on the germination rates of *Astartea fascicularis*, *Corymbia calophylla*, *Melaleuca viminea*, *Neurachne alopecuroidea* and *Xanthorrhoea preissii* (Figure 13).

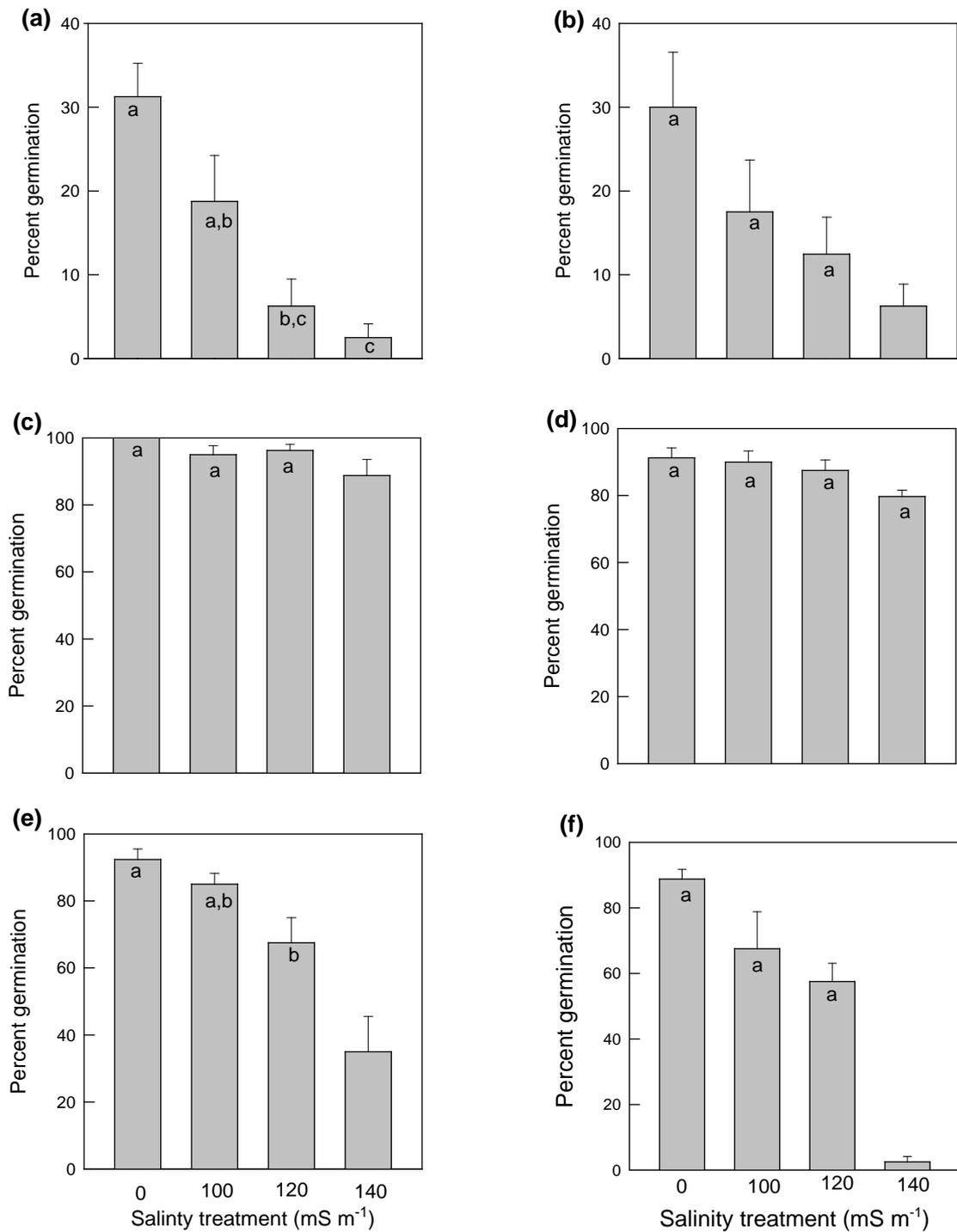


Figure 13. Percentage germination (with standard error bars) at four different salinity treatments for 6 different plant species; (a) *Bossiaea eriocarpa*, (b) *Hypocalymma angustifolium*; (c) *Hakea lissocarpa*, (d) *Hakea prostrata*, (e) *Hakea undulata*, (f) *Hakea varia*. For each species, values with the same letter are not significantly different by Tukey's HSD test.

Discussion

It has been estimated that more than 20,000 km of streams and rivers in Australia are currently affected by secondary salinisation (National Land and Water Resources Audit 2001). Riverine salinisation occurs as a result of the direct seepage of saline groundwater into the river system, by salt deposited on the land surface being washed into drainage lines, and by the discharge of saline groundwater from drainage and pumping schemes used to repair salinised land. The adverse effects of secondary salinisation on aquatic biota has now been demonstrated in a range of observational and experimental studies (Hart *et al.* 1990, 1991; Bunn and Davies 1992; Mitchell and Richards 1992; Metzeling 1993; Zalizniak *et al.* 2006; Horrigan *et al.* 2007; Nielsen *et al.* 2008; James *et al.* 2009). It has also been postulated that stream salinisation will affect terrestrial, riparian flora and fauna (Hart *et al.* 1990, 1991), although empirical confirmation of these effects from field studies are rare.

We have previously reported negative correlations between soil salinity and riparian plant species diversity, both among adjacent catchments and along a salinity gradient within the same catchment of the Collie River in the south west of Western Australia (Lymbery *et al.* 2003; Doupé *et al.* 2006). These findings were confirmed in the present study and are presumably due to the adverse effects of high soil salinity (or some correlated environmental variable) on particular plant species. Soil salinities ($EC_{1.5}$) in the upstream area of Lemon catchment were as high as 72 mS m^{-1} , much greater than values which have found to impair the growth or survival of riparian plants in glasshouse studies (Hart *et al.* 1990, 1991).

As in our previous studies (Lymbery *et al.* 2003; Doupé *et al.* 2006), we found evidence that increasing salinity was associated with a change in the species composition of riparian plants. By comparing plant species composition both between catchments with different soil salinities and along a salinity gradient within a catchment, we have been able to provide some more detailed inferences of relationships between soil salinity and plant species abundance. In a previous study of differences among catchments, for example, we found a reduction in the relative abundance of perennial grasses and herbs, and an increase in the relative abundance of annual grasses and herbs in the more saline Lemon catchment (Lymbery *et al.* 2003). Comparisons over the salinity gradient in Lemon catchment, however, now indicate that the greater relative abundance of annuals compared to perennials does not appear to be related to differences in soil salinity; rather it may be a function of clearing at Lemon leading to invasion of more annual weed species in the whole catchment. Despite the absence of any general associations between plant life forms and soil salinity, by considering the results of this study and of Lymbery *et al.* (2003) and Doupé *et al.* (2006), it does appear that certain plant species are consistently more abundant at low salinities, while other species are consistently more abundant at high salinities.

The present study has provided experimental support for differences in the salinity tolerance of some of these plant species. In controlled experiments, increasing salinity adversely affected germination rates in *Bossiaea eriocarpa*, *Hypocalymma angustifolium*, *Hakea lissocarpha*, *Hakea undulata* and *Hakea varia*, all of which were relatively more common at low soil salinities, while there was no effect of salinity on germination rates in *Melaleuca viminea*, *Xanthorrhoea preissii* and *Corymbia calophylla*, which were all relatively more common at high soil salinities. There are however, some unexplained anomalies which need further investigation. First, much greater salinities were required to significantly reduce germination rates in laboratory experiments than are associated with differences in plant

composition in the field in Lemon catchment. Second, some species which appeared to be adversely affected by high salinity conditions in the field, such as *Astartea fascicularis* and *Neurachne alopecuroidea*, showed no germination response to increasing salinity in laboratory experiments. Germination is, of course, only one component of life history which may be affected by soil salinity and further experiments are needed to examine the effects of increasing salinity on growth and reproduction of riparian plants. Such data will not only provide an explanation of observed associations in the field between salinity and plant species composition, but will also provide valuable data on the environmental requirements of riparian plant species, which are essential for rehabilitation programs in riparian zones in south western Australia (Hancock *et al.* 1996).

Despite the demonstrated associations between soil salinity and plant species diversity and composition, we have found no evidence from this or our previous studies of a relationship between salinity and plant life form, functional group diversity or primary productivity. We have interpreted this as a potential decoupling between plant species diversity and riparian ecosystem functioning in response to increasing soil salinity in Lemon catchment (Doupé *et al.* 2006). One problem with our previous studies, however, is that they considered only one trophic level (i.e. riparian plants). There is increasing consensus that relationships between biodiversity and ecosystem function need to consider not only horizontal diversity within trophic levels, but also vertical diversity among different trophic levels (Duffy 2002; Montoya *et al.* 2003; Ives *et al.* 2005; Schmitz 2006).

In the present study, we found that increasing soil salinity was associated not only with reduced diversity of riparian plant species and a change in plant species composition, but also with reduced diversity of plant-dwelling invertebrate species and a change in invertebrate species composition. Furthermore, increasing soil salinity in Lemon catchment was also associated with a change in invertebrate functional group composition, caused largely by a negative relationship between salinity and the number of species of predatory spiders. Spiders are an important component of terrestrial ecosystems, with potentially large effects on ecosystem function through their role in trophic cascades (Churchill 1997; New 1999; Schmitz 2006). The decrease in spider species which is associated with increasing soil salinity in riparian ecosystems may have both top-down effects, through decreasing predation on herbivorous invertebrates, and bottom-up effects, through loss of a valuable prey item for vertebrates, including amphibians, reptiles, birds and mammals, that use the riparian zone for foraging and protection (Naiman and Decamps 1997).

There are a number of possible causal explanations for the negative association found in this study between soil salinity and spider diversity. Although the direct effects of salinity on terrestrial invertebrates have been poorly studied, there may be adverse physiological consequences through contact with increased salt concentrations in soil or plant tissue. A more likely causal pathway is through the indirect effects of salinity on plant species diversity. Reduced spider diversity may be associated with reduced plant diversity because plant diversity is correlated with the diversity of herbivorous prey species such as hemipterans, thysanopterans and collembolans. Alternatively (or additionally), less diverse plant communities may provide fewer feeding niches, as spiders often show highly specific associations with certain plant species and/or structural strata (Uetz 1991). We found much stronger associations between soil salinity and spider diversity than between soil salinity and herbivore diversity, which may indicate that salinity is acting upon spiders via reduced habitat complexity rather than reduced diversity of prey. Previous studies have also shown that species diversity, particularly of web-building spiders, is more often associated with

vegetation structural diversity or habitat complexity than with prey availability (Robinson 1981; Greenstone 1984; McNett and Rypstra 2000; Sanders *et al.* 2008).

Conclusions

This study has provided strong evidence that increasing stream salinity in the south west of Western Australia is associated a reduced diversity of riparian plant species and plant-dwelling invertebrate species. These changes may affect the functioning of the riparian ecosystem, particularly because they appear to impact on the functional diversity of riparian invertebrates, with a negative correlation between salinity and the number of predatory spider species. Field observations of correlations between particular plant species and soil salinity, and laboratory studies of the effect of salinity on germination rates, have identified a number of species of riparian plants which appear to be particularly salt-sensitive and are therefore most at risk from increasing stream salinisation. A number of other plant species appear to be more salt-tolerant and could be more widely utilised in riparian revegetation programs.

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