5 Species richness and composition of freshwater fish communities in New South Wales rivers

P. C. Gehrke

Cooperative Research Centre for Freshwater Ecology, NSW Fisheries Research Institute, PO Box 21 Cronulla, New South Wales 2230

Summary

Fish from montane, slopes and both regulated and unregulated lowland river reaches were studied in the North Coast, South Coast, Darling and Murray regions of New South Wales to identify fish communities on a large spatial scale within the State, and to contrast characteristics of fish communities in New South Wales with communities elsewhere. Five replicate rivers of each type were selected from each region. Fish communities were sampled during summer and winter in two consecutive years using a standardised suite of gear which maximised the range of species caught at each site. Spatial differences among regions and river types were the major sources of variation in the composition of fish communities. In contrast, surprisingly little temporal variation was detected over two years. Each region showed a distinct character to its fish communities, which converged in montane reaches and diverged with increasing distance downstream. Thus rivers formed five groups based on fish distributions, identified as montane rivers, and the slopes and lowland rivers of the four geographic regions. Similarly, the fish fauna could be separated into four groups representing montane species, South Coast species, North Coast species, and a combined group of Murray-Darling species. Using catchment area and stream length upstream of each site as surrogates for habitat availability, species richness increased with increasing habitat availability in both North Coast and South Coast regions by both replacement and the addition of new species. In contrast, species richness in the Darling and Murray regions reached a maximum in the slopes reaches and then declined, reflecting a loss of species with increasing distance downstream in the lowland reaches. Dominant fish species display an increase in trophic diversity with increasing distance downstream, suggesting a downstream increase in diversity of available food types that is not reflected in species richness in inland rivers. The small number of species in the freshwater fish fauna of New South Wales is typical of the faunas of similar climatic regions world-wide. The New South Wales fish fauna can be classified into Montane, North Coast, South Coast and Murray-Darling units for fisheries management purposes, while there is some biological justification for managing the Darling River and its tributaries as separate entities from rivers in the Murray region. The decline in species richness with increasing distance downstream in lowland reaches of inland rivers is contrary to general longitudinal patterns of species richness in rivers, and suggests a need for remedial management of lowland rivers to increase habitat diversity. The fish communities identified in this study form logical entities for fisheries management, and provide an opportunity to manage fisheries to ensure the sustainability of riverine ecosystems rather than sustaining single species.

INTRODUCTION

Between 180 and 200 native species of freshwater fish are known from Australia (Merrick and Schmida 1984; Allen 1989; Paxton *et al.* 1989; McDowall 1996), with 81 species recorded from New South Wales (Harris 1995). These numbers are very low when compared with other continents, leading some authors to describe the fauna as impoverished or depauperate (Allen 1989). However, considering the limited availability of aquatic habitats in what is recognised as the driest continent apart from Antarctica, it is hardly surprising that Australia contains so few species (Harris 1984; Lake 1995).

The influence of habitat diversity and availability on the composition of riverine fish communities is well known (e.g. Sheldon 1968; Whiteside and McNatt 1972; Gorman and Karr 1978; Evans and Noble 1979; Lake 1982; Beecher 1988; Pusey *et al*. 1995), with the greatest species diversity occurring in areas offering the greatest variety of habitats. Consequently, progressive increases in habitat availability downstream in many rivers tend to coincide with increases in the number of fish species, and the development of biotic zonation patterns within rivers (Rahel and Hubert 1991; Paller 1994).

The presence of biotic zonation in riverine fish communities raises the question of whether such communities can be considered as appropriate units for fisheries management. There is a strong rationale for joint management of fish communities and riverine environments (Karr *et al.* 1986). Indeed, Evans *et al*. (1987) argue convincingly that fish communities possess functional properties that are not only amenable to fisheries management, but which are essential to sustain optimal fishery yields. Adoption of individual species as the units of management ignores the interactions between species and between trophic levels that ultimately determine the structure and status of aquatic ecosystems. In contrast, freshwater fish communities appear to be sufficiently discrete and self-regulating that they can be managed as ecological units which both depend on and indicate the condition of the whole system (Evans *et al.* 1987). One pre-requisite for fisheries management based on fish communities is that discrete communities can actually be defined and recognised.

This chapter has three objectives. The first is to determine whether the size of the freshwater fish fauna of New South Wales is consistent with global trends, and to examine possible implications for managing fish resources. The second objective is to examine longitudinal trends in species richness in New South Wales rivers and to examine the implications of such trends. The third objective is to define large-scale fish communities that may serve as ecologically derived units for fisheries management in New South Wales.

METHODS

Site classification and selection

Sites were selected from four geographic regions as described in Chapter 1 from rivers in the North Coast, South Coast, Darling and Murray regions of New South Wales. Within each region, four river types were recognised, and defined as montane reaches (>700 m altitude), slopes reaches (>300 m and <700 m altitude in the Darling and Murray regions, or >40 m and <700 m altitude in the North Coast and South Coast regions), and lowland rivers (<300 m altitude in the Darling and Murray regions, or <40 m altitude in the North Coast and South Coast regions). Lowland rivers were further classified as being either regulated or unregulated. ‘Regulated’ rivers were defined as rivers with flows that were substantially modified from the natural condition by the operation of a dam upstream. In contrast, ‘unregulated’ rivers were defined as rivers with flows that are either completely natural, or where tributary inflows create a minimally regulated flow regime despite the existence of a dam some distance upstream. Within each region, five replicates were selected for each river reach using a modified-random process which reduced the likelihood of selecting sites that had been extensively examined in other projects by NSW Fisheries. The 80 sites selected by this process are shown in Figure 5.1. For a full description of these sites see Chapter 2.

Sampling methods

Fish were caught by applying a standardised suite of quantitative sampling methods at each site. In montane sites, the gear consisted of four 50 m passes with a backpack electrofisher (two passes in pool-edge habitats, two passes in riffle habitats), three single-wing fyke nets (30 mm stretched mesh), nine Gee traps (350x200 mm diameter, 3 mm square wire mesh) and two panel nets. Panel nets consisted of three 5 m long 2 m deep gill net panels with mesh sizes of 38, 67 or 100 mm. The different mesh-sized panels were hung in random sequence. In slopes sites, effort was increased to include three panel nets, and ten shots of two minutes duration from a boat electrofisher, in addition to the gear used in montane sites. The suite of sampling gear used in lowland sites was identical to that used in slopes reaches, with the exception that backpack electrofishing was omitted. Fyke nets and Gee traps were set 3 h before sunset and cleared 2 h after sunset. Panel nets were set 2 h before sunset and cleared 1 h after sunset. All electrofishing was conducted during daylight hours. For further details of how these methods were applied and standardised, refer to Chapter 1. Variability and selectivity among gear types are presented separately in Chapter 10.

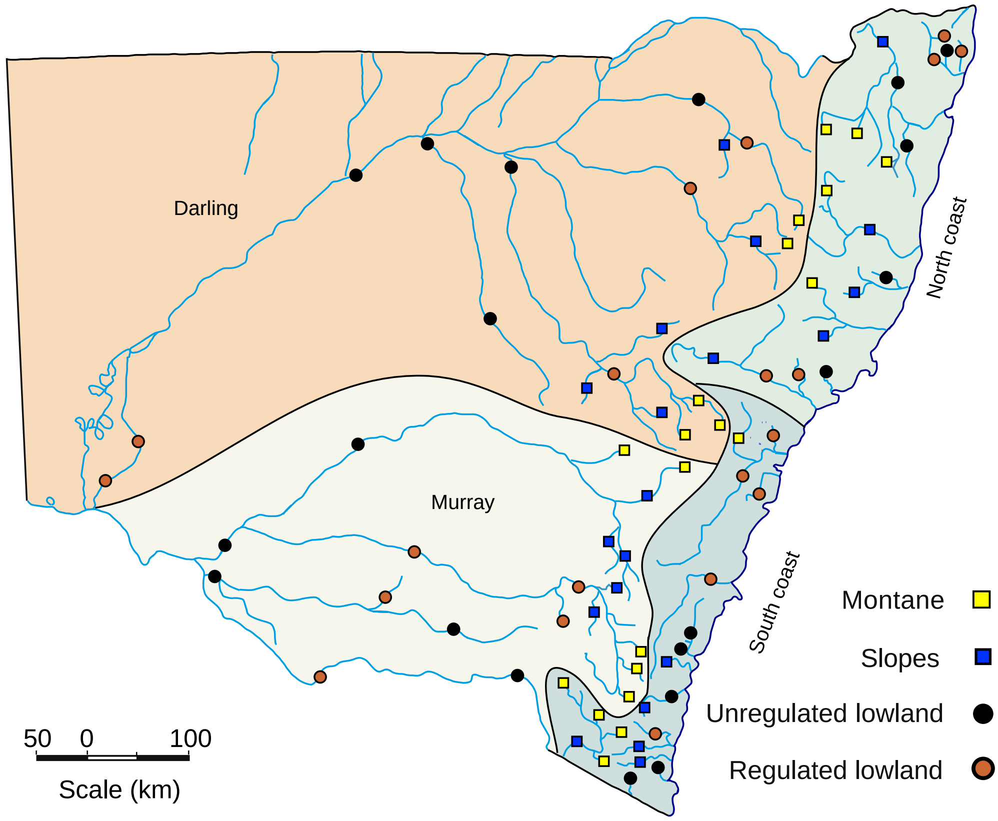

Figure 5.1 Map of study sites in New South Wales showing the types of rivers sampled in each of the four geographic regions.

Sites were sampled twice yearly over two years to obtain an indication of variability in fish communities within and between years. The sample design therefore provided four factors for analysis: river type (montane, slopes, regulated/unregulated lowland); regions (4), years (2) and times (2).

Statistical methods

Analyses in this study used the pooled catch from all sampling methods, including eels that were recorded as observed, to represent an estimate of the total fish community at each site. Scale issues relating to the number of species collected at each site and the distance of each site from the headwater source were analysed using curves of the form derived by Shepherd (1982):

$$R = aL / [\, 1 + (L / K)^{\beta} \,]$$

where *R* represents the number of species at a site and *L* is the stream length from the headwaters to the site, measured from 1:100,000 scale maps. This equation was originally derived as a generic stock-recruitment relationship, but possesses a number of desirable properties in analyses of species richness and habitat availability. As described by Shepherd (1982), the parameter *a* represents the slope of the curve at the origin, and has the dimensions in this case of species-per-kilometre of river upstream of the site sampled. The parameter *K* has the dimensions of kilometres, and represents the length of river above which density-dependent effects dominate. This may be considered as the length of river beyond which colonisation by new species occurs by replacing other species rather than by occupying vacant niches. The parameter β represents a degree of compensation for density dependence (Shepherd 1982). In an ecological context, β may be viewed as an indicator of the existence of density-independent unexploited niches, such that where $\beta < 1$, the curve continues to rise as stream length increases, indicating a capacity to support a greater number of species. Conversely, where $\beta > 1$, the curve reaches a maximum number of species and then declines with increasing stream length, suggesting a loss of niches or habitat diversity in longer streams. A value of $\beta = 1$ indicates perfect density-dependence in terms of species richness at increasing stream length, whereby new species can only enter the community at the expense of an existing species. The curve was fitted by iterative parameter estimation using the non-linear solver in Microsoft Excel 5.0 to minimise the residual sum of squares.

The same curve was used to describe the relationship between species richness and catchment area, in which case the dimensions of the parameters *a* and *K* become square kilometres, and *L* becomes the catchment area upstream of the site.

Power curves of the form $R = a\,X^b$ have often been used to describe the relationship between species richness, *R*, and some measure of the amount of habitat available *X* (Lake 1982). The performance of the power curve was evaluated against Shepherd's curve by comparing the residual sum of squares from each method.

Fish community data were analysed using PRIMER 4.0 (Plymouth Marine Laboratory) to perform hierarchical agglomerative classification analysis and multi-dimensional scaling (MDS) ordinations. Analyses of all 80 sites used catch data that was pooled over the four sampling times to conform to the data limits in PRIMER. Species abundances were transformed to the fourth root, and similarities were calculated using the Bray-Curtis similarity measure (Bray and Curtis 1957). Both classification and ordination were done on similarities among sites, as determined from relative species abundances, as well as inverse analyses of similarities among species, as determined from sites in which they occurred. All classifications used the group-average linking algorithm.

Two-way ANOSIM (ANalysis Of SIMilarities) comparisons (Clarke 1993) were done at differing strata to identify differences in fish community composition among regions, among river types, and among river types within regions. Permutation tests to estimate the probability of observed results used 5000 Monte Carlo randomisations for each comparison. SIMPER (SIMilarity PERcentages) analyses were used to identify species that contribute most to the average similarity within each region or river type treatment, and to the average dissimilarity between paired treatments.

Other fish community variables - specifically, the number of species per site, total fish abundance per site, species diversity (Shannon's H') and the proportion of native fish in the catch were analysed by factorial analysis of variance using regions, river type, years and time as factors to establish the importance of spatial (i.e. region and river type) and temporal (i.e. time and year) factors in determining community structure. The assumption for homogeneity of variances was tested using Cochran's test. Data for total abundance and species richness were transformed by $\log_{10}(x+1)$ to homogenise variances and to ensure that variances were independent of treatment means. The proportion of native fish in the catch was normalised using the arcsine transformation. Species diversity data were not transformed.

RESULTS

Catch summary

A total of 29,788 fish representing 55 species and 29 families was collected during the survey (Table 5.1). The most abundant species were the western carp gudgeon complex, *Hypseleotris* spp. (13.2%), followed by empire gudgeon, *Hypseleotris compressa* (9.7%), long-finned eels, *Anguilla reinhardtii* (8.7%), Australian smelt, *Retropinna semoni* (8.5%), carp, *Cyprinus carpio* (7.1%) and bony herring, *Nematalosa erebi* (7.0%) (Table 5.1).

Table 5.1 Summary of the fish catches from geographic regions of New South Wales sampled during the NSW Rivers Survey. Species numbers provide a key to species identification in Figure 5.8. Includes numbers of eels observed as well as those caught (see Table 2.6).

Family	Species	Species no.	Darling	Murray	North Coast	South Coast	Total
Ambassidae	*Ambassis agassizii*	2	1	0	61	0	62
	Ambassis nigripinnis	3	0	0	494	0	494
Anguillidae	*Anguilla australis*	4	0	0	1	78	79
	Anguilla reinhardtii	5	0	0	1318	1284	2602
Ariidae	*Arius graeffei*	6	0	0	59	0	59
Atherinidae	*Craterocephalus fluviatilis*	11	0	1	0	0	1
	Craterocephalus marjoriae	12	0	0	10	0	10
	Craterocephalus stercusmuscarum	13	208	14	0	0	222
Bovichtidae	*Pseudaphritis urvillii*	50	0	0	0	29	29
Carangidae	*Gnathanodon speciosus*	21	0	0	3	0	3
Carcharhinidae	*Carcharhinus leucas*	10	0	0	1	0	1
Clupeidae	*Herklotsichthys castelnaui*	24	0	0	7	0	7
	Nematalosa erebi	41	1982	100	0	0	2082
	Potamalosa richmondia	48	0	0	604	1	605
Cyprinidae	*Carassius auratus*	9	271	97	138	15	521
	Cyprinus carpio	14	1067	925	83	37	2111
Eleotridae	*Gobiomorphus australis*	22	0	0	576	399	975
	Gobiomorphus coxii	23	0	0	31	817	849
	Hypseleotris compressa	25	0	0	2616	271	2887
	Hypseleotris galii	26	0	0	295	442	737
	Hypseleotris spp.	27	3717	111	115	0	3943
	Philypnodon grandiceps	45	4	42	309	334	689
	Philypnodon sp1	46	0	0	41	109	150
Gadopsidae	*Gadopsis bispinosus*	15	0	3	0	0	3
	Gadopsis marmoratus	16	21	0	0	1	22
Galaxiidae	*Galaxias brevipinnis*	17	0	7	0	8	15
	Galaxias maculatus	18	0	0	0	533	533
	Galaxias olidus	19	162	289	260	5	716
Gobiidae	*Redigobius macrostoma*	52	0	0	0	1	1
Hemirhamphidae	*Arrhamphus sclerolepis*	7	0	0	4	0	4
Melanotaeniidae	*Melanotaenia duboulayi*	35	0	0	314	0	314
	Melanotaenia fluviatilis	36	99	2	0	0	101
Mordaciidae	*Mordacia praecox*	37	0	0	0	33	33
Mugilidae	*Liza argentea*	29	0	0	22	0	22
	Mugil cephalus	38	0	0	657	100	757
	Myxus elongatus	39	0	0	0	2	2
	Myxus petardi	40	0	0	641	118	759
Percichthyidae	*Maccullochella peelii*	30	52	0	0	0	52
	Macquaria ambigua	31	191	37	0	0	228
	Macquaria australasica	32	0	22	0	0	22
	Macquaria colonorum	33	0	0	8	1	9
	Macquaria novemaculeata	34	0	0	428	651	1079
Percidae	*Perca fluviatilis*	44	263	131	0	26	421
Platycephalidae	*Platycephalus fuscus*	47	0	0	1	0	1
Plotosidae	*Tandanus tandanus*	55	58	0	488	1	547
Poeciliidae	*Gambusia holbrooki*	20	633	30	648	360	1671
Prototroctidae	*Prototroctes maraena*	49	0	0	0	64	64
Pseudomugilidae	*Pseudomugil signifer*	51	0	0	193	0	193
Retropinnidae	*Retropinna semoni*	53	424	354	586	1164	2527
Salmonidae	*Oncorhynchus mykiss*	43	7	77	1	11	96
	Salmo trutta	54	116	70	0	97	283
Scorpaenidae	*Notesthes robusta*	42	0	0	63	8	71
Sparidae	*Acanthopagrus australis*	1	0	0	8	0	8
Teraponidae	*Bidyanus bidyanus*	8	7	2	0	0	9
	Leiopotherapon unicolor	28	106	1	0	0	107
Grand Total			**9390**	**2315**	**11084**	**7000**	**29788**

Scaling effects and species richness

Fitting curves to data requires a reasonably even distribution of values of the independent variable if the relationship is to be meaningful. In this study, both stream lengths and catchment areas tended to clump at the lower end of the scale. However, the relationships between catchment area and species richness in the South Coast region, and to a lesser extent, the Darling region, were the only curves that were under-determined, with few points at the upper end of the area scale. Relationships between species richness, and stream length and catchment area in all four regions showed a rapid initial acquisition of species with increasing habitat availability (Figure 5.2 and Figure 5.3). The rate of species acquisition, represented by the parameter a, was more rapid in coastal regions than in either the Murray or Darling regions (Table 5.2 and Table 5.3). Rivers in both the Murray and Darling regions showed a slight decline in species richness (ß > 1) with increasing stream length or catchment area into the lowland riverine reaches. In contrast, curves for rivers in the coastal regions showed an increase in the number of species with increasing distance downstream and increasing catchment area (ß < 1), with one exception of an imperceptible decline in species richness with increasing stream length in the North Coast region. The maximum number of species per site was recorded at relatively short stream lengths, near the transition zone between slopes and lowland river reaches.

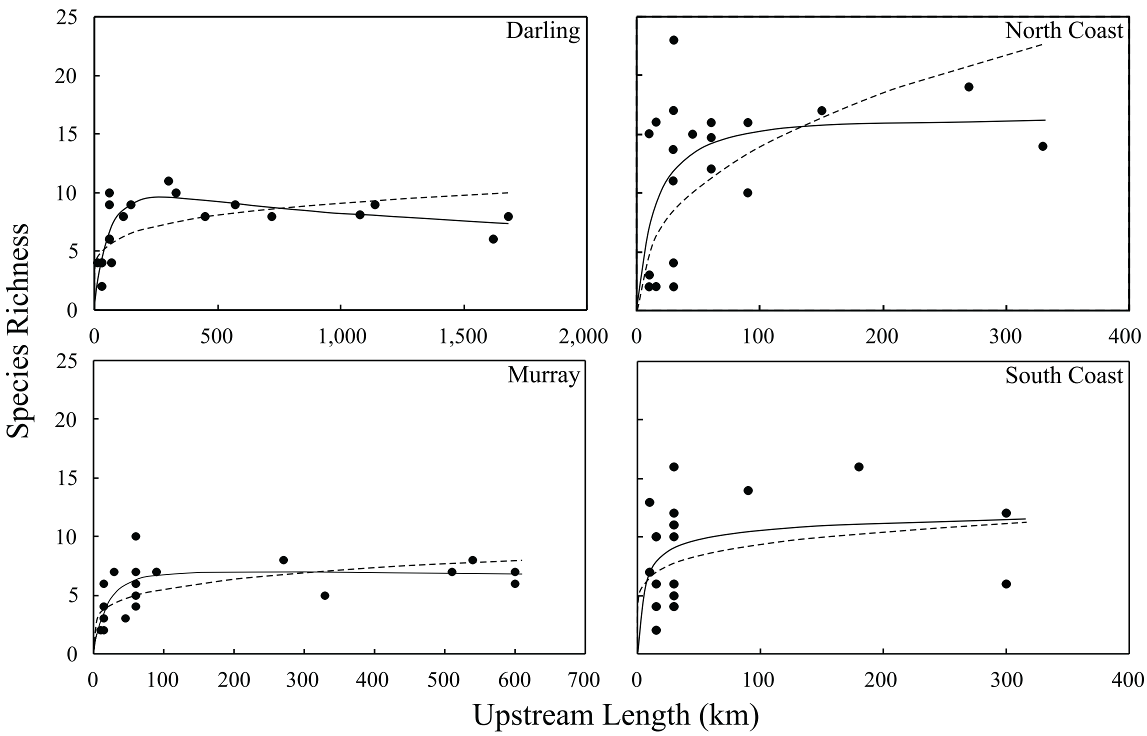

Figure 5.2 Relationships between stream length and species richness in four regions of New South Wales. Solid lines show Shepherd's curve, broken lines show power curve. Parameters for each curve are given in Table 5.2. Shepherd's curve provided a superior fit, indicated by a smaller residual sum of squares, in all regions.

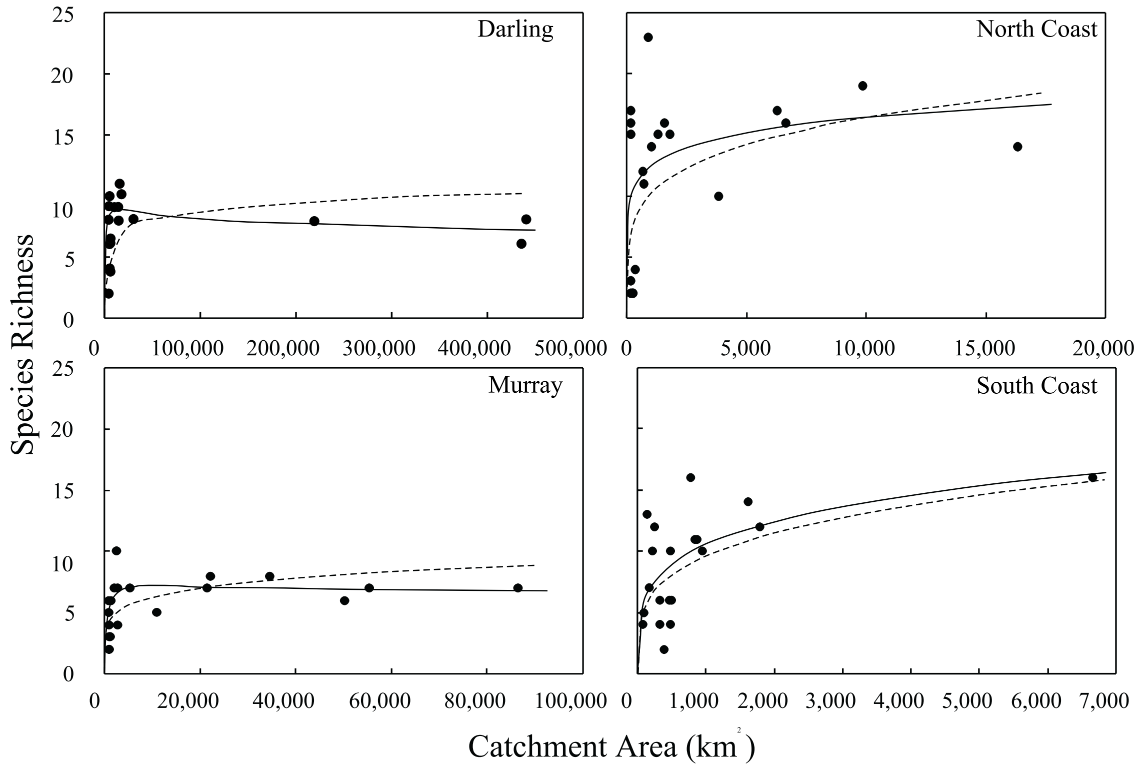

Figure 5.3 Relationships between catchment area and species richness in four regions of New South Wales. Solid lines show Shepherd's curve, broken lines show power curve. Parameters for each curve are given in Table 5.3. Shepherd's curve provided a superior fit, indicated by a smaller residual sum of squares, in all regions.

Table 5.2 Estimated parameters for Shepherd's curves and power curves fitted to relationships between stream length and species richness in fish communities in New South Wales (Figure 5.2). The residual sum of squares from fitting Shepherd's curve is smaller than the value obtained from fitting a power curve.

	Shepherd				**Power**		
Region	*a*	K	ß	RSS	*a*	*b*	RSS
Darling	0.172	93.377	1.269	43.78	2.730	0.173	95.05
Murray	0.390	23.740	1.090	49.55	2.167	0.204	68.21
North Coast	1.096	17.163	1.035	570.44	2.077	0.412	765.93
South Coast	2.251	4.230	0.952	307.45	4.519	0.158	329.39

Table 5.3 Estimated parameters for Shepherd's curves and power curves fitted to relationships between catchment area and species richness in fish communities in New South Wales (Figure 5.3). The residual sum of squares from fitting Shepherd's curve is smaller than the value obtained from fitting a power curve.

	Shepherd				**Power**		
Region	*a*	K	ß	RSS	*a*	*b*	RSS
Darling	0.009	1305.864	1.082	56.38	2.670	0.103	108.21
Murray	0.014	600.800	1.048	34.58	1.397	0.163	60.49
North Coast	0.745	9.959	0.885	648.02	2.378	0.210	712.37
South Coast	0.398	10.000	0.783	221.96	1.576	0.262	232.30

Shepherd's curve provided a superior fit to relationships between species richness and stream length, and between species richness and catchment area, as indicated by the residual sum of squares (Table 5.2 and Table 5.3). Shepherd's curve has additional advantages because use of the power curve assumes that the number of species continues to increase with increasing habitat availability. This assumption was not justified in the present study, with the result that there were serious discrepancies between the fits of the power curve and Shepherd's curve to the species-habitat relationship.

There was a good relationship between the logarithms of stream length and catchment area in all regions, with adjusted R^2 values of 0.80 for the South Coast, and 0.84 for the North Coast, Darling, and Murray regions.

Composition of fish communities

Analyses of variance for species richness, total fish abundance, species diversity and the proportion of native fish in samples all showed a significant interaction ($P<0.01$) between geographical region and river types, as well as significant main effects ($P<0.001$) for these factors (Table 5.4). The only other significant effect was the time of sampling for species diversity ($P<0.05$), and for total abundance ($P<0.01$), reflecting lower catchability of fish in winter.

Table 5.4 Summary of four-way analyses of variance for fish community variables and abundant species. Treatment effects are abbreviated R - region (fixed), T - river types (fixed), Y - year (random), S - sampling occasion (fixed). Only F-values and probabilities are given. *P<0.05, **P<0.01, ***P<0.001.

Effect	Degrees of freedom	Species richness	Shannon's diversity	Total abundance	Proportion of native species
R	3	66.657***	42.504***	30.276***	93.744***
T	3	172.448***	100.311***	50.218***	7.298***
R*T	9	10.484***	12.879***	2.640**	16.793***
Y	1	0.221 ns	0.103 ns	0.316 ns	0.003 ns
R*Y	3	1.253 ns	0.840 ns	0.480 ns	0.874 ns
T*Y	3	0.201 ns	0.139 ns	0.230 ns	2.341 ns
R*T*Y	9	0.214 ns	0.339 ns	0.434 ns	0.510 ns
S	1	6.726*	0.939 ns	15.687***	0.994 ns
R*S	3	0.457 ns	0.292 ns	1.338 ns	1.159 ns
T*S	3	0.813 ns	1.511 ns	0.251 ns	0.740 ns
R*T*S	9	0.828 ns	1.730 ns	0.401 ns	0.397 ns
Y*S	1	0.021 ns	1.597 ns	0.632 ns	0.006 ns
R*Y*S	3	1.234 ns	0.920 ns	1.098 ns	1.837 ns
T*Y*S	1	1.477 ns	0.627 ns	0.401 ns	1.238 ns
R*T*Y*S	9	0.553 ns	0.737 ns	0.413 ns	0.525 ns
Residual	256				

The interactions between region and river type for species richness, species diversity and total abundance reflect a greater abundance and diversity of fish in slopes and lowland rivers in the North Coast region compared to other regions (Figure 5.4) region. The proportion of native fish sampled in the South Coast region was high in slopes and lowland reaches, and low in montane rivers due to the dominating presence of rainbow trout. In contrast, the proportion of native fish sampled in the Murray region was lowest in slopes and lowland reaches, reflecting the dominance of carp, and high in montane rivers due to the abundance of mountain galaxias. Native species were proportionally dominant in all river types in the North Coast region.

These results demonstrate, not surprisingly, that spatial differences between regions and between river types account for a large amount of the total variation in the composition of fish communities in New South Wales rivers. Furthermore, variation between river types attributable to the low numbers of fish in montane rivers accounts for a larger proportion of total variance than variation between regions. In comparison, differences between times of sampling and between years of the survey contribute relatively little to the variability in fish communities.

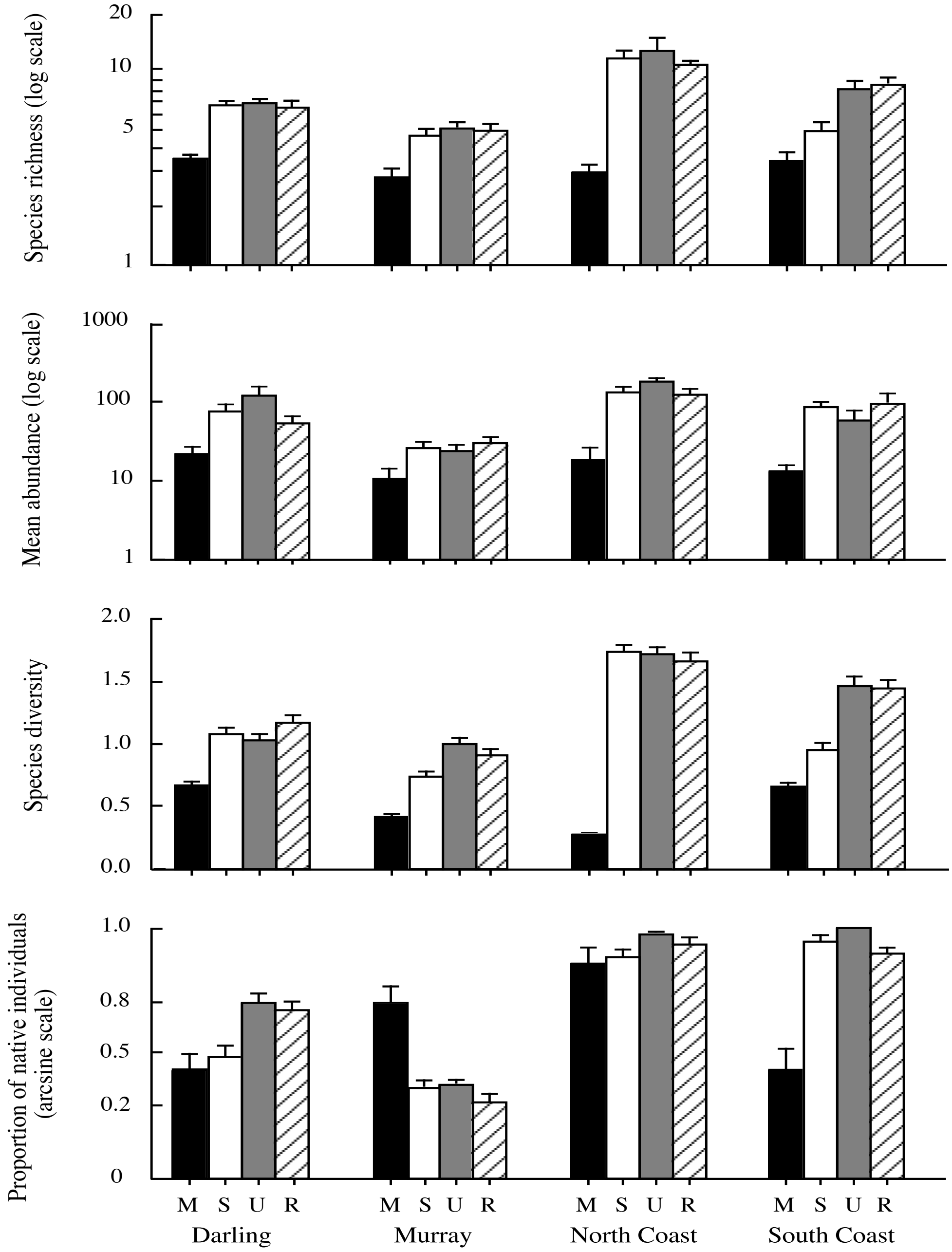

Figure 5.4 Interactions for fish community variables between geographic regions and river types (M= montane rivers, S = slopes rivers, U = unregulated lowland rivers, R = regulated lowland rivers).

Site classification (Figure 5.5) and ordination (Figure 5.6) revealed three primary groups representing fish communities from montane rivers, inland rivers, and coastal rivers. The divergence of coastal and inland fish faunas from the montane group represents an interaction in ordination space. This interaction indicates that fish communities at high altitude tend to be similar in all regions of New South Wales, but at lower altitudes, the regional character has a greater influence on the composition of fish communities than the river type in which the fish occur. Such a divergence is consistent with the degree of separation between rivers themselves, as most headwater streams originate at high altitude on the Great Dividing Range, and then flow either eastward in discrete catchments to the coast across a relatively narrow coastal plain, or westward across the vast Murray-Darling Basin, and converging to form a single river, the Murray, before continuing west and southward to the coast. Within the montane group, sites from the North Coast and South Coast regions formed distinct sub-groups, but montane sites from inland rivers displayed a degree of overlap between Darling and Murray regions. The inland group contained two major sub-groups representing rivers from the Murray and Darling regions, but with a large degree of overlap between regions. Rivers in the coastal regions displayed a clear separation between South Coast slopes and unregulated lowland rivers and remaining rivers, with most of the regulated rivers from the South Coast region grouping with rivers from the North Coast (Figure 5.6).

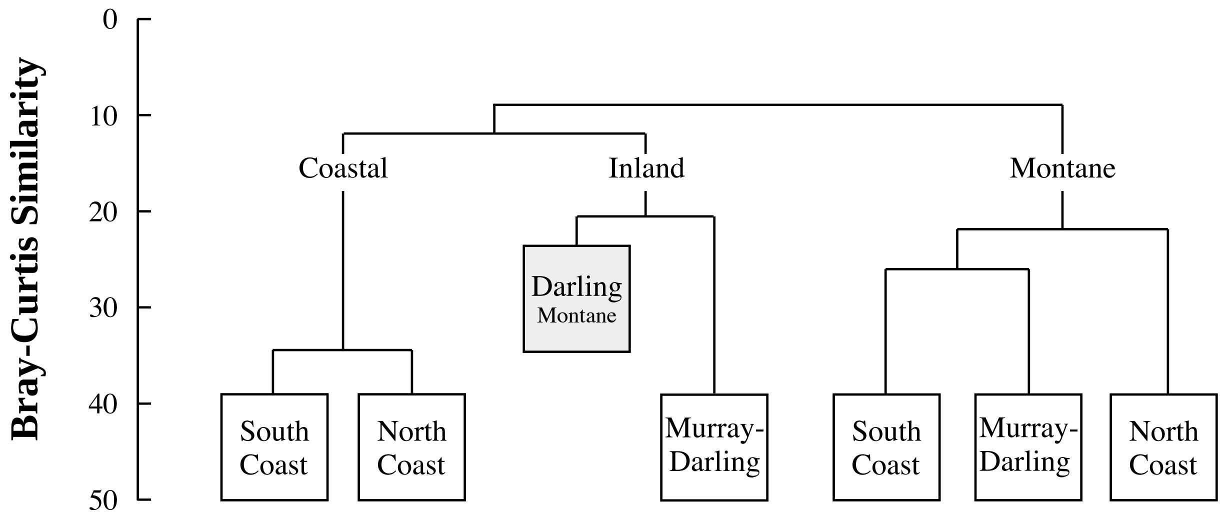

Figure 5.5 Simplified hierarchical agglomerative classification of river sites in New South Wales, based on similarities between fish communities. Three primary divisions are apparent at less than 20% similarity, representing coastal, inland and montane rivers. At greater than 30% similarity, lowland and slopes rivers in the South Coast and North Coast regions form separate groups. No separation of rivers in the Murray and Darling regions is apparent at this resolution. Montane rivers fell into three categories representing the North Coast, South Coast, and combined Murray-Darling regions. Two montane rivers in the Darling region showed greater similarity with lower altitude inland rivers than with other montane rivers.

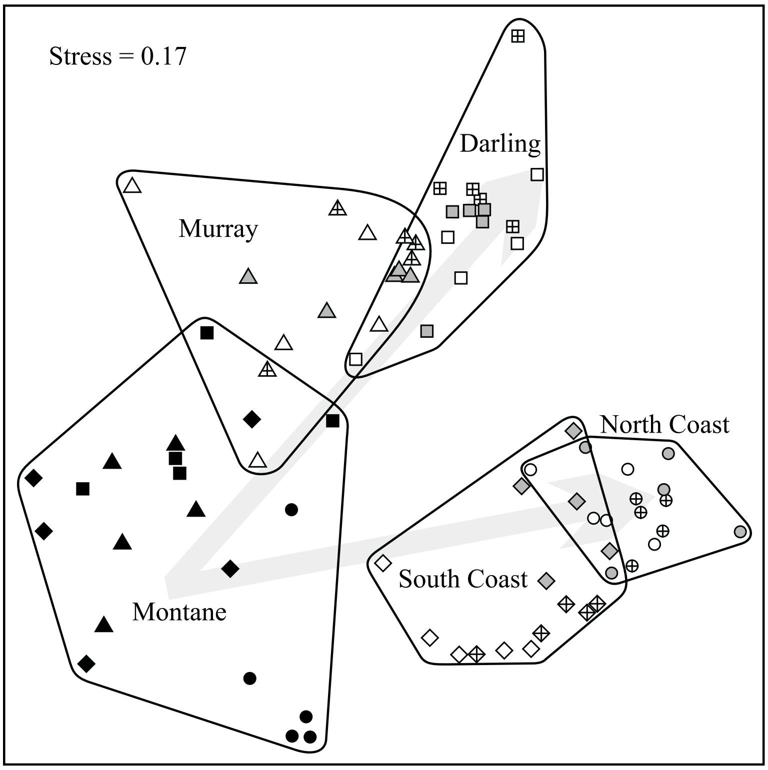

Figure 5.6 Two-dimensional MDS ordination of New South Wales rivers based on similarities between fish communities at each site. Fish communities in all regions show a high degree of similarity in montane river types and diverge to form characteristic coastal and inland communities. Different symbols show the regions in which sites occurred: ● North Coast; ◆ South Coast; ▲ Murray; ■ Darling. Fill patterns depict river types: ■ montane; □ slopes; ⊞ unregulated lowland; ■ regulated lowland.

Inverse classification and ordination of fish species complemented the site analyses by again revealing the existence of a montane species group, a diverse Murray-Darling group and, at a high similarity level, separate North Coast and South Coast species groups (Figure 5.7 and Figure 5.8). The divergence among these groups again indicates the existence of an interaction between river type and region in ordination space.

Two-way ANOSIM detected significant differences in the composition of fish communities among regions ($P<0.001$), with each region differing significantly from other regions ($P<0.001$) (Table 5.5). Similarly, fish communities differed among river types ($P<0.001$), with the largest differences occurring between the montane rivers and both regulated and unregulated lowland reaches. However, the interaction observed between river type and regions in the site ordination could not be quantified in the PRIMER implementation of two-way ANOSIM.

One-way ANOSIM comparisons among river types within each region (Table 5.5) identified only the comparison between unregulated and regulated rivers in the Murray region as non-significant ($P>0.05$).

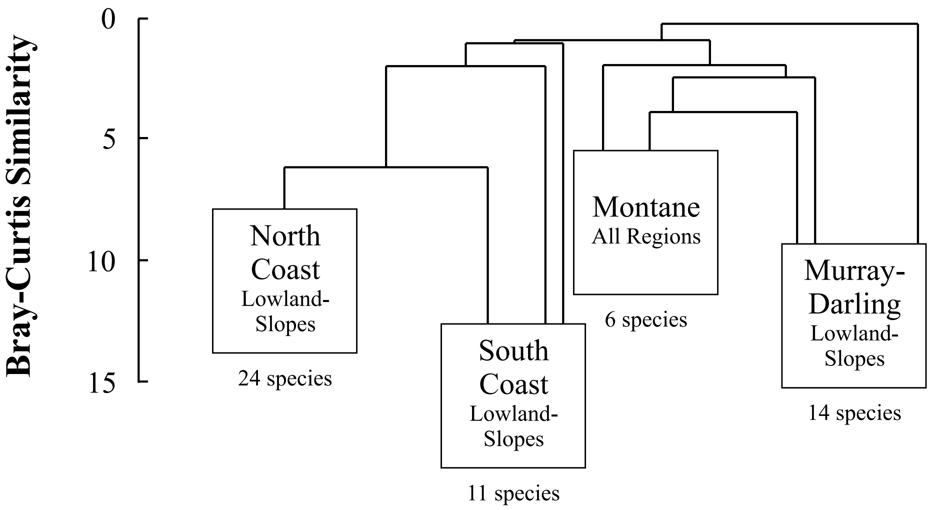

Figure 5.7 Simplified hierarchical agglomerative classification of fish species in New South Wales, based on similarities between species distributions among rivers. Six characteristically montane species form one group based upon the river type they inhabit, whereas the remaining species form associations based more upon geographic regions rather than characteristic distributions in slopes, unregulated or regulated lowland rivers. The Murray-Darling and South Coast groups are heterogeneous, with either one or two rare species with low similarity included in the groups shown.

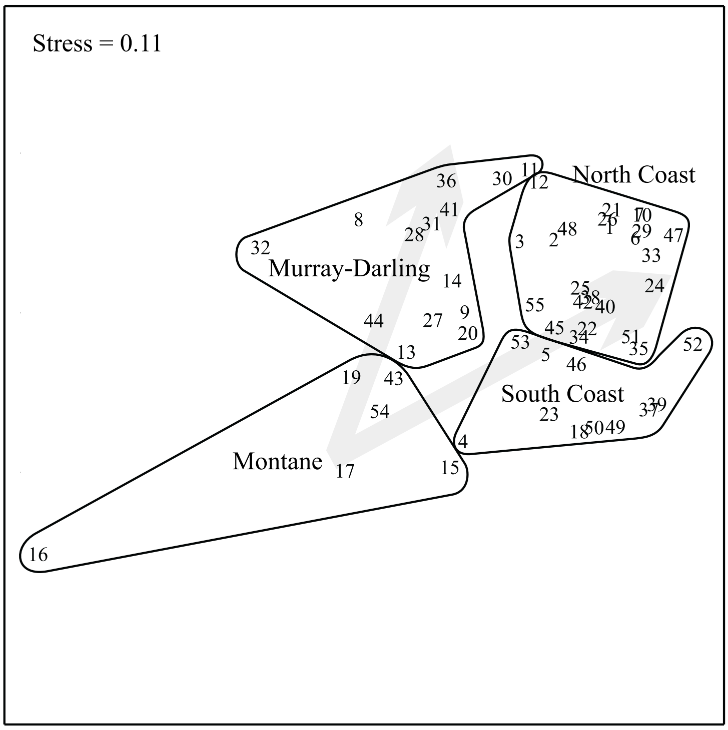

Figure 5.8 Two-dimensional MDS ordination of fish species in New South Wales rivers, based on similarities between species distributions among rivers, pooled over four sampling occasions. Four characteristic species groups are apparent, with one group of montane species which occur in all regions, which diverges to form a single inland group of Murray-Darling species, and two coastal groups of North Coast and South Coast species. Refer to Table 5.1 for key to species numbers.

Table 5.5 Summary ANOSIM results of two-way analysis with regions and river type as factors. Catches at each site were pooled over four times of sampling, providing five spatially-replicated sites in each cell of the design. 5000 permutations were used to estimate the probabilities of Type 1 error associated with each comparison. Comparisons among river types within each region were done as one-way analyses with no pooloing of catches over time.

Source		R	Probability
Among Regions		0.801	< 0.001
	Darling v Murray	0.462	< 0.001
	Darling v North Coast	0.888	< 0.001
	Darling v South Coast	0.844	< 0.001
	Murray v North Coast	0.928	< 0.001
	Murray v South Coast	0.935	< 0.001
	North Coast v South Coast	0.796	< 0.001
Among River Types		0.532	< 0.001
	Montane v Slopes	0.778	< 0.001
	Montane v Unregulated	0.930	< 0.001
	Montane v Regulated	0.935	< 0.001
	Slopes v Unregulated	0.241	< 0.001
	Slopes v Regulated	0.331	< 0.001
	Unregulated v Regulated	0.133	< 0.001
Within Darling Region Among River Types		0.474	< 0.001
	Montane v Slopes	0.522	< 0.001
	Montane v Unregulated	0.700	< 0.001
	Montane v Regulated	0.687	< 0.001
	Slopes v Unregulated	0.437	< 0.001
	Slopes v Regulated	0.308	< 0.001
	Unregulated v Regulated	0.124	0.003
Within Murray Region Among River Types		0.492	< 0.001
	Montane v Slopes	0.637	< 0.001
	Montane v Unregulated	0.898	< 0.001
	Montane v Regulated	0.896	< 0.001
	Slopes v Unregulated	0.196	< 0.001
	Slopes v Regulated	0.156	< 0.001
	Unregulated v Regulated	0.013	0.307 ns
Within North Coast Region Among River Types		0.577	< 0.001
	Montane v Slopes	0.988	< 0.001
	Montane v Unregulated	1.000	< 0.001
	Montane v Regulated	0.969	< 0.001
	Slopes v Unregulated	0.260	< 0.001
	Slopes v Regulated	0.089	0.021
	Unregulated v Regulated	0.115	0.006
Within South Coast Region Among River Types		0.629	< 0.001
	Montane v Slopes	0.820	< 0.001
	Montane v Unregulated	0.879	< 0.001
	Montane v Regulated	0.875	< 0.001
	Slopes v Unregulated	0.338	< 0.001
	Slopes v Regulated	0.768	< 0.001
	Unregulated v Regulated	0.272	< 0.001

Species contributing most to the dissimilarity between fish communities in the Darling and other regions were the *Hypseleotris* species complex, *Nematalosa erebi* and *Cyprinus carpio* which were all more abundant in the Darling region (Table 5.6). Differences between fish communities in the Murray and coastal regions were largely attributable to the greater abundance of *Anguilla* *reinhardtii* and *Retropinna semoni* in the North and South Coast rivers, and the higher abundance of *Cyprinus carpio* in the Murray region. Differences between fish communities in the North and South coast regions were mostly associated with the greater abundance of *Retropinna semoni* and *Hypseleotris compressa* in the North Coast region.

Species which contributed to typical fish communities in each river type were confounded by the interaction between river type and regions. Therefore, typical species for each river type were identified separately for each region (Table 5.7). *Galaxias olidus* was the most typical species in montane rivers, making a large contribution to the similarity within regional communities in the Darling, Murray and North Coast regions. Other dominant montane species varied among regions. Slopes reaches in the inland regions were dominated by *Cyprinus carpio*, whereas *Retropinna semoni* was the only consistently typical species in coastal slopes rivers. Unregulated lowland rivers in the Darling region shared the same dominant species as regulated rivers, being dominated by *Cyprinus carpio* and *Nematalosa erebi*, with *Macquaria ambigua* and *Hypseleotris* spp. also typical. In the Murray region, however, *Cyprinus carpio* still dominated lowland fish communities, but other typical species differed between unregulated and regulated river types. Fish communities in coastal lowland rivers were dominated by *Macquaria novemaculeata*, which consistently contributed to the similarity within river types in both regions, but was only the most dominant species in South Coast regulated rivers. Other species that were consistently co-dominant in regulated and unregulated rivers were *Myxus petardi* in the North Coast region and *Gobiomorphus australis* in the South Coast region. Species whose contributions to fish communitites are likely to be underestimated because of sampling bias are *Craterocephalus stercusmuscarum*, *Gambusia holbrooki*, *Hypseleotris galii*, *Mordacia praecox*, *Nematalosa erebi* and *Retropinna semoni*. However, this bias was not considered suffecient to change the dominant species in each river type or region. Typical dominant species in fish communities for each river type in all regions are summarised in Figure 5.9.

Table 5.6 Species contributing to differences in fish communities between regions, as determined by SIMPER analysis. Mean dissimilarity values indicate the magnitude of differences between communities in each region. Ratio values indicate the consistency of each species in discriminating between communities in each region, with larger ratios indicating greater consistency. The percent column indicates the average contribution of each species to differences between regions.

Species	Mean Abundance		Consistency	Percent	Cumulative
	Region 1	Region 2	ratio		%
	Darling	Murray		Mean dissimilarity = 70.3	
Hypseleotris spp	185.9	5.6	1.4	11.0	11.0
Nematalosa erebi	99.1	5.0	1.0	9.6	20.6
Cyprinus carpio	53.3	46.2	1.0	8.7	29.3
Gambusia holbrooki	31.7	1.5	1.0	7.7	37.0
Perca fluviatilis	13.2	6.6	1.3	7.0	44.0
Retropinna semoni	21.2	17.7	1.2	6.9	50.9
Galaxias olidus	8.1	14.5	0.8	6.8	57.7
Carassius auratus	13.6	4.9	1.1	6.1	63.9
Macquaria ambigua	9.6	1.8	1.2	5.6	69.5
	Darling	North Coast		Mean dissimilarity = 89.9	
Anguilla reinhardtii	0.0	65.9	1.9	8.2	8.2
Hypseleotris spp	185.9	5.8	1.2	6.5	14.8
Cyprinus carpio	53.3	4.2	1.2	5.8	20.5
Nematalosa erebi	99.1	0.0	0.9	5.2	25.7
Hypseleotris compressa	0.0	130.8	1.1	4.7	30.4
Gambusia holbrooki	31.7	32.4	0.8	4.6	35.0
Retropinna semoni	21.2	29.3	1.2	3.9	38.9
Myxus petardi	0.0	32.1	1.4	3.9	42.8
	Darling	South Coast		Mean dissimilarity = 90.4	
Hypseleotris spp	185.9	0.0	1.4	7.8	7.8
Anguilla reinhardtii	0.0	64.2	1.4	7.3	15.0
Cyprinus carpio	53.3	1.9	1.3	6.6	21.6
Nematalosa erebi	99.1	0.0	1.0	6.1	27.7
Retropinna semoni	21.2	58.2	1.4	5.9	33.6
Gambusia holbrooki	31.7	18.0	0.9	5.2	38.8
Gobiomorphus coxii	0.0	40.9	1.0	4.5	43.3
Carassius auratus	13.6	0.8	1.3	4.3	47.6
	Murray	North Coast		Mean dissimilarity = 91.3	
Anguilla reinhardtii	0.0	65.9	1.7	9.8	9.8
Cyprinus carpio	46.2	4.2	1.0	6.7	16.5
Hypseleotris compressa	0.0	130.8	1.1	5.1	21.6
Retropinna semoni	17.7	29.3	1.0	4.7	26.3
Galaxias olidus	14.5	13.0	0.7	4.6	30.9
Myxus petardi	0.0	32.1	1.4	4.2	35.1
Tandanus tandanus	0.0	24.4	1.4	4.2	39.4
Mugil cephalus	0.0	32.9	1.2	4.2	43.6
	Murray	South Coast		Mean dissimilarity = 87.1	
Anguilla reinhardtii	0.0	64.2	1.4	9.0	9.0
Cyprinus carpio	46.2	1.9	1.2	8.1	17.1
Retropinna semoni	17.7	58.2	1.2	7.2	24.3
Gobiomorphus coxii	0.0	40.9	1.0	5.5	29.8
Perca fluviatilis	6.6	1.3	1.1	5.0	34.8
	North Coast	South Coast		Mean dissimilarity = 72.8	
Retropinna semoni	29.3	58.2	1.0	5.9	5.9
Hypseleotris compressa	130.8	13.6	1.2	5.8	11.6
Anguilla reinhardtii	65.9	64.2	0.8	5.1	16.7
Macquaria novemaculeata	21.4	32.5	1.1	4.8	21.6
Gobiomorphus australis	28.8	20.0	1.2	4.7	26.2
Myxus petardi	32.1	5.9	1.3	4.6	30.9
Tandanus tandanus	24.4	0.1	1.4	4.6	35.4
Mugil cephalus	32.9	5.0	1.1	4.6	40.0

Table 5.7 Species contributing most to the similarity in fish communities within river types for each region, as determined by SIMPER analysis. Ratio values indicate the consistency with which each species contributes to communities. Larger ratios indicate greater consistency. The percentage contribution gives the average contribution of each species to the total similarity within each region and river type.

Region	River Type	Species	Consistency ratio	Contribution %
Darling	Montane	*Salmo trutta*	0.7	38.7
		Galaxias olidus	0.6	34.4
		Gambusia holbrooki	0.5	17.2
	Slopes	*Cyprinus carpio*	1.4	31.3
		Hypseleotris spp.	1.4	26.0
		Gambusia holbrooki	1.0	16.8
	Unregulated	*Cyprinus carpio*	3.6	29.1
	Lowland	*Nematalosa erebi*	1.6	27.7
		Macquaria ambigua	1.1	14.2
		Hypseleotris spp.	0.7	11.7
	Regulated	*Cyprinus carpio*	3.4	35.2
	Lowland	*Nematalosa erebi*	1.1	23.4
		Hypseleotris spp.	0.8	13.4
		Macquaria ambigua	0.7	9.7
Murray	Montane	*Galaxias olidus*	2.1	81.4
		Oncorhynchus mykiss	0.6	14.6
	Slopes	*Cyprinus carpio*	1.2	60.1
		Oncorhynchus mykiss	0.5	10.0
		Macquaria australasica	0.5	9.6
	Unregulated	*Cyprinus carpio*	2.9	57.4
	Lowland	*Macquaria ambigua*	0.6	11.7
		Perca fluviatilis	0.6	8.7
	Regulated	*Cyprinus carpio*	3.2	59.7
	Lowland	*Retropinna semoni*	0.6	12.5
		Perca fluviatilis	0.6	9.9
North Coast	Montane	*Anguilla reinhardtii*	2.3	82.5
		Galaxias olidus	0.6	15.6
	Slopes	*Anguilla reinhardtii*	4.8	23.0
		Tandanus tandanus	2.3	16.5
		Retropinna semoni	1.4	10.8
		Hypseleotris compressa	1.0	7.9
		Mugil cephalus	0.9	7.8
		Macquaria novemaculeata	0.9	7.7
	Unregulated	*Hypseleotris compressa*	1.8	14.7
	Lowland	*Gobiomorphus australis*	5.1	14.0
		Myxus petardi	3.9	13.6
		Macquaria novemaculeata	3.1	11.6
		Anguilla reinhardtii	1.3	9.6
	Regulated	*Anguilla reinhardtii*	2.0	17.7
	Lowland	*Tandanus tandanus*	2.3	16.2
		Macquaria novemaculeata	1.4	14.0
		Myxus petardi	1.0	9.3
South Coast	Montane	*Salmo trutta*	1.0	43.3
		Anguilla australis	0.8	40.4
	Slopes	*Anguilla reinhardtii*	2.1	44.2
		Retropinna semoni	1.4	26.4
		Gobiomorphus coxii	1.1	22.4
	Unregulated	*Anguilla reinhardtii*	3.2	26.7
	Lowland	*Retropinna semoni*	1.6	21.0
		Macquaria novemaculeata	0.9	9.9
		Gobiomorphus australis	0.8	9.7
		Galaxias maculatus	0.7	8.9
	Regulated	*Macquaria novemaculeata*	4.3	26.6
	Lowland	*Anguilla reinhardtii*	4.1	21.9
		Philypnodon grandiceps	1.2	13.2
		Gobiomorphus australis	0.8	8.5

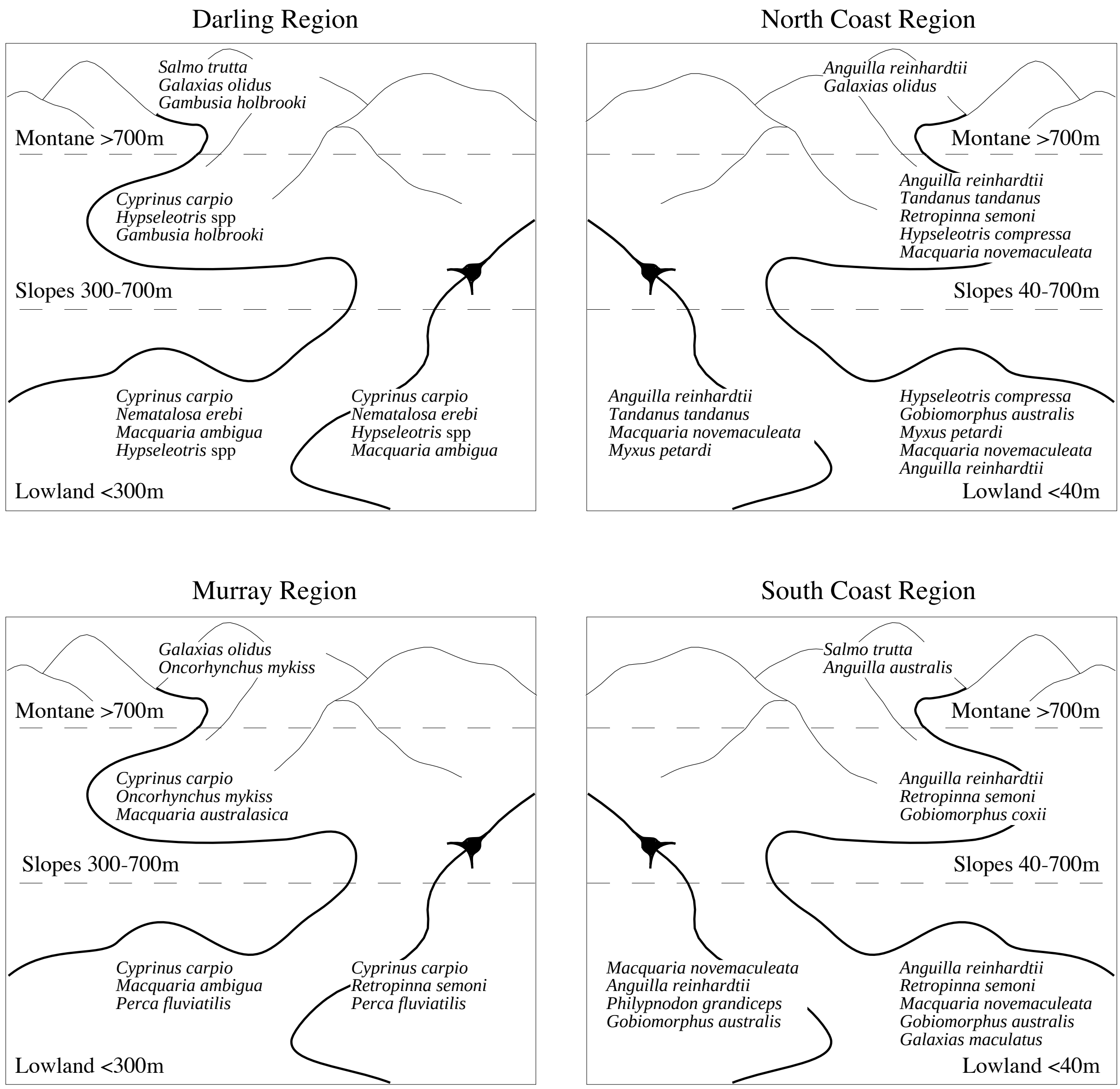

Figure 5.9 Typical species dominating fish communities in different river types in the four geographical regions in this study. Refer to Table 5.7 for details of the contribution of each species to type communities. regulated rivers are depicted by a dam at the base of the slopes reaches, while unregulated rivers are depicted by rivers without a dam.

DISCUSSION

It is generally considered that Australian rivers contain relatively few freshwater fish species, with a total of approximately 180 native species and 26 additional alien species (Merrick and Schmida 1984; Allen 1989; Paxton *et al.* 1989; McDowall 1996). This view is exemplified by comparisons with regions that possess a large number of fish species, such as the Amazon River, with 1500 to 2000 species. However, direct comparisons of species numbers alone fail to consider the role of habitat availability, which is related in some way to catchment area, stream length, gradient, channel morphology and flow (Lake 1982), latitudinal gradients and productivity of different rivers.

This study recorded a total of 55 species, with a minimum of two species per site and a maximum of 23 species. In relation to catchment area, the number of species collected in this study were: Darling - 20 species from 639,989 km^2; Murray - 20 species from 222,611 km^2; North Coast - 35 species from 85,418 km^2; and South Coast - 31 species from 65,552 km^2. These numbers are consistently below the global average number of fish species per unit catchment area (Welcomme 1985) (Table 5.8). However, global predictions of species richness are elevated by the large number of species in the Amazon and other tropical South American rivers (Welcomme 1985), and the rivers of Ontario (Eadie *et al.* 1986), and do not provide reliable estimates of fish species richness in temperate Mediterranean and arid climates that prevail in much of Australia. In comparison, predictive equations for fish species richness from rivers in Portugal, Southern Europe, Northern Europe and Greece provide estimates of the number of species per unit catchment area that are either similar to or less than the values recorded in New South Wales (Daget and Economidis 1975; Welcomme 1979; Welcomme 1985). Taking stream length and discharge into account, African rivers of similar size to the Murray-Darling River system would contain between 78 and 108 fish species (Livingston *et al.*, 1982), well below global estimates. In the arid interior basins in western North America, drainage area bears no relationship to the number of species (Moyle and Herbold, 1987). For example, the Colorado River contains only 36 native species for a basin area of 632,000 km^2 (Carlson and Muth, 1989), which is similar to rivers in the Darling region in this study with 31 recorded species and a catchment area of 639,989 km^2. These comparisons highlight some of the differences in species richness between tropical or northern temperate rivers and semi-arid river systems, such as the Murray-Darling. Thus the freshwater fish fauna of New South Wales, and the rest of Australia, rather than being impoverished, is reasonably typical for regions with temperate Mediterranean and arid climates. This study therefore supports the view that the fish fauna of Australian rivers is remarkably rich considering the aridity of much of the continent and the resultant scarcity of aquatic habitats.

Table 5.8 Catchment area and number of species in regions surveyed in this study, compared to estimated number of species from other world regions with an equivalent catchment area.

				Predicted No. Species							
NSW Region	Catchment Area (km^2)	No. Species (This study)	No. Species (recorded)	Africa[a]	South America[a]	Southern Europe[b]	Northern Europe[a]	Greece[c]	Portugal[c]	Ontario[d]	Global[a]
Darling	639989	20	29	149	271	79	35	57	23	205	175
Murray	222611	20	36	94	151	61	29	45	19	152	106
North Coast	85418	35	54	62	89	49	24	35	15	116	67
South Coast	65552	31	47	55	77	46	23	33	15	108	59
Total	1013570	55	82	181	349	88	38	64	25	233	218

Sources: [a] Welcomme (1985), [b] Welcomme (1979) - values for Southern Europe published in Welcomme (1985) appear to be in error, [c] Daget and Economidis (1975), [d] Eadie et al. (1986).

Within Australia, Pusey *et al.* (1993) recorded a total of 19 species of fish from a catchment area of 4,687 km^2 in the Mary River system in south-east Queensland. The Mulgrave and South Johnstone rivers, in the Wet Tropics region of North Queensland, represent extreme cases of high species richness for eastern Australia, with 36 species from a catchment area of only 970 km^2 for the Mulgrave River, and 25 species from a catchment of 640 km^2 (Pusey *et al.* 1995). These examples illustrate the variation in species richness associated with latitude and climate that occurs even within Australia, and emphasise the need for comparisons of species richness to be made over similar climatic regions in studies where the objective is to determine whether the fauna of a region is rich or impoverished.

Livingstone *et al.* (1982) reconciled what appeared to be vast differences between the fish faunas of the Nile River, which is 6,650 km long and contains 115 species, and the Zaire River, which is much shorter at 4,700 km yet contains 669 species. They found that while stream length, catchment area and mean annual discharge were all highly correlated, mean annual discharge accounted for over 72% of the variation in species richness among rivers, while stream length and catchment area did not contribute significantly to the model. They concluded on the basis of discharge that the Nile did not have a depauperate fish fauna, but rather contained the number of species that might be expected for a river of its size. They subsequently concluded that discharge and climatic stability during the Quaternary period were largely responsible for the number of fish species in African rivers.

Application of species-habitat relationships from other river systems to all four regions of this study assumes that stream lengths, areas and discharges are additive among rivers within regions. This is undoubtedly true for the Darling and Murray regions, where all rivers sampled drain into a single river. However, in coastal regions, which comprise a large number of discrete catchments separated by mountain ranges and the sea, the assumption that the expected number of species should equal that expected for a single catchment equal in area to the sum of the individual catchments, may not be valid. This assumption requires more detailed analysis of the relationship between stream discharge, habitat availability and the number of fish species in New South Wales' rivers. Estimates of species richness cannot be derived from rivers in other continents with different geological and climatic histories.

It is intuitive that the abundance and species richness of freshwater fish might increase with an increase in habitat availability (Lake 1982), and there have been many studies that demonstrate relationships between surrogate estimates of habitat availability and the number of fish species (eg. Kuehne 1962; Whiteside and McNatt 1972; Gorman and Karr 1978; Karr *et al.* 1986; Rahel and Hubert 1991; Paller 1994). The only similar study on Australian fish communities was that of Lake (1982) who used power curves to describe the relationship between the number of fish species, and catchment area and stream length in small streams in south-eastern Australia. However, Lake's data covered only small streams and cannot be extrapolated to the large areas for each region in the present study.

Many of these studies have found that the downstream increase in species richness occurs mostly through the addition of new species, rather than by replacement of species that only occur in more upstream reaches (eg. Sheldon 1968; Hocutt and Stauffer 1975; Evans and Noble 1979; Lake 1982; Beecher *et al.* 1988; Rahel and Hubert 1991; Paller 1994). However, faunal zonation may impose discontinuities in patterns of species addition such that replacement takes a greater role between zones (Rahel and Hubert 1991; Paller 1994). This study has demonstrated three faunal zones operating over a large spatial scale in each region. These zones correspond to the montane, slopes and lowland river types recognised in the stratified process used for site selection. Within the montane and slopes reaches of all four regions in New South Wales, species addition occurs rapidly with increasing living space in the river. However, in the lowland reaches of inland regions, species replacement occurs, eventually leading to a net loss of species as typical slopes species disappear without being replaced. In coastal regions, species addition continues in lowland reaches, but at a slower rate than in higher altitude rivers.

Basic theories of stream fish ecology suggest that downstream addition of species occurs as a result of increased living space in larger streams, increased habitat diversity such as access to floodplain habitats and backwaters, and greater habitat stability such as reduced flow variability (Horwitz 1978; Schlosser 1987; Rahel and Hubert 1991). The interpretation of longitudinal patterns in fish distributions may be confused at large spatial scales where both zonation and addition occur. For instance, a downstream increase in species richness, as in the coastal regions of the present study, supports the basic theory and suggests that living space increases at lower altitudes. However, species reduction in lowland reaches in inland regions of New South Wales suggests that there may be less, not more, living space in these reaches. Reduced habitat availability may be a natural feature of lowland rivers with limited tributary inflows in semi-arid regions where evaporation rates are high. Certainly, many inland rivers in New South Wales terminate in shallow lakes and emphemeral channels which do not offer permanent habitats for fish. Rahel and Hubert (1991) found a natural decrease in habitat diversity downstream in a Wyoming stream. However, other explanations invoking extensive degradation of inland river systems (Gehrke *et al.* 1995; Mallen-Cooper 1993) and reduced habitat diversity caused by alienation of floodplain habitats (Gehrke 1992) appear more likely in New South Wales. Sheldon and Walker (1997) showed a decline in populations of gastropods in the lower Murray River that they attributed to a change in the composition of littoral biofilms from microbial periphyton before river regulation, to filamentous algae under stable, regulated flow conditions. Whilst Sheldon and Walker (1997) examined sites downstream of the sites selected for the present study, a shift in composition of lower trophic levels may partly account for the loss of fish species in inland lowland rivers in New South Wales. Carp are able to assimilate energy at lower trophic levels than large-bodied, piscivorous native species such as Murray cod and golden perch. Additionally, they can attain large population densities (see Chapter 9) and grow rapidly (Brumley 1996) to large sizes that reduce the risk of predation. The dominance of carp in lowland rivers may therefore limit energy transfer to higher trophic levels, which in turn is likely to contribute to the decline in species richness with increasing distance downstream.

Fish distributions in Australia are determined to a large degree by the boundaries of major river drainage divisions, thus fish communities of New South Wales contain elements from the South-east Coastal, Murray-Darling, Bullo-Bancannia and Lake Eyre drainage basins. As the New South Wales sections of the latter two drainage basins are highly ephemeral, they were not included in this study. To improve resolution of geographical patterns in species distributions within New South Wales, the South-east Coastal and Murray-Darling drainage divisions were sub-divided for this study into North Coast and South Coast, and Darling and Murray regions. Multivariate analysis of fish community composition has supported the separation of these regions within New South Wales, although previous analyses (Llewellyn 1983) have not recognised the clear distinction between fish communities in montane rivers and those at lower altitudes. Thus habitat characteristics associated with high altitudes have a stronger influence on the composition of montane fish communities in New South Wales than the drainage system in which the river occurs. These results expand upon recent studies by Gehrke *et al.* (1995), who found significant differences between fish communities in lowland reaches of the Darling and Murray river catchments, but no differences between the Murray and Murrumbidgee rivers which were treated as a single region in the present study.

The dominant species in montane fish communities are brown trout, rainbow trout, gambusia, mountain galaxias and long-finned eels. The occurrence of both trout species reflects the degree to which these fish have been stocked into high-altitude streams in all regions. While the distribution of these alien species may imply a degree of artificiality to the montane fish community, both species have been implicated in the decline of native montane species such as mountain galaxias, and spotted galaxias, *Galaxias truttaceus*, (Fletcher 1979; Jackson and Williams 1980; Koehn and O’Connor 1990; Ault and White 1994). Thus the recognition of a distinct montane fish community is fully justified on the grounds that if trout were absent, the high-altitude fish fauna would be more strongly dominated by mountain galaxias.

The dominant fish species in typical communities display a gradient in trophic diversity in each region and river type (Table 5.9), with an increasing number of trophic guilds, as defined by Harris (1995), occuring at greater distances downstream. Montane communities have the simplest trophic base, being dominated by microphagic and macrophagic carnivores, and by macrophagic carnivores alone in the South Coast region. Slopes reaches possess a more diverse trophic range, with the inclusion of both microphagic and macrophagic omnivores among the dominant species. Only in the low gradient lowland rivers of the Darling region do herbivores and detritivores rank among the dominant species. Thus it can be seen that the number of trophic guilds within fish communities in New South Wales rivers increases with decreasing altitude, with the Darling region having the greatest trophic diversity among the dominant species, and the South Coast rivers having the least trophically diverse dominant species. This result is similar to the findings of other studies (e.g. Schlosser 1987; Paller 1994) in North American rivers. The low trophic diversity in South Coast lowland rivers reflects the low abundance or absence of carp, empire gudgeons, freshwater mullet and bony herring that constitute the macrophagic omnivores, microphagic omnivores and herbivores or detritivores among the dominant species in other regions.

Table 5.9 Distribution of trophic guilds represented by dominant species among regions and river types. Guilds defined after Harris (1995): 1 = herbivores/detritivores; 2 = microphagic omnivores; 3 = macrophagic omnivores; 4 = microphagic carnivores; 5 = macrophagic carnivores. In the Darling region, *Hypseleotris* spp. is shown as 2/4 because this species complex contains both microphagic omnivores and microphagic carnivores.

	Darling	Murray	North Coast	South Coast	**Total**
Montane	4,5	4,5	4,5	5	4,5
Slopes	2,3,4	3,4,5	2,4,5	4,5	2,3,4,5
Unregulated Lowland	1,2/4,3,5	3,5	2,4,5	4,5	1,2,3,4,5
Regulated Lowland	1,3,2/4,5	3,4,5	2,4,5	4,5	1,2,3,4,5
Total	1,2,3,4,5	3,4,5	2,4,5	4,5	1,2,3,4,5

The recognition of distinct communities in the riverine freshwater fish fauna of New South Wales raises the question of the degree to which communities can be considered as the appropriate units for fisheries management. Traditional fisheries management has adopted individual species as the units to be managed for the purposes of ensuring sustainable production over time from exploited fish populations (Edwards and Megrey 1989; Hilborn and Walters 1992). From time to time this approach is broadened to include multiple species that may be targeted by a single fishery (e.g. Pope 1989) but, almost by definition, fisheries management focuses on managing target species and the various human roles that combine to create a fishing industry. This approach leaves unexploited species out of the equation of sustainability, and ignores the important ecological interactions which change when the abundance or population dynamics of a target species are modified by harvesting. In marine fish communities, which may consist of many species with complex trophic interactions, modification of the population dynamics of a single species may have little effect on the community at large. However, in simpler freshwater communities that may consist of only eight species - as occurs at some sites in the present study - of which the only two macrophagic carnivores such as golden perch and Murray cod are targeted by a fishery, depletion of both species may result in effective removal of an entire trophic level. This destabilises the fish community to such an extent that the fishery can in no way be considered sustainable.

Evans *et al.* (1987) observed that freshwater fish communities can be managed as discrete functional units, but noted that the dependence of communities on broader ecosystem dynamics required an ecosystem perspective. The knowledge requirements for the community approach are challenging, and include food web structure, ontogenetic histories, resource partitioning and body-size dependency of interactions between species (Evans *et al.* 1987), and better understanding of complexity, spatial and temporal variability, succession, energy transfer through aquatic ecosystems and better ways of estimating biomass distributions in fish communities. Since

1987, substantial gains have been made in the understanding of these issues, especially through whole-lake and bioenergetic studies in the Northern Hemisphere (e.g. Havens 1993; Dettmers *et al*. 1996; Werner *et al.* 1996). The relative simplicity of freshwater fish communities in New South Wales makes these knowledge requirements more attainable than in more complex systems with greater numbers of species, and presents an opportunity to test the concept of managing fish communities as opposed to managing species. Management of riverine fish communities is much more attuned to the need for an ecosystem focus to catchment management.

Implications for management

Despite consisting of only a small number of species, the freshwater fish fauna of New South Wales is typical of the faunas of similar climatic regions on a global scale. The New South Wales fish fauna can be classified into Montane, North Coast, South Coast and Murray-Darling communities, while there is some biological justification for managing the Darling River and its tributaries as separate entities from rivers in the Murray region. The decline in species richness with increasing distance downstream in lowland reaches of inland rivers is contrary to general longitudinal patterns of species richness in rivers, and suggests a need for remedial management of lowland rivers to increase habitat diversity. The fish communities identified in this study form logical entities for fisheries management, and provide an opportunity to manage fisheries to ensure the sustainability of riverine ecosystems rather than sustaining single species.

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REFERENCES

Allen, G.R. (1989). *Freshwater Fishes of Australia*. (T.F.H. Publications: Sydney.) 240 pp.

Ault, T.R. and White, R.W.G. (1994). Effects of habitat structure and the presence of brown trout on the population density of *Galaxias truttaceus*. *Transactions of the American Fisheries Society* **123,** 939-949.

Beecher, H.A., Dott, E.R. and Ferneau, R.F. (1988). Fish species richness and stream order in Washington State streams. *Environmental Biology of Fishes* **22,** 193-209.

Bray, J.R. and Curtis, J.T. (1957). An ordination of the upland forest communities of Southern Wisconsin. *Ecological Monographs* **27,** 325-349.

Brumley, A.R. (1996). Family Cyprinidae. In Freshwater *Fishes of South-eastern Australia* (Ed. R.M. McDowall.) (2nd edn). (Reed: Sydney.) pp. 99- 106.

Carlson, C.A. and Muth, R.T. (1989). The Colorado River: lifeline of the American southwest. In 'Proceedings of the International Large Rivers Symposium'. (Ed. D.P Dodge.) *Canadian Special Publication of Fisheries and Aquatic Sciences*. **106,** 220-239.

Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18,** 117-143.

Daget, J. and Economidis, P.S. (1975). Richnesse spécifique de l'ichthyofaune de Macédione oriental et de Thrace occidentale (Gréce). *Bulletin du Museum National d'Historoire Naturelle Ecologie Generale* **27,** 81-84.

Dettmers, J.M., DeVries, D.R. and Stein, R.A. (1996). Quantifying responses to hybrid striped bass predation across multiple trophic levels: implications for reservoir biomanipulation. *Transactions of the American Fisheries Society* **125,** 491-504.

Eadie, J.McA., Hurly, T.A., Montgomerie, R.D. and Teather, K.L. (1986). Lakes and rivers as islands: species-area relationships in the fish faunas of Ontario. *Environmental Biology of Fishes*, **15,** 81-89.

Edwards, E.F. and Megrey, B.A. (Eds) (1989). 'Mathematical Analysis of Fish Stock Dynamics'. *American Fisheries Society Symposium* **6,** (American Fisheries Society: Bethesda)214 pp.

Evans, D.O., Henderson, B.A., Bax, N.J., Marshall, T.R., Oglesby, R.T. and Christie, W.J. (1987). Concepts and methods of community ecology applied to freshwater fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences* **44,** (Supplement 2) 448-470.

Evans, J.W. and Noble, R.L. (1979). The longitudinal distribution of fishes in an East Texas stream. *American Midland Naturalist* **101,** 333-343.

Fletcher, A.R. (1979). Effects of *Salmo trutta* on *Galaxias olidus* and macroinvertebrates in stream communities. unpublished M.Sc. thesis, Monash University, Clayton 179 pp.

Gehrke, P.C., Brown, P. Schiller, C.B., Moffatt, D.B and Bruce, A.M. (1995). River regulation and fish communities in the Murray-Darling River system, Australia. *Regulated Rivers: Research & Management* **11,** 363-375.

Gehrke, P.C. (1992). Enhancing recruitment of native fish in inland environments by accessing alienated floodplain habitats. In 'Recruitment Processes'. (Ed. D.A. Hancock.) pp. 205-209. Australian Society for Fish Biology Workshop, Hobart, 21 August 1991, Bureau of Rural Resources Proceedings No.16, (AGPS: Canberra.)

Gorman, O.T. and Karr, J.R. (1978). Habitat structure and stream fish communities. *Ecology* **59,** 507-515.

Harris, J.H. (1984). Australian freshwater fish. In: 'Vertebrate Zoogeography and Evolution in Australasia'. (Eds M. Archer and G. Clayton.) pp. 211-218. (Hesperian Press: Perth).

Harris, J.H. (1995). The use of fish in ecological assessments. *Australian Journal of Ecology* **20,** 65-80.

Havens, K.E. (1993). Responses to experimental fish manipulations in a shallow, hypereutrophic lake: the relative importance of benthic nutrient recycling and trophic cascade. *Hydrobiologia* **254,** 73-80.

Hilborn, R. and Walters, C.J (1992). 'Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty'. (Chapman and Hall: New York and London.) 570 pp.

Hocutt, C.H. and Stauffer, J.R. (1975). Influence of gradient on the distribution of fishes in Conowingo Creek, Maryland and Pennsylvania. *Chesapeake Science* **16,** 143-147.

Horwitz, R.J. (1978). Temporal variability patterns and the distributional patterns of streamfishes. *Ecological Monographs* **48,** 307-321.

Jackson, P.D. and Williams, W.D. (1980). Effects of brown trout, *Salmo trutta* L., on the distribution of some native fishes in three areas of southern Victoria. *Australian Journal of Marine and Freshwater Research* **31,** 61-67.

Karr, J.R., Fausch, K.D., Angermeier, P.L. Yant, P.R. and Schlosser, I.J. (1986). Assessing biological integrity in running waters, a method and its rationale. *Illinois Natural History Survey Special Publication 5.*

Koehn, J.D. and O'Connor, W.G. (1990). *Biological Information for Management of Native Freshwater Fish in Victoria*. (Government Printer: Melbourne.) 65 pp.

Kuehne, R.A. (1968). A classification of streams, illustrated by fish distribution in an eastern Kentucky creek. *Ecology* **43,** 608-614.

Lake, P.S. (1982). The relationship between freshwater fish distribution, stream drainage area and stream length in some streams of south-east Australia. *Bulletin of the Australian Society for Limnology* **8,** 31-37.

Lake, P.S. (1995). Of floods and droughts: river and stream ecosystems of Australia. In: 'River and Stream Ecosystems'. (Eds C.E. Cushing, K.W. Cummins and G.W. Minshall.) pp. 659-694. *Ecosystems of the World* **22**. (Elsevier: Amsterdam.)

Llewellyn, L.C. (1985). The distribution of fish in New South Wales. *Australian Society for Limnology Special Publication No. 7*, 23 pp.

Livingstone, D.A., Rowland, M. and Bailey, P.E. (1982). On the size of African fish faunas. *American Zoologist* **22,** 361-369.

Mallen-Cooper, M. (1993). Habitat changes and declines of freshwater fish in Australia: what is the evidence and do we need more? In *Sustainable Fisheries through Sustaining Fish Habitat*. (Ed. D.A. Hancock.) pp. 118-123. Australian Society for Fish Biology Workshop, Victor Harbor SA 12-13 August 1992, Bureau of Resource Sciences Proceedings, (AGPS: Canberra.)

McDowall, R.M. (Ed.) (1996). *Freshwater Fishes of South-eastern Australia*. (2nd edn) (Reed: Sydney.) 247 pp.

Merrick, J.R. and Schmida, G.E. (1984). '*Australian Freshwater Fishes: Biology and Management'*. (John R. Merrick: North Ryde.) 409 pp.

Moyle, P.B. and Herbold, B. (1987). Life-history patterns and community structure in stream fishes of western North America: comparisons with eastern North America and Europe. In *Community and Evolutionary Ecology of North American Stream Fishes*. (Eds W.J. Matthews and D.C. Heins.) pp. 25-32. (University of Oklahoma Press: Norman.)

Paller, M.H. (1994). Relationships between fish assemblage structure and stream order in South Carolina coastal plain streams. *Transactions of the American Fisheries Society* **123,** 150-161.

Paxton, J.R., Hoese, D.F., Allen, G.R., Hanley, J.E. (1989). 'Zoological Catalogue of Australia', Volume 7, Pisces (Petromyzontidae to Carangidae). (Australian Government Publishing Service: Canberra.) 665 pp.

Pope, J.G. (1989). Multispecies extensions to age-structured assessment models. *American Fisheries Society Symposium* **6,** 102-111.

Pusey, B.J., Arthington, A.H. and Read, M.G. (1993). Spatial and temporal variation in fish assemblage structure in the Mary River, south-eastern Queensland: the influence of habitat structure. *Environmental Biology of Fishes* **37,** 355-380.

Pusey, B.J., Arthington, A.H. and Read, M.G. (1995). Species richness and spatial in fish assemblage structure in two rivers of the Wet Tropics of northern Queensland, Australia. *Environmental Biology of Fishes* **42,** 181-199.

Rahel, F.J. and Hubert, W.A. (1991). Fish assemblages and habitat gradients in a Rocky Mountain-Great Plains stream: biotic zonation and additive patterns of community change. *Transactions of the American Fisheries Society* **120,** 319-332.

Schlosser, I.J. (1987). A conceptual framework for fish communities in small warmwater streams. In *Community and Evolutionary Ecology of North American Stream Fishes*. (Eds W.J. Matthews and D.C. Heins.) pp. 12-24. (University of Oklahoma Press: Norman.)

Sheldon, A.L. (1968). Species diversity and longitudinal succession in stream fishes. *Ecology* **49,** 193-198.

Sheldon, F. and Walker, K.F. (1997). Changes in biofilms, induced by flow regulation, may explain the extinction of gastropods in the lower River Murray, Australia. *Hydrobiologia* [in press].

Shepherd, J.G. (1982). A versatile new stock-recruitment relationship for fisheries, and the construction of sustainable yield curves. *Journal du Conseil International pour l'Exploration de la Mer* **40,** 67-75.

Welcomme, R.L. (1979). *Fisheries Ecology of Floodplain Rivers*. (Longman: London and New York.) 317 pp.

Welcomme, R.L. (1985). River fisheries. *FAO Fisheries Technical Paper 262*, 330 pp.

Werner, R.G., Jonckheere, B.V., Clapsadl, M.D. and Farrell, J.M. (1996). A bioenergetic exploration of piscivory and planktivory during the early life history of two species of freshwater fishes. *Marine and Freshwater Research* **47,** 113-121.

Whiteside, B.G. and McNatt, R.M. (1972). Fish species diversity in relation to stream order and physiochemical conditions in the Plum Creek drainage basin. *American Midland Naturalist* **88,** 90-101.

6 Assessing the condition of rivers in New South Wales, Australia: A test of the index of biotic integrity

J. H. Harris and R. Silveira

Cooperative Research Centre for Freshwater Ecology, NSW Fisheries Research Institute, PO Box 21 Cronulla, New South Wales 2230.

Summary

Rivers in Australia have suffered extensively from the effects of catchment degradation, regulation for water supply, pollution and alien species. Efforts to manage these factors and to restore riverine ecosystems need effective tools for assessing river condition and for adaptive management. Ways are needed to measure the ‘health’ of rivers. This implies a need for a comprehensive, sensitive and quantitative indicator for measuring the condition of the complex of variables which constitutes river health. To be effective, this indicator needs to be ecologically based, efficient and rapid. It needs to be applicable consistently in different ecological regions. It should also preserve information about the nature of any environmental changes which may alter the value of the indicator for a river reach in response to some environmental change.

The NSW Rivers Survey provided data for an assessment of the performance of the Index of Biotic Integrity (IBI) as a river-health indicator. The IBI predicts the fish-community attributes of a river reach of excellent environmental quality from regional and river-size data, using metrics of species richness, abundance, community structure, and the health of individual fish. Following a provisional index, IBI metrics were modified to suit the freshwater fish of south-eastern Australia. The IBI was able to discriminate relative environmental quality within a diverse set of stream systems and presumptive ecological regions. The 80 sites of the NSW Rivers Survey were spread through the full range of qualitative IBI rankings from ‘Excellent’ to ‘Very poor’ and ‘No fish’. Results supported by habitat data show that the Murray region’s rivers and a large proportion of montane rivers in New South Wales coastal regions are in a degraded condition compared to other regions and river types. The index’s underlying assumptions were generally met and it performed satisfactorily with 12 metrics based on attributes of the fish fauna totalling 55 freshwater species. One metric, based on trophic guild, performed poorly and should be deleted from the index, leaving an 11-metric IBI. Six other recommendations are made to enhance the performance of the IBI as a rapid, efficient and sensitive ecological tool for monitoring river health. Results supported the designation of four ecological regions (Murray, Darling, South Coast and North Coast) for New South Wales freshwater fauna, and they provide a useful basis for management. Future work is recommended to complete validation of the IBI including comparisons with independent measures of river and catchment health, parallel assessments with other river-health indicators, studies of IBI responsiveness to known impacts, and tests of repeatability.

The IBI produces relative assessments and give the capacity to assess spatial and temporal changes in the relative health of rivers. The main immediate values of the present IBI results lie in their capacity to provide a baseline for monitoring river health, and an efficient and sensitive means of checking the relative condition of any river site in New South Wales.

INTRODUCTION

Rivers in Australia have suffered extensively from the effects of regulation for water supply, catchment degradation, pollution and alien species. Efforts to manage these factors and to restore riverine ecosystems need effective tools for assessing river condition and for future adaptive management. Ways are needed to measure the 'health' of rivers, despite the present lack of a clear, generally accepted interpretation of this concept.

We suggest that river health can be defined as the degree to which a river's aquatic and riparian biota and their habitats (including structural components, water quality and flows), and its ecological processes at all scales, match the natural condition. This definition of river health implies a need for a comprehensive, sensitive and quantitative indicator for integrating and assessing the condition of each of the five components, one which can indicate the cause of any changes. To be effective, such an indicator needs to be ecologically based (to support an ecosystem approach), efficient (because of the need to assess many river reaches with limited resources) and rapid (so that assessment can proceed quickly for many rivers of concern). It should also preserve information about the nature of any environmental changes which may alter the indicator value, rather than combining separate bits of information in some difficult-to-decipher index. A useful river-health indicator must also be consistently applicable in ecologically different regions so that meaningful assessments can be made, despite differences in the biota of the regions. In New South Wales it is necessary to be able to interpret river health in a consistent way in the coastal drainages and the Murray-Darling basin; in the north of the state, as well as the south. There must also be some way in which the indicator can be responsive to river size so that any given river reach can be assessed, from those in alpine areas to those near the tidal limit.

Furthermore, a river-health indicator needs to show the condition of ecosystems rather than narrowly defined components of ecosystems. 'Water quality' has tended to be used in the literature synonymously for the much larger concept of aquatic environmental quality (Cranston *et al.* 1996; Norris and Norris 1995), and this unrealistically narrow focus has done little to resolve many problems of river management. CSIRO (1992) listed eight direct causes of change to rivers: manipulating stream channels, damming watercourses, manipulating streamflow, draining wetlands, transferring water to urban and industrial consumers, disposing of waste, extracting groundwater and irrigating agricultural land. Among these eight causes there is only one impact (waste disposal) that is a direct 'water-quality' effect. The remainder are all impacts on the ‘structural’ qualities of rivers. (Surprisingly, stream siltation was not listed among the direct disturbances, although it is among the most widespread processes degrading Australian rivers.) Preoccupation with issues of chemical pollution and declining water quality has tended to divert attention from the structural and biotic problems damaging the overall quality of aquatic environments. Structural problems can be broadly classed as alteration of streamflow regimes, loss of habitat area for aquatic biota, loss of habitat diversity, obstruction of the physical connections of stream systems, interrupted nutrient flow and riparian degradation. These changes are at least as significant as water-quality effects.

Biotic indicators are needed to help overcome the narrow focus and poor time-integrating capacity of most chemical and physical monitoring tools. They are also needed for comprehensively assessing the condition of all of the components of aquatic ecosystems. The Index of Biotic Integrity (IBI) was designed to meet these needs (Karr 1981, 1987) and its use of a complementary series of metrics to assess biological integrity as an indicator makes the IBI sensitive to a variety of changes in environmental conditions. The IBI predicts biotic attributes of a river reach from regional and river-size data, using metrics of species richness, abundance, community structure and, when used with fish, the health of individual animals (Karr 1981, 1987; Fausch *et al.* 1984, 1990; Karr *et al.* 1986). These metrics are designed to assess biotic integrity as an indicator of relative levels of environmental quality in river reaches on a consistent, standardised basis within ecological regions. There is no assumption of independence among the metrics. From survey data, values are derived for individual metrics which are summed to provide the IBI score. The score is usually reported as a qualitative rank of biotic integrity indicating environmental quality, with levels of ‘Excellent’, ‘Good’, ‘Fair’, ‘Poor’, and ‘Very Poor’. When used with fish-community data, the IBI provides assessments of aquatic environmental quality at large spatial (river reach) and temporal (season) scales, as well as functioning at much smaller scales. A preliminary index has been proposed (Harris 1995) in which 12 basic IBI metrics were developed to suit the fish communities of south-eastern Australia. This fauna is considerably less diverse and ecologically specialised than the North American fauna which provided the origins of the IBI.

Assessing biotic integrity provides a means of indirectly assessing the degree to which a river’s aquatic and riparian biota, habitats and ecological processes match the natural condition. Ideally, undisturbed stream reference sites would provide a standard against which all other comparable sites of the particular region could be directly assessed. But this ideal is no longer possible in Australia. The great majority of rivers have been affected by hydrological disturbances, channel changes, siltation from catchment degradation, chemical and thermal pollution, alien species, desnagging and/or riparian damage (CSIRO 1992; Australia State of the Environment 1996; Roberts and Sainty 1996). In New South Wales, some of the headwaters rivers in national parks may be virtually undisturbed, but it is impossible to identify more than a few undisturbed lowland or slopes-zone rivers that could serve as standards for comparative measurement in coastal drainages, and none remain in the Murray-Darling basin. Because there are no such ecological standards, some different method is required, one which can be used indirectly to predict levels of the relevant biotic attributes that might otherwise have been expected at a particular site or reach in its natural condition. The IBI is an attempt to overcome this obstacle and to enable prediction of natural conditions to be made by first defining the important elements of biotic integrity. These elements, such as abundance of individual animals, their species-richness and condition, the level of intrusion of alien species, and the representation in the biotic community of relevant trophic and habitat guilds, provide the basis for metrics (Harris 1995). Actual values for metrics at an 'Excellent' reach, *i.e., 'one which is comparable to the best situation without disturbance'* (Karr 1981), are predicted from the highest values measured in extensive surveys of the ecological region, and standardised for river size. In this way the IBI integrates the best remnant examples of each main element of biotic integrity to compensate for the unavailability of undisturbed reference sites across the spectrum of habitat types. It provides an approximate mechanism (albeit with some untested assumptions about undisturbed community structure) for predicting natural conditions so that deviations from nature can be estimated.

The structure of stream biotic communities is controlled by the flow regime, energy source, water quality, biotic interactions and habitat structure, and there are predictable relationships between Australian fish-community attributes and a wide range of stream-habitat variables (Gehrke *et al.* 1995*;* Harris 1995; Lake 1982). An IBI based on fish has the potential to integrate and reflect all these ecosystem attributes at the river-reach spatial scale and seasonal temporal scale, as well as at smaller scales. The NSW Rivers Survey has provided data for a test of the performance of the IBI in assessing river health. An important part of the test of the IBI for New South Wales streams concerned its ability to discriminate environmental quality using assessments of the biotic integrity of a relatively small fauna, totalling 55 freshwater species (Chapter 2). The objective of this work was therefore to apply the IBI with Rivers Survey data, and to examine its success in discriminating the relative environmental quality of a diverse set of rivers on a consistent, standardised basis within presumptive ecological regions. Future work will test the validity of IBI assessments using comparisons with independent indicators of catchment and river health (such as those described by Plafkin *et al.* (1989), Roth *et al.* (1995), Wright (1995) and Allen *et al.* (1997)), together with examination of spatial and temporal variation and study of the IBI's performance in the presence of known environmental impacts.

METHODS

The NSW Rivers Survey design, sampling methods and other details are described in Chapter 1. In developing and applying the IBI, the four drainage regions of the Rivers Survey (North Coast, South Coast, Darling and Murray) were treated as presumptive ecological regions because of their differing climatic, geographic and faunal characteristics (McDowall 1981, 1996; Paxton *et al.* 1989). A separate index was therefore derived for each region. Twelve IBI metrics were calculated for the 20 Rivers Survey sites in each region, using all 55 fish species recorded in the survey and applying the index outlined in Table 6.1. Data on observed fish, as well as those fish actually caught in sampling (Chapter 2) in each of the four presumptive ecological regions, and from all four of the standardised sampling surveys, were used for the IBI analyses.

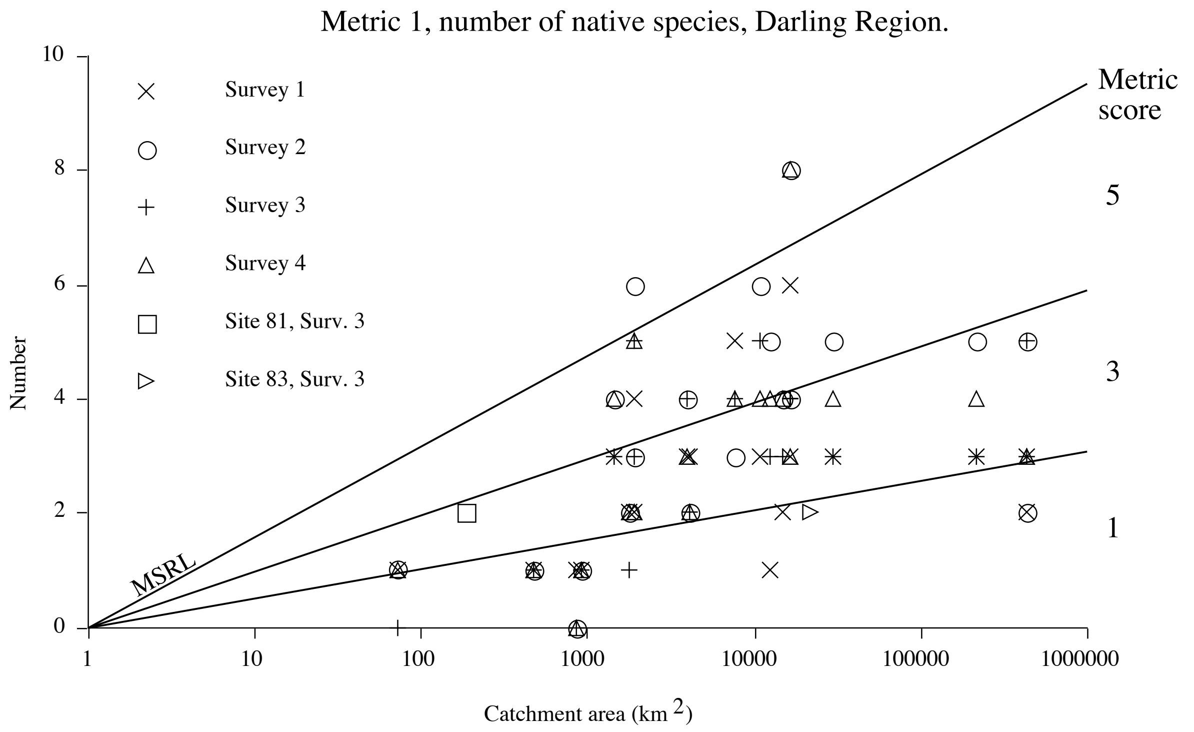
Table 6.1 Fish community metrics used to calculate the Index of Biotic Integrity for NSW Rivers Survey sites. Trophic groups, habitat guilds and other fish-community categories are identified in Chapter 2.

Category	**Metric**	**Scores and criteria**		
		5	**3**	**1**
Species richness and composition	1. Total number of native species.	Expectations for metrics 1-5 vary with stream size and region and are discussed in text.		
	2. Number of riffle-dwelling benthic species			
	3. Number of pool-dwelling benthic species			
	4. Number of pelagic pool species			
	5. Number of intolerant species			
	6. Percent native fish individuals	>67%	33-67%	<33%
	7. Percent native species	>67%	33-67%	<33%
Trophic composition	8. Proportion of individuals as microphagic omnivores	<33%	33-67%	>67%
	9. Proportion of individuals as microphagic carnivores	>67%	33-67%	<33%
	10. Proportion of individuals as macrophagic carnivores	>10%	3-10%	<3%
Fish abundance and condition	11. Number of individuals in sample	Expectations for metric 11 vary with stream size and region and are discussed in the text		
	12.Proportion of individuals with disease, parasites, & abnormalities	0-2%	2-5%	>5%

* Metric 5 includes species with intolerance of various factors including water quality and barriers to migration.

Fish community attributes such as species richness, abundance, and the numbers of species in specialised habitat guilds vary with catchment size (Fausch et al. 1984). These are the six attributes assessed with metrics 1 to 5 and 11. A fundamental assumption of the IBI is that the values of such attributes decline with environmental disturbance, but this assumption is no longer testable in regions where virtually all rivers are degraded to some degree. Nevertheless, the assumption leads to two hypotheses: that sites frequently scoring 3 or 1 for these metrics will show evidence of degradation, and that low or high scores for one metric will tend to be associated with similar scores for other metrics at a site. These hypotheses were examined through the independent assessments of field-sampling teams about habitat conditions at sites, and with analyses of scores at sites (see below).

To account for variation in river size among sites, predictive scoring criteria for fish community attributes assessed by metrics 1 to 5 and 11 were established for the range of river sizes in each ecological region, following the graphical procedure established by Fausch et al. (1984) and Karr et al. (1986) to standardise for river size through catchment area. For each of these six 'predictor' metrics the fish-community-attribute values for the 80 samples in each region (four river-type classifications, each with five replicate sites and four sampling occasions) were plotted against the catchment area of the site on a Log_{10} scale. On each plot a line which passed through the origin and enclosed at least 95% of the data was drawn by eye. This 'maximum predictor' line, originally named the 'maximum species richness line' (MSRL) (Fausch et al. 1984), predicts the maximum likely value of that attribute for an undisturbed fish community of that region in a site of given catchment size which is in excellent environmental condition. The area beneath the maximum predictor line was then trisected, and observations falling in the upper, middle and lower thirds of the graph were allocated metric scores of 5, 3, or 1, respectively. Figure 6.1 illustrates this procedure. Scores for the remaining six metrics were derived from the original IBI model, the sampling results and the trophic and habitat classifications are tabulated in Chapter 2. Metric scores were summed to give the IBI score for each site and each sampling occasion. Qualitative assessment ranges for these IBI scores followed those provided by Karr *et al.* (1986), giving classes of Excellent (58 - 60), Good (48 - 52), Fair (40 - 44), Poor (28 - 34) and Very poor (12 - 22) to enable the biotic integrity of any river reach to be simply classified, both quantitatively and qualitatively. The IBI is undefined if no fish are caught at a site and such sites are simply designated 'No Fish'.



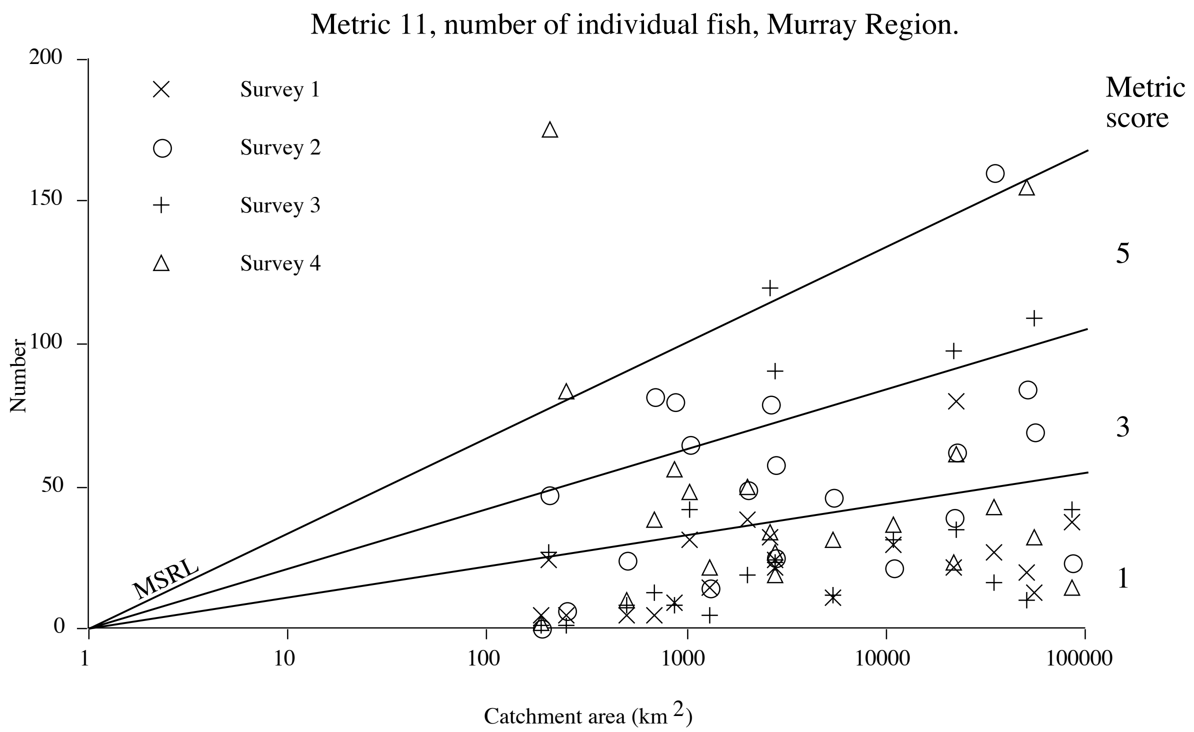
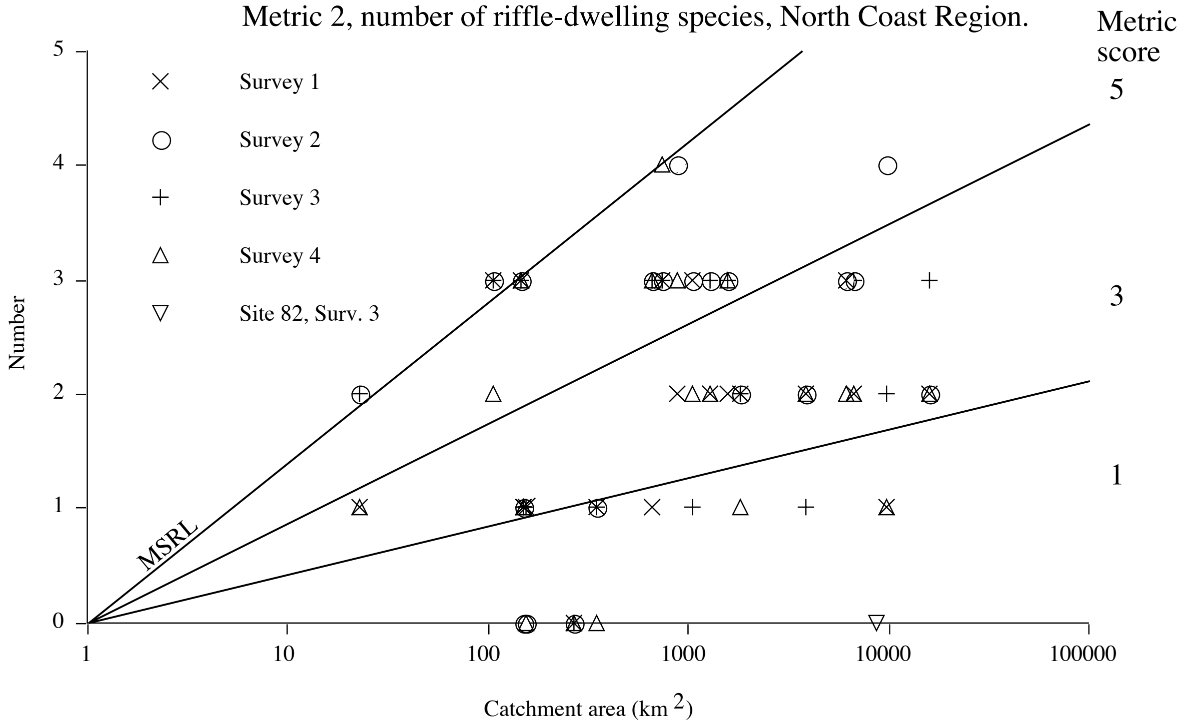


Figure 6.1 Examples of the procedure for establishing the maximum predictor line (MSRL) and for setting values for predictor metrics 1-5 and 11. Values are plotted for each of the four surveys at the 20 sites.

Metric 2, number of riffle-dwelling species, North Coast Region.



Metric 4, number of pool-dwelling species, South Coast Region.

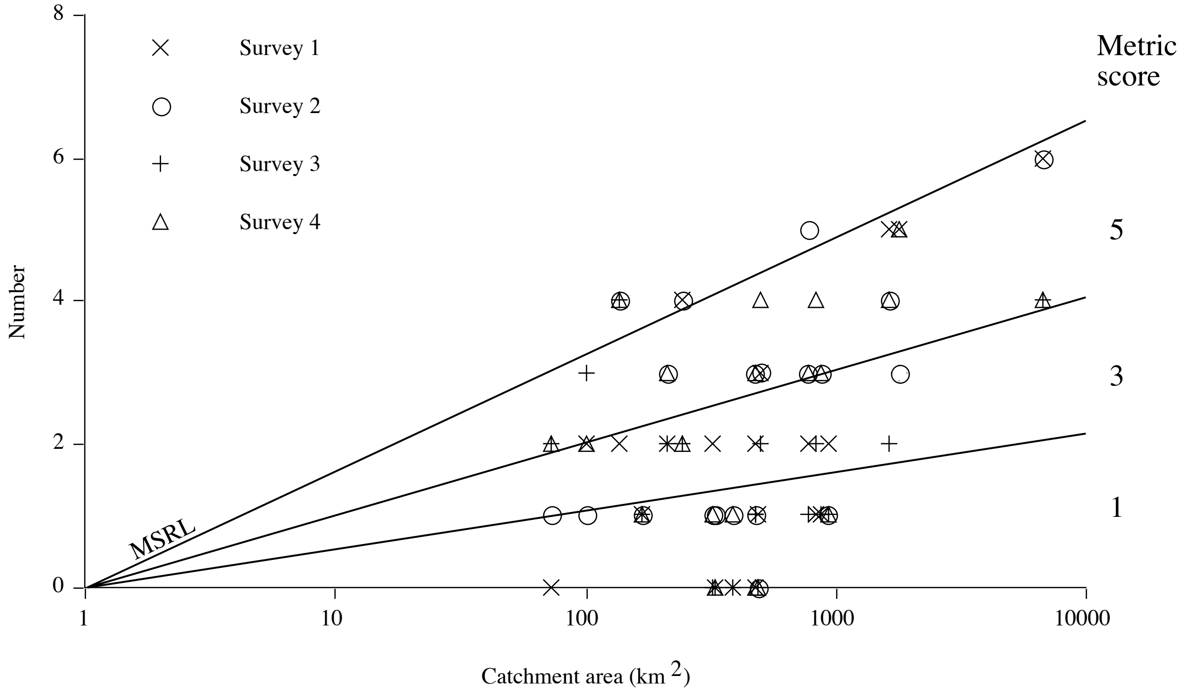


Figure 6.1 continued.

IBI Assumptions

Eight underlying assumptions of the IBI concerning how Australian stream fish communities change with environmental degradation are listed in Table 6.2.

Table 6.2 Assumptions underlying the behaviour of the IBI, as applied to south-east Australian streams, in response to declining environmental condition (modified after Fausch et al. 1990).

1. Number of native species and those in specific habitat guilds declines (metrics 1-4)
2. Number of intolerant species declines (Metric 5)
3. Proportion of individuals that are members of native species declines (Metric 6)
4. Proportion of native fish species declines (Metric 7)
5. Proportion of trophic generalists, especially omnivores, increases (Metric 8)
6. Proportion of carnivores declines (metrics 9 and 10)
7. Fish abundance generally declines (Metric 11)
8. Incidence of externally evident disease, parasites and morphological abnormalities increase (Metric 12)

Metric 1 measures total native species richness, an attribute of faunal communities used in most biologically based environmental-health models to give sensitivity to habitat complexity and relative stability. Metric 2 (number of riffle-dwelling species) provides an indicator of degradation of riffle zones in rivers, a consequence of catchment erosion and siltation or stream-channel instability. Metric 3 (number of pool-dwelling benthic species) and Metric 4 (pool-dwelling pelagic species) provide the index with similar sources of sensitivity to changes in stream geomorphology through the habitat specialisations of these guilds of fishes. These specialist adaptations are not sharply defined among the Australian freshwater fish fauna (McDowall 1996), so that the discriminant ability of metrics 2, 3 and 4 is strong only when several species are involved, usually in the slopes and lowland river reaches.

Metric 5 makes use of the limited available knowledge on variation among Australian fish in their tolerance to different environmental variables, especially through those species particularly intolerant of migration barriers, and others sensitive to particular aspects of water quality (Harris 1995).

Metric 6 (proportion of native fish individuals) and Metric 7 (proportion of native species) both emphasise the value of knowledge about the extent to which the community is native to the reach as a measure of the aquatic environment’s health. There are many examples in the literature (summarised in Harris 1995) of the connection between habitat disturbance and the presence of non-native species.

Metrics 8, 9 and 10 use trophic functions of community structure to indicate various aspects of environmental condition. The main rationale for Metric 8 was the hypothesis that eutrophication, with poor water quality and dominance of algal production, will favour microphagic omnivores in the fish community, such as the alien species *Carassius auratus* and *Cyprinus carpio*. Metric 9 (proportion of microphagic carnivores) is designed to detect variation in abundance of macroinvertebrate prey from the riparian strip or the benthos, and hence reflect the condition of these two ecosystem components. The value of the relative abundance of macrophagic predators in providing an integrative measure of the health of the community in which they form the trophic peak is the basis for Metric 10.

Data supporting Metric 11 (number of individuals) have an increased error component because of the inclusion of estimates of fish observed during sampling, as well as those actually caught and enumerated. This was necessary because large schools of small fish such as *Retropinna semoni* or *Gambusia holbrooki* can overwhelm field teams' capacity to capture them during electrofishing. Rather than ignore useful information, estimates of fish numbers observed were used, even though the accuracy of data was less certain. As an indicator of biomass and ecosystem production, this metric is a reflection of the condition of the river reach. The proportion of visible abnormalities in the fish caught, measured for Metric 12, could possibly be influenced by seasonal factors unrelated to ecosystem condition, for instance the winter outbreaks of fungus infections common among *Nematalosa erebi* (Puckridge 1986). But the relationship between poor water quality and outbreaks of red-spot disease (Callinan and Fraser 1989), as well as more general interpretations of the role of disease as an environmental indicator, provides good predictive value for Metric 12.

The IBI assumes that changes in communities are linearly related to degradation; that increasing disturbance results in proportional decreases in metric values and IBI scores. The index integrates four independent sources of data (number of species, proportion of alien fish, number of individual fish, and incidence of abnormalities), as well as including other, non-independent sources. But the IBI assessment remains a simple, quantitative description designed to provide sensitivity and robustness by combining the key attributes of the fish community of a region that respond to various kinds of ecosystem disturbance. These attributes are additive in the index, and no assumption of statistical independence is required.

Accuracy of assessments

As noted, validation of the IBI using a series of independent measures is being undertaken as the next and final stage of the current tests of the practicality and value of the index. Nevertheless, a preliminary, informal test of the accuracy of IBI rankings of environmental condition was made by relating the rankings to subjective observations of fish habitats recorded at sites during field teams' visits. In addition to recording standard habitat data (Chapter 2), team members were surveyed for their impressions about the condition of each site and any observations on obvious environmental impacts in the area. These qualitative observations were combined with information on known impacts, such as river regulation, and compared with IBI rankings.

Data analysis

A three-way factorial analysis of variance was used to assess the similarity of IBI scores over four different river types, two years and two survey times, at sites in each of the four ecological regions. Each region was analysed independently, as the six predictive metrics were separately derived and differed among regions. Sites with samples receiving an integrity class of ‘No fish’ were omitted from the analysis. The underlying assumptions of normality and homogeneity of variance were verified, Cochran’s test statistic was used to examine homogeneity of variances. Scheffe’s multi-comparisons test was used to find the sources of differences in mean values of the variables which were found to differ.

The hypothesis that low or high scores for one metric at a site will tend to be associated with similar scores for other metrics was investigated by analysing correlations between each metric and the IBI score using data for the 80 sites from Survey 2, the first summer survey.

RESULTS

The IBI revealed a large degree of variation which discriminated (in the sense of aiding recognition of differences) strongly among sites and river types within ecological regions, with samples scoring across most of the possible range, from a minimum of 18 (‘Very Poor’) to the maximum possible, of 60 (‘Excellent’). Because different predictor-metric values were set for each of the four different ecological regions, the index was able to standardise scores within regions, despite strong variations in the species richness and abundance of their respective fish communities. Only one sample at each of two degraded sites, MM24 (*i.e.,* Murray region, montane river type, site number 24) and SCM63 (South Coast, montane, site 63), failed to produce any fish. Details of individual IBI scores for each site and sampling occasion are given in Table 6.3.

Table 6.3 IBI scores for NSW Rivers Survey sites for each survey arranged by river type (M- Montane, RL - Regulated Lowland, S - Slopes, UL - Unregulated Lowland). Sites 81-83 are the three 'rejected' sites, sampled once only.

			Survey							Survey			
Region	**River Type**	**Site No.**	**1**	**2**	**3**	**4**	**Region**	**River Type**	**Site No.**	**1**	**2**	**3**	**4**
Darling	M	1	34	30	32	42	North Coast	M	41	38	38	28	38
		2	32	36	28	34			42	34	34	28	36
		3	38	28	22	26			43	30	32	28	32
		4	32	36	28	40			44	36	34	28	34
		5	38	36	20	44			45	34	30	20	28
	RL	6	44	36	34	44		RL	46	40	38	38	40
		7	28	28	30	34			47	32	38	32	42
		8	34	36	34	34			48	46	52	50	48
		9	46	32	38	46			49	36	40	40	40
		10	26	54	38	40			50	48	50	52	50
	S	11	32	38	34	36		S	51	46	40	42	44
		12	36	50	32	52			52	40	46	38	46
		13	34	32	30	32			53	42	44	38	38
		14	29	38	30	38			54	34	30	28	40
		15	38	38	42	50			55	36	50	44	50
	UL	16	32	36	32	36		UL	56	50	52	50	56
		17	36	44	34	36			57	40	46	46	40
		18	34	38	42	40			58	40	42	38	50
		19	40	54	38	40			59	40	54	38	40
		20	28	34	30	28			60	40	50	50	46
Murray	M	21	38	38	34	44	South Coast	M	61	36	32	30	36
		22	42	40	32	38			62	34	32	28	32
		23	42	46	36	32			63	34	30	NF	32
		24	42	NF	34	38			64	32	34	24	34
		25	42	48	18	42			65	34	36	32	30
	RL	26	24	24	42	22		RL	66	42	44	38	46
		27	28	34	22	24			67	38	42	34	38
		28	42	40	34	36			68	48	52	34	60
		29	36	30	30	40			69	52	50	40	48
		30	32	24	26	28			70	48	40	34	38
	S	31	36	36	28	42		S	71	36	36	32	40
		32	26	38	30	34			72	38	36	28	40
		33	46	38	30	44			73	46	56	38	50
		34	36	48	44	36			74	48	50	36	44
		35	28	40	24	40			75	34	42	36	40
	UL	36	30	30	36	20		UL	76	48	52	44	52
		37	32	36	26	48			77	42	52	38	48
		38	28	32	34	28			78	52	50	38	56
		39	22	40	28	30			79	56	56	38	50
		40	18	26	24	32			80	44	38	36	44
							Darling	UL	81				36
							Darling	S	82				24
							North Coast	S	83				24

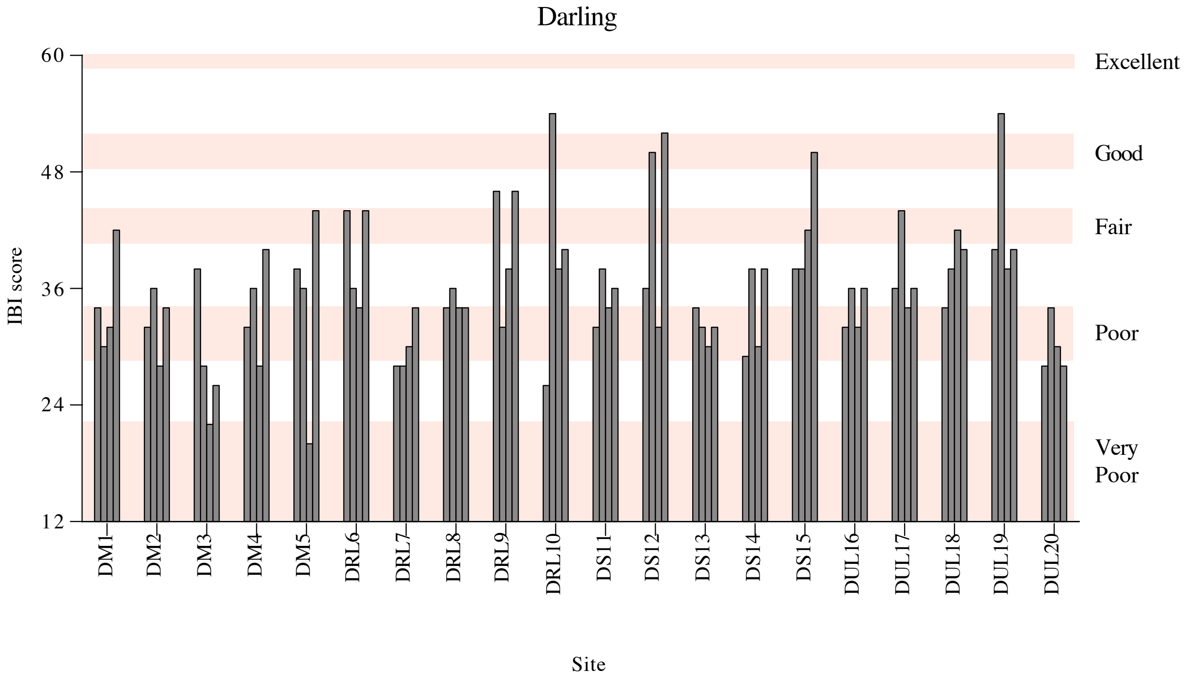
NF - No Fish

Maximum estimated values for each of the predictor metrics in each ecological region, standardised for a catchment area of 10,000 km^2, are given in Table 6.4. Values in all cases were lower in Murray region sites than those in the other three regions. Numbers of individual fish (Metric 11) predicted for a Murray region catchment of 10,000 km^2 were almost an order of magnitude lower than those in the Darling region. These marked differences are interpreted as reflecting, at least in part, the degraded condition of most or all of the 20 sites chosen in the Murray region (see Discussion). The effect of this general site degradation is to lower the MSRL values predicted for the six metrics, thus limiting the ability of the IBI to assess biotic integrity. This problem can be overcome by sampling additional sites of high environmental quality in the Murray region to seek better remnant examples of each of the main elements of the region's biotic integrity and enable re-calculation of the six predictor metrics with more data.

Table 6.4 Maximum values estimated for the six predictor metrics for a catchment area of 10,000 km^2 in each of the four ecological regions.

Metric No.	Metric	Murray	Darling	South Coast	North Coast
1	No. Native species	3.6	6.5	17	25
2	No. Riffle species	1.8	2.3	6.8	5.5
3	No. Benthic species	1	1.3	4.3	5.8
4	No. Pelagic species	5.8	7.5	6.4	14
5	No. Intolerant species	3.5	3.8	6.2	12.5
11	No. Individuals	145	1300	735	1100

Significant temporal variation occurred in IBI scores from the four survey samples at each of the sites (Figure 6.2). But there was a consistent pattern of scores which enabled discrimination among individual sites. Figure 1.3 shows that, among averaged IBI scores for regional and river-type groups of sites, the pattern of variation reflected seasonal changes in catchability and abundance of the fish fauna (Chapters 2 and 10). The seasonal effect accounted for much of the variation in scores at individual sites (Figure 6.2), which showed greater overall consistency between summer scores. IBI scores (Figure 6.3) and qualitative grades (Figure 6.4) based on winter samples were lower than those in the subsequent summer. Qualitative discrimination among sites within regions was also greater in summer samples (Figure 6.4). While average scores for the two summer surveys in each region were very similar, results from the second winter survey were consistently lower than the first (Figure 6.3), indicating reduced values for several metrics, especially metrics 1, 6, 7 and 11. Widespread major flooding had occurred between the two winter samples and was suspected of disturbing fish communities and altering metric values. Overall, the index performed well when based on summer samples for each of the 80 sites, consistently discriminating relative environmental quality by distributing sites along the gradient of IBI scores and qualitative rankings.



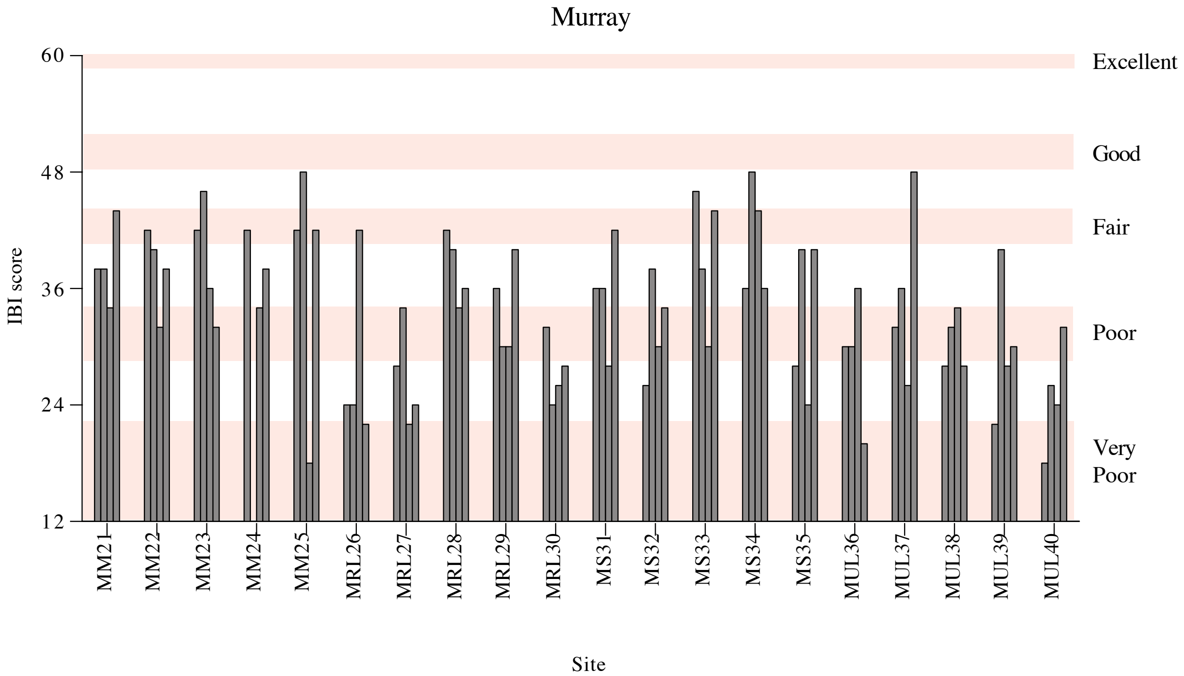


Figure 6.2 IBI scores for the four surveys at all sites, by region. The four columns for each site represent surveys 1-4 respectively; the second and fourth surveys were done in summer months. Sites codes identify region (D - Darling, M - Murray, NC - North Coast, SC - South Coast), river type (M - Montane, S - Slopes, UL - Unregulated Lowland, RL - Regulated Lowland), and site number. Qualitative rankings follow Karr et al. 1986.

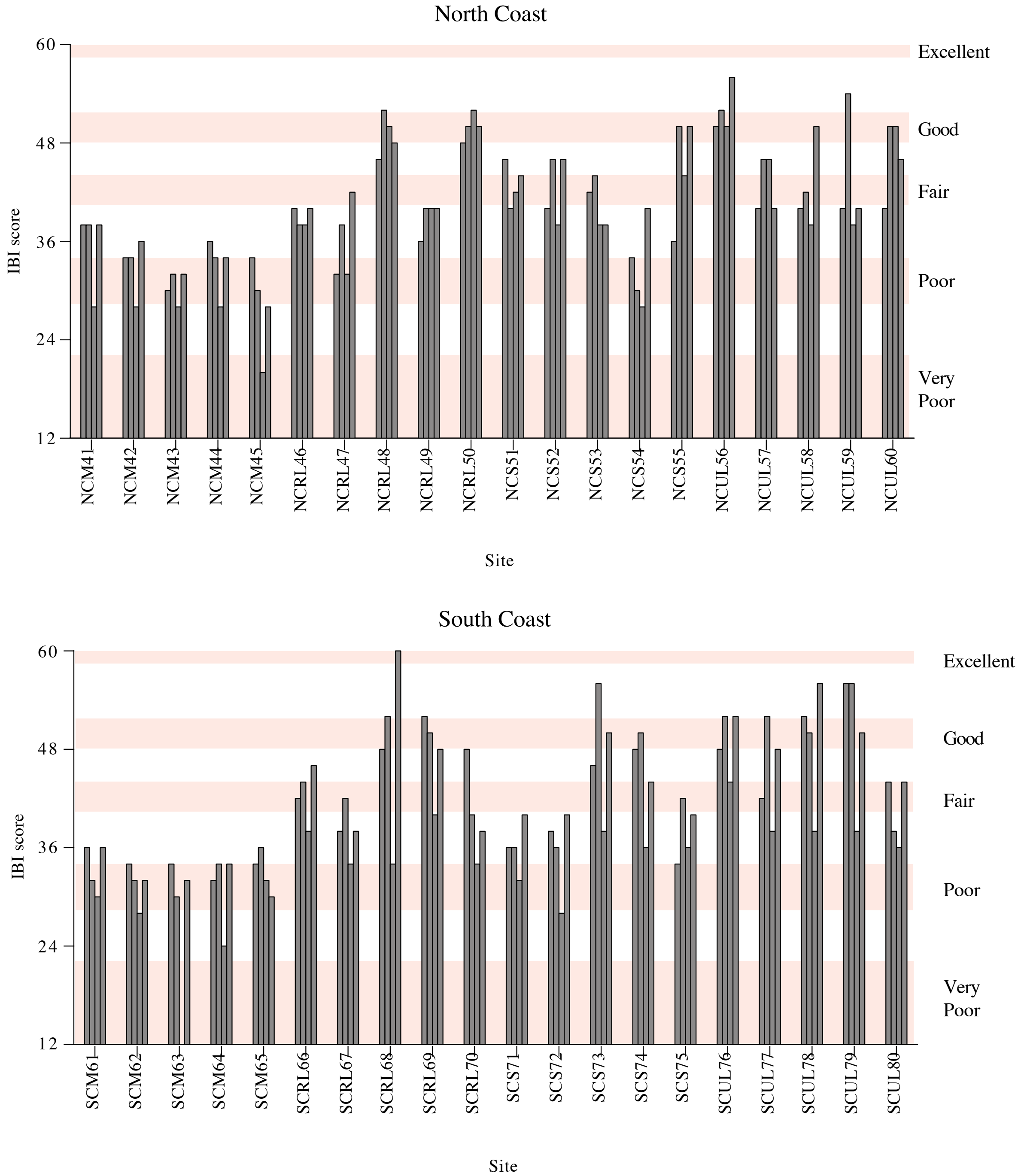


Figure 6.2 (continued)

The three 'rejected' river reaches, rejected from repeated sampling because of gross degradation (Chapter 1), had IBI scores well below mean scores for other sites in their region and river type (Figure 6.3). They were assessed as having 'Very Poor-Poor' biological integrity in two cases (Goulburn River and Castlereagh River). The third rejected reach, in the Cockburn River, was assessed as 'Poor-Fair', with metric 11 (numbers of individuals) having been influenced by observation of very large numbers of the alien species, *Gambusia holbrooki*, a fish known for its adaptability to degraded habitats (McDowall 1996).

All four river types in the Darling region showed similar IBI scores (Figure 6.3). In the two coastal regions, montane sites had lower biotic integrity than other river types, while the reverse pattern occurred in the Murray region, apparently because of low levels of the six predictor metrics. Average variability (R) between consecutive summer scores at individual sites was also much higher in the Murray region (R = 5.8) than in other regions (*e.g.*, South Coast R = 3.9). Figure 6.4 shows how qualitative rankings discriminated among sites on the basis of relative environmental quality. Sites were placed into between four and seven of the ten possible rankings in each region and survey round. Discrimination was strongest in summer samples, with higher ranks and a broader spread of sites.

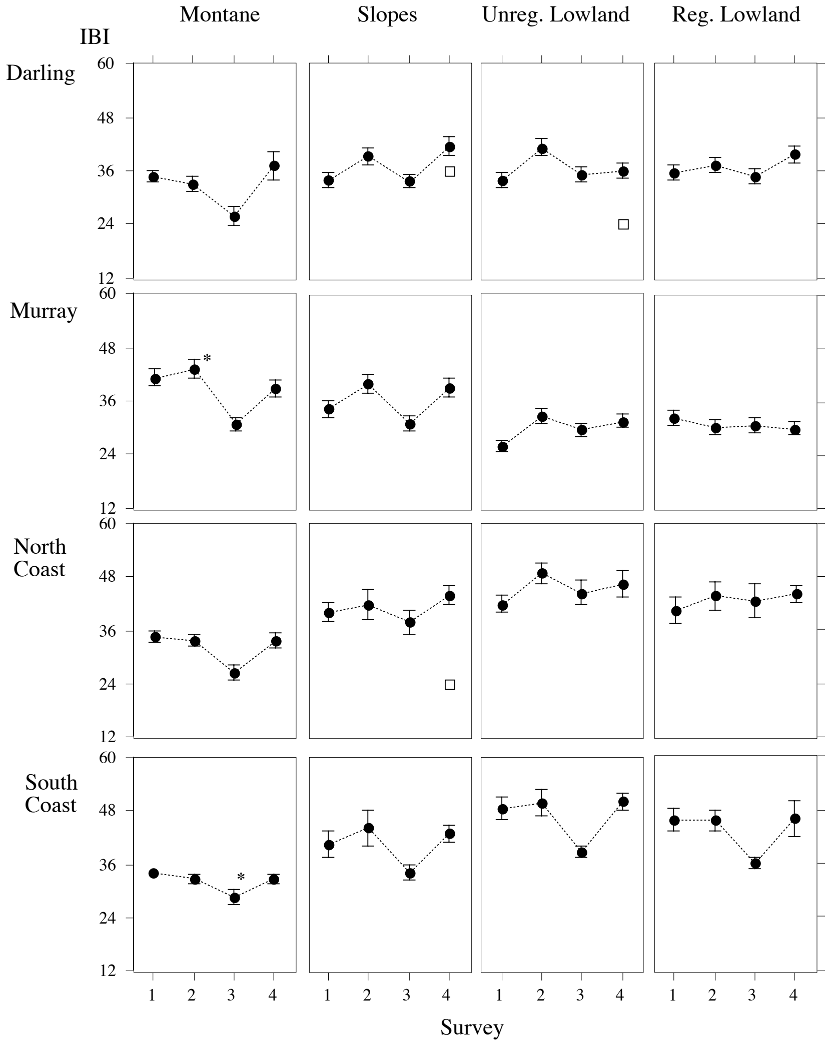
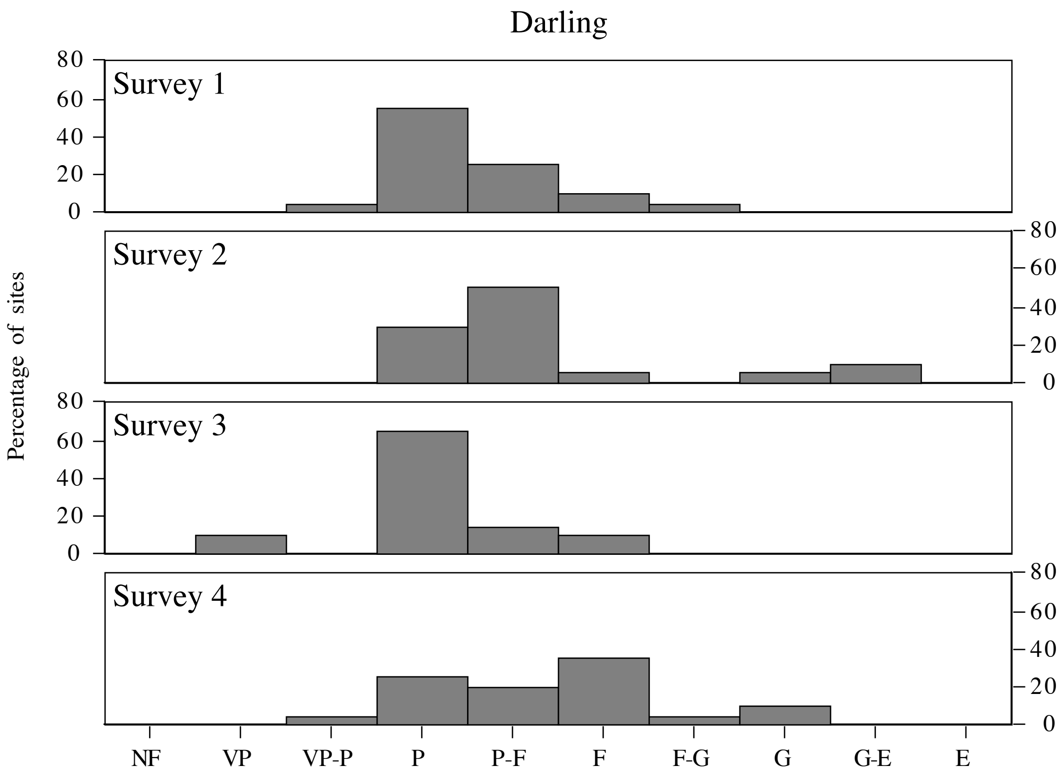


Figure 6.3 Multiway plot showing IBI scores (+/- standard error) for groups of sites at each survey. Scores were averaged over the five sites sampled in each river type in each region. * Indicates that four sites were averaged instead of five, as no fish were caught at one site and therefore no IBI score can be allocated. ☐ Indicates IBI score for 'rejected' sites sampled once only, in Survey 4 (Chapter 2).



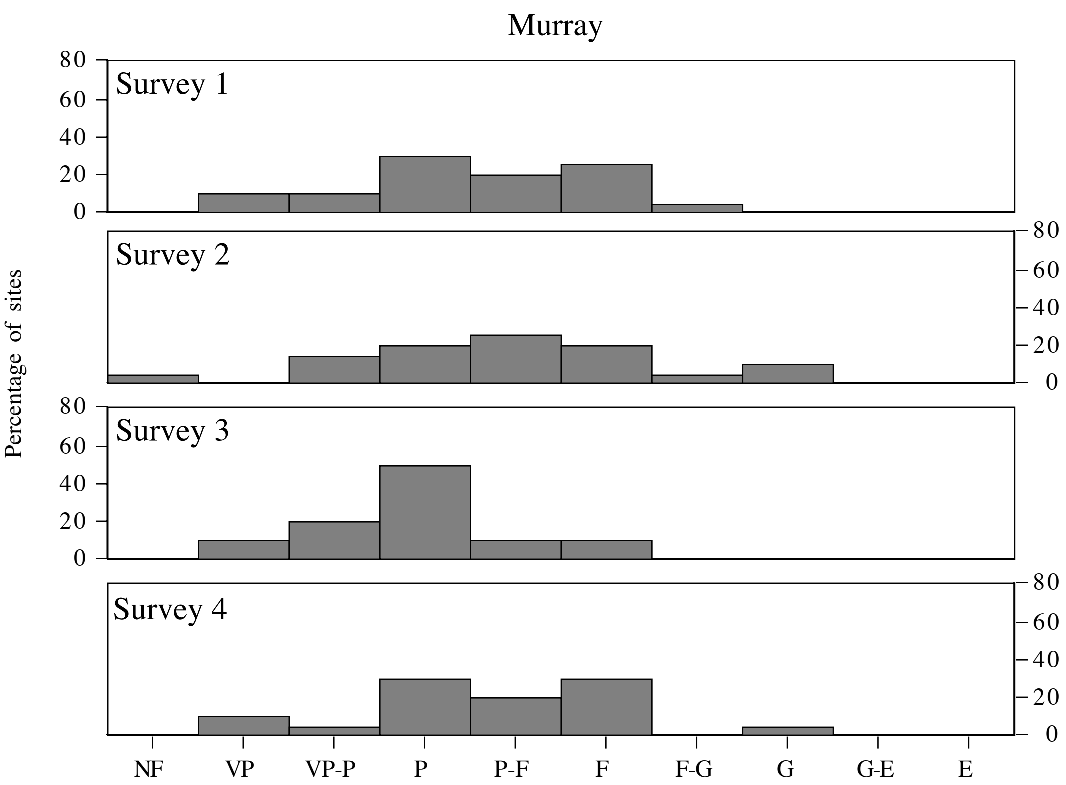
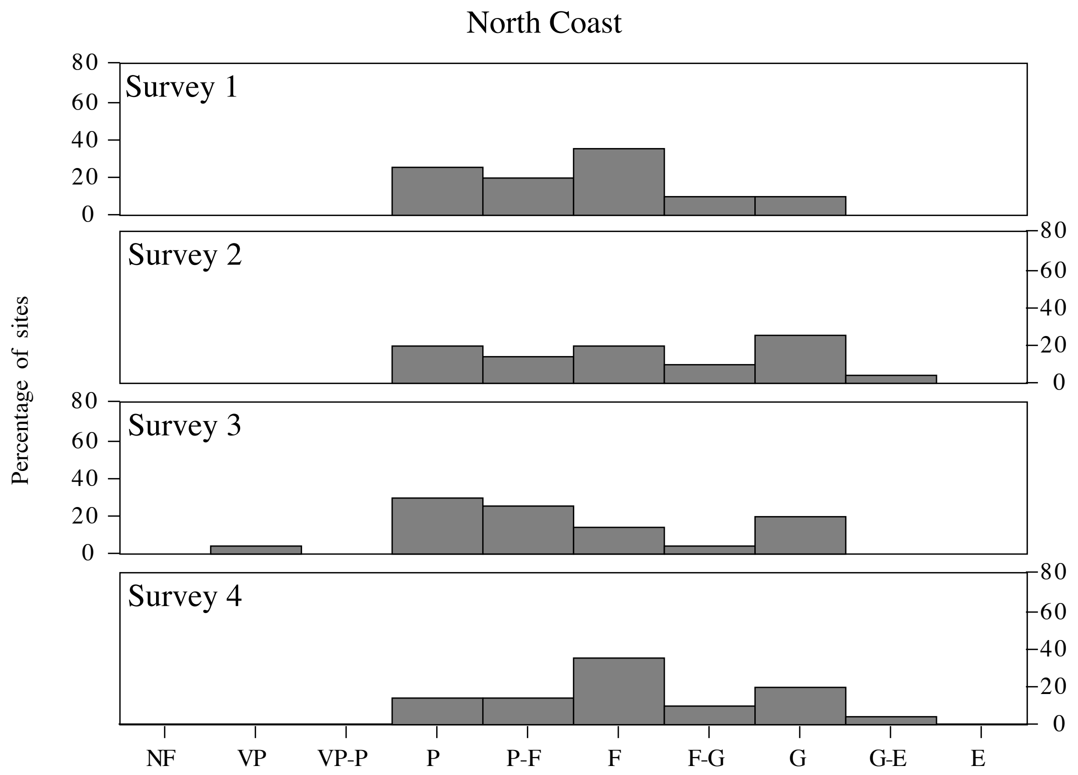


Figure 6.4 Distribution of sites by biotic integrity classes for each region within each survey. Qualitative rankings follow Karr et al. (1986) (NF - No Fish, VP - Very Poor, P - Poor, F - Fair, G - Good, E - Excellent). (N=20 as results were combined for each site within a river type and not averaged.)



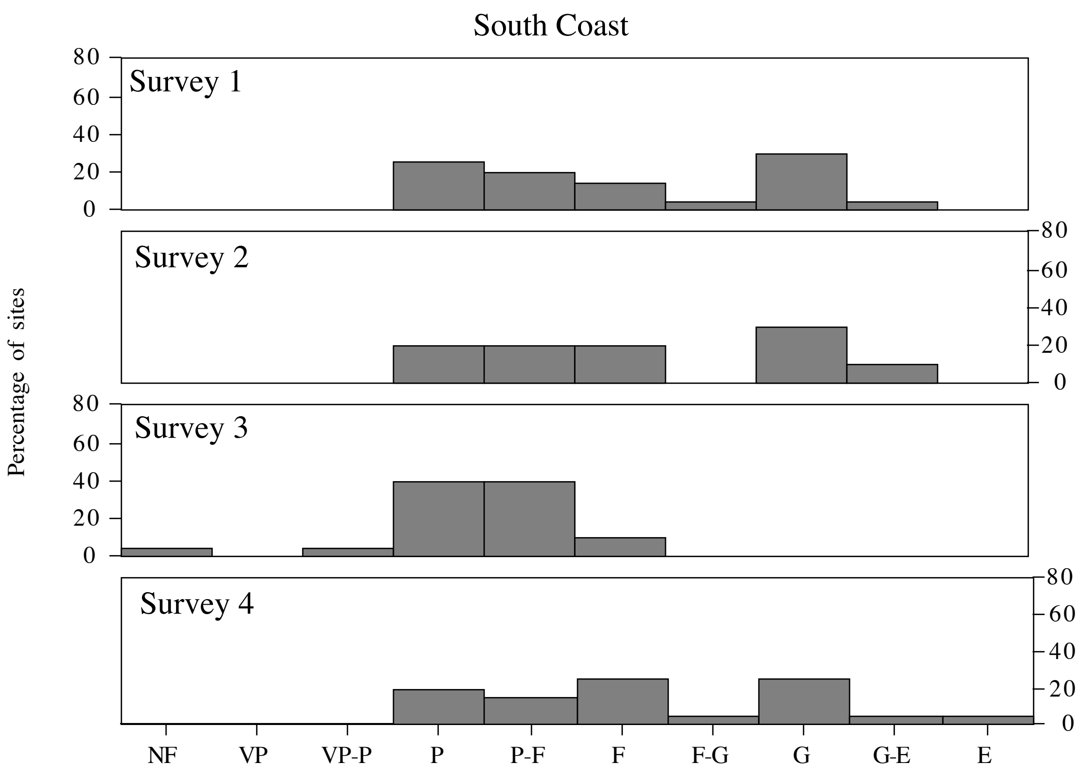


Figure 6.4 continued

As noted, most of the variation in IBI scores arose from differences between summer and winter samples. The coefficients of variation (CV) of IBI scores over all sites, and over the four ecological regions were generally low (Table 6.5). Figure 6.5 and 1.6 show that CV for river types and surveys ranged from 9-23% and 13-27% respectively. No particular pattern of variation was evident from CVs among river types and surveys. At the finer scale of individual sites and surveys, CVs ranged from 4% to 32% (Table 6.5), with the low average level of 11.9% illustrating high repeatability of IBI scores, despite the importance of seasonal effects.

Table 6.5 Coefficients of variation of IBI scores within the four regions, and for the sites, sorted by river types and surveys

		Darling	**Murray**	**North Coast**	**South Coast**
Effect		**CV (%)**	**CV (%)**	**CV (%)**	**CV (%)**
Region		18.6	25.1	18.7	23.0
River Type/ Survey					
Montane	Survey 1	8.7	4.3	8.6	4.2
	Survey 2	11.7	11.1	8.8	6.9
	Survey 3	18.8	23.7	13.6	11.9
	Survey 4	19.6	11.9	11.4	6.9
Slopes	Survey 1	10.3	23.0	12.1	15.4
	Survey 2	16.8	11.7	18.1	20.1
	Survey 3	14.8	24.2	16.2	11.8
	Survey 4	21.3	10.6	10.9	10.2
Unregulated	Survey 1	13.2	22.4	10.6	11.8
Lowland	Survey 2	19.6	16.5	10.1	13.8
	Survey 3	13.7	17.5	13.7	7.8
	Survey 4	13.6	32.4	14.7	8.9
Regulated	Survey 1	25.6	21.6	16.6	12.2
Lowland	Survey 2	26.8	22.5	15.7	11.4
	Survey 3	9.6	24.9	19.8	7.9
	Survey 4	14.0	25.8	10.7	19.7

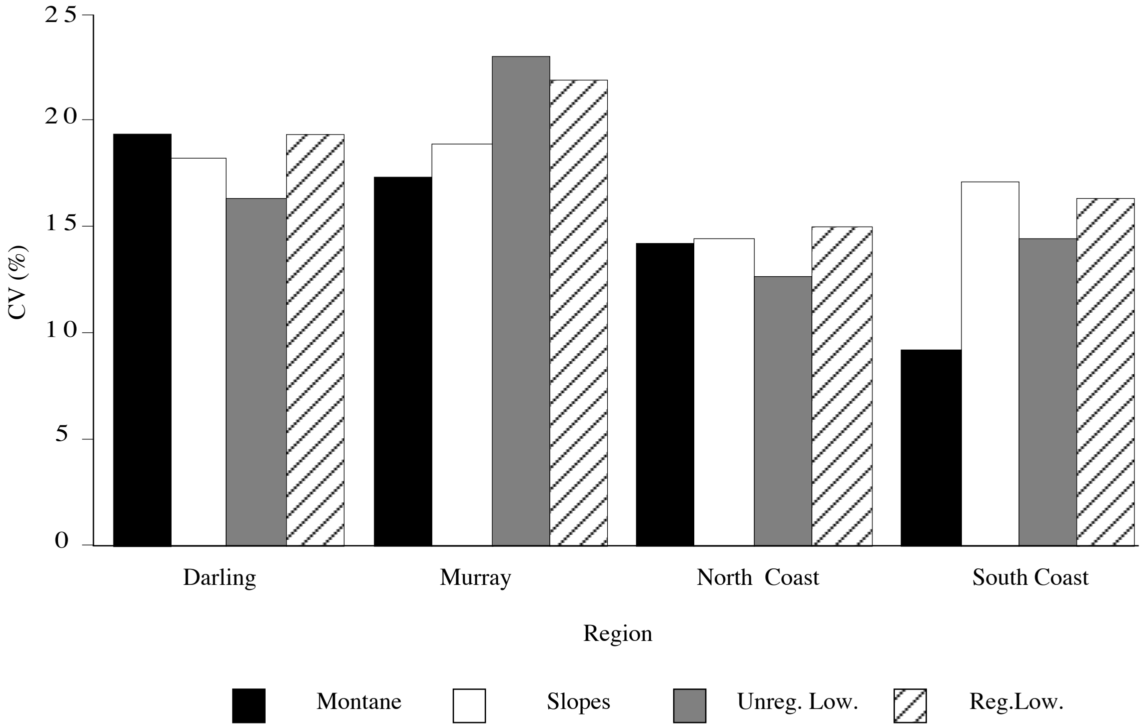


Figure 6.5 Coefficients of variation of IBI scores for each river type within each region.

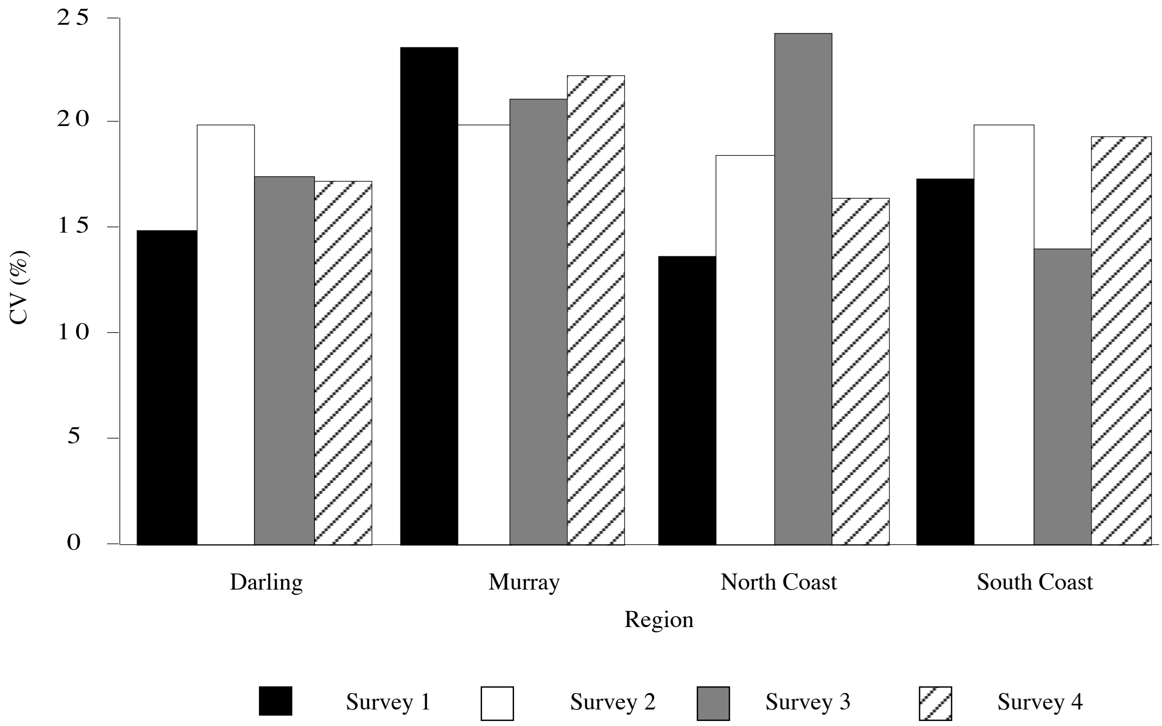


Figure 6.6 Coefficients of variation of IBI scores for each survey within each region.

Comparisons of mean IBI scores in the four surveys for each region are shown in Figure 6.7, and by river types in Figure 6.8. Means for the two summer and two winter surveys were significantly different in all regions except the South Coast, where the second winter survey scored lower than the other three surveys. Summer IBI scores were higher in all regions. No significant difference was found for river-type IBI means in the Darling region. In the Murray region, slopes and montane reaches had similar mean IBI scores, and there were no differences between the regulated and unregulated lowlands river types, where IBI scores were lower than those of the slopes and montane groups (Figure 6.8). In the North Coast, montane river types had a substantially lower IBI than the other groups, and there was a similar result in the South Coast, where the slopes rivers also had a lower IBI than the unregulated lowland reaches.

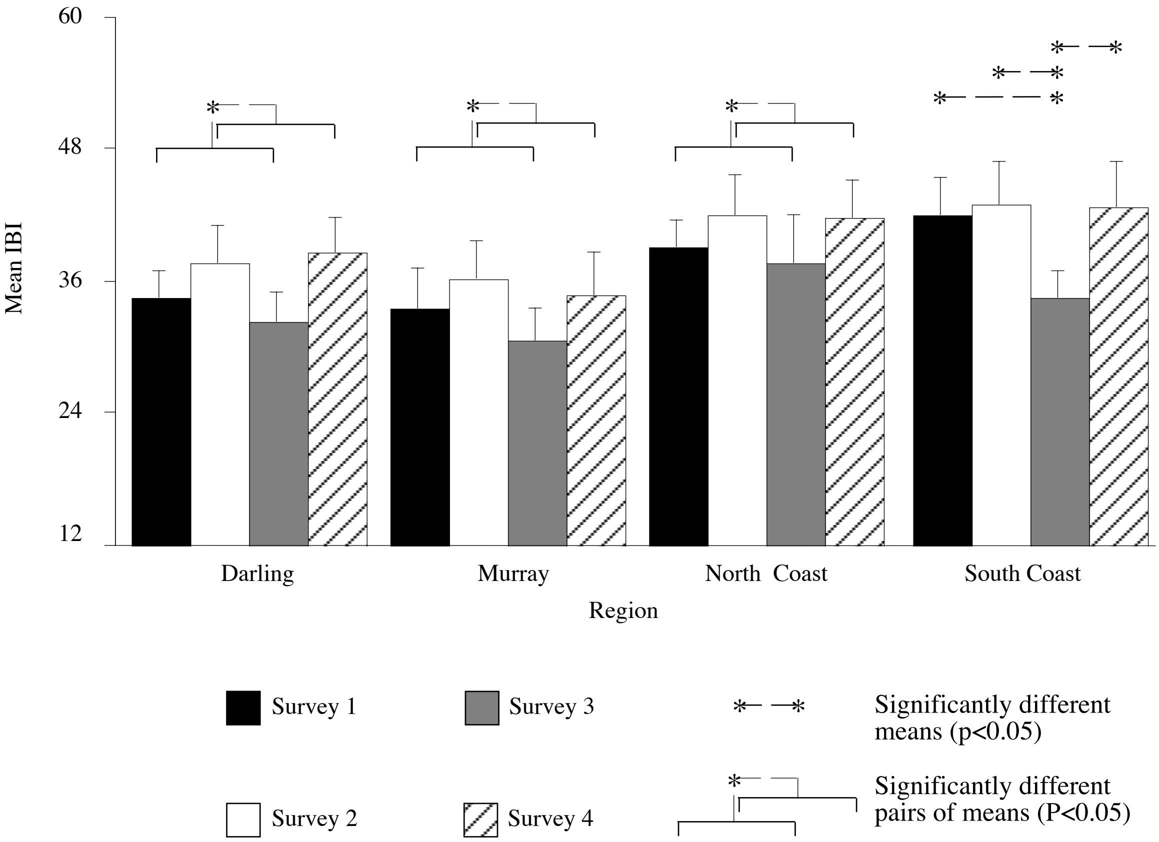


Figure 6.7 Bar plot of mean IBI score (+95% CI) by survey for each region, together with results of Scheffe's pairwise differences test.

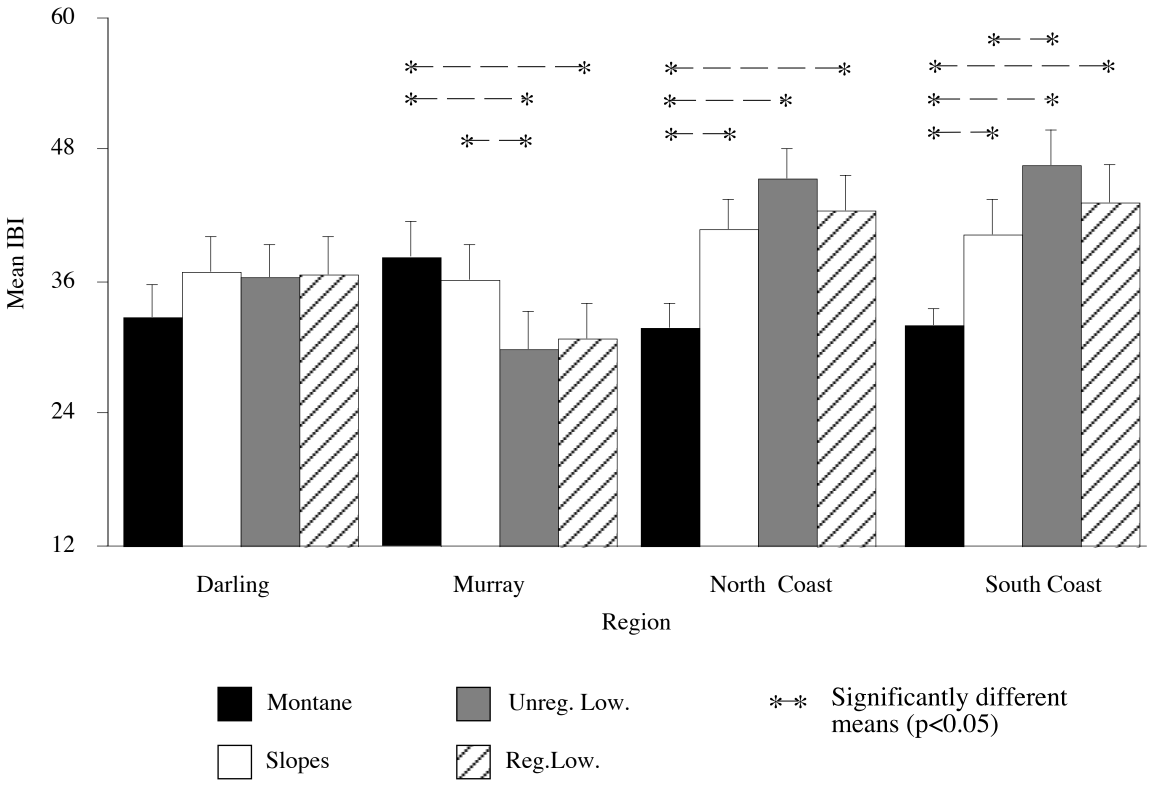


Figure 6.8 Bar plot of mean IBI score (+95% CI) by river type for each region, together with results of Scheffe's pairwise differences test.

Performance of metrics

Correlation analysis (Table 6.6) showed the relative levels of association between each of the 12 metrics and the IBI scores over the 80 sites, as well as biases of metrics. Predictor metrics, especially metrics 1, 2, 3 and 5, and Metric 6, had the strongest relationships to the IBI scores. Only Metric 8 (proportion of microphagic omnivores) failed to correlate strongly ($\alpha = 0.05$) with IBI score. Furthermore, Metric 8 also showed a slightly negative correlation, apparently because some species in this trophic guild are successful in degraded habitats, while others are not (see *Discussion*).

Table 6.6 Correlation analysis of metric scores and bias of individual metrics for Survey 2

Metric	Direction and level of bias*	Correlation coefficient
1	-0.16	0.74
2	-0.24	0.66
3	-0.69	0.54
4	-0.01	0.52
5	-0.41	0.62
6	0.56	0.55
7	0.59	0.45
8	1.36	-0.26
9	-0.69	0.40
10	0.39	0.23
11	-1.31	0.33
12	0.59	0.29

$$* \text{ Bias} = \frac{\sum(\text{metric score-metric mean})}{80}$$

where metric score = 1,3,or 5 and,
metric mean = IBI/12 calculated for each site

Predictor metrics of the index discriminated effectively among sites and river reaches, producing broad distributions of scores and qualitative rankings (Figure 6.2 and Figure 6.4) but with some variability in the strength of their discriminant ability. Discrimination among sites was often quite strong despite the presence of small numbers of species (as low as three) within some habitat-guild metrics. Metric 3 (number of benthic pool-dwellers) did not contribute effectively to the discrimination of sites in the Murray region, with all sites there recording either 0 or 1 species for this metric, thus scoring either 1 or 5. Metric 11 (number of individuals) was based on skewed data for each of the regions, especially the Darling, with a few samples producing very large numbers of fish. This distortion was associated with the inclusion of ‘observed’ data because of the large numbers able to be recorded in this way, and it reduced the discriminant ability of Metric 11. This metric also showed considerable negative bias in IBI scores (Table 6.6). Future applications of the index may need to transform abundance data to normalise the distribution and improve the performance of Metric 11. The six predictor metrics performed satisfactorily over the lower range of site-catchment areas, with generally consistent results down to the minimum catchment size recorded, of 23km^2.

Site classifications

Average IBI score is not generally used in assessments, as combining samples from multiple surveys at a site may conceal patterns of variation and can lead to false expectations about species richness when seasonal movements occur among different species (Karr *et al.*, 1986). This was demonstrated by site SCUL78 in the Tuross River where the fauna is dominated by catadromous species (fish which migrate to marine environments to breed), many of which move from rivers to estuaries in winter. The IBI category of ‘Poor-Fair’ in Survey 3 (winter) rose to ‘Good-Excellent’ in Survey 4 (summer) partly because of strong differences in seasonal numbers of catadromous fish (Harris *et al.* 1996) and partly because of the general winter decline in fish abundance and species richness (Chapters 2 and 10).

An exception to the ‘no averaging’ rule was made to identify the ten highest-scoring and ten lowest-scoring sites of the Rivers Survey (Table 6.7), using average IBI score over the four surveys. The ten highest-scoring sites are all in coastal drainages, and nine of them are near the tidal limit of the estuary. Several factors probably contribute to high scores in sites near estuaries. Species that are essentially estuarine or marine commonly move into fluvial habitats for brief periods (McDowall 1996; Paxton *et al.* 1989), thus increasing species counts and scores for Metric 1 (Chapter 3). Furthermore, as rivers are largest at this point in their course before reaching the estuary, they are most resistant to scale-dependent environmental impacts, such as river regulation and forms of pollution. Estuaries also provide a buffering reservoir of fish that can quickly move upstream to recolonise following disturbance.

The most consistent high-scoring site was NCUL56, in Wilsons River, with an IBI category of either Good (50, 50 and 52) or Good-Excellent (56). This site had a total species count of 17 throughout the four surveys and no alien species were recorded. The highest IBI score at this site occurred in Survey 4, largely because of the high abundance of *Retropinna semoni* observed on that visit. Wilsons River failed to reach a perfect score of 60 because of relatively low numbers of macrophagic carnivores. Habitat attributes supported the high IBI scores (Harris *et al.* 1996).

Lowest-scoring sites included two of the three ‘Rejected’ reaches (NCS82, Goulburn River and DUL83, Castlereagh River) and four Montane sites (SCM63, NCM45, DM3 and MM24) whose habitat attributes, supported by the subjective assessments of the field-teams, were classed as severely degraded. Three of the remaining four lowest-scoring sites (Table 6.7) are all in highly regulated reaches of the Murray region, previously shown by Gehrke *et al.* (1995) to have fish communities severely affected by regulation. The last site, MUL40, is in Billabong Creek in the Murray region, whose original survey classification as ‘Unregulated’ now appears to have been questionable.

Two Murray-region sites (MUL40 and MM25) recorded the lowest observed IBI score of 18. Site MUL40, (Billabong Creek) had low scores throughout all surveys, ranging from Very Poor-Poor, to Poor. The lowest score of 18 in Survey 1 for this site can be attributed to the presence of only two species, carp and goldfish, both of which are alien species. The score of 18 for Site MM25 on the Retreat River occurred in in Survey 3, and appeared to be primarily a winter-sampling effect. The worst-scoring site, on average, of the NSW Rivers Survey was MM63 in Wullwye Creek. This site is affected by sewage effluent and catchment erosion. Its IBI ranking went from ‘Poor’ in Survey 1 to ‘No Fish’ in Survey 3.

Nine of the ten ‘best’ reaches as judged by the IBI (Table 6.7) (all but Site SCRL68, in the Brogo River) recorded habitat variables which were indicative of good-to-excellent conditions. At the opposite end of the scale, six of the ten ‘worst’ sites (SCM63, NCS82, DUL83, NCM45, DM3 and MM24) had clearly degraded habitats, suffering from catchment erosion, channel instability, siltation, riparian devegetation, weed infestation and/or sewage-effluent discharge. Three of the ten ‘worst’ sites (MRL30 Murrumbidgee River, MRL27 Murray River, and MRL26 Bundigerry Creek) recorded habitat attributes that were fair to good, but these reaches are heavily regulated for irrigation supply. As noted earlier, MUL40 Billabong Creek is also more intensively regulated than was apparent at the original classification.

Table 6.7 Ten sites with the highest (a) and lowest (b) IBI scores averaged over all four surveys. (Codes as for Figure 6.2.)

a.

Site	Waterway	Survey 1	Survey 2	Survey 3	Survey 4
NCUL56	Wilsons River	50	52	50	56
NCRL50	Rocky Creek	48	50	52	50
SCUL79	Buckenbowra River	56	56	38	50
NCRL48	Emigrant Creek	46	52	50	48
SCUL76	Clyde River	48	52	44	52
SCUL78	Tuross River	52	50	38	56
SCRL68	Brogo River	48	52	34	60
SCRL69	Woronora River	52	50	40	48
SCS73	Deua River	46	56	38	50
NCUL60	Leycester Creek	40	50	50	46

b.

Site	Waterway	Survey 1	Survey 2	Survey 3	Survey 4
SCM63	Wullwye Creek	34	30	NF	32
NCS82	Goulburn River	-	-	-	24
DUL83	Castlereagh River	-	-	-	24
MUL40	Billabong Creek	18	26	24	32
MRL27	Murray River	28	34	22	24
MRL30	Murrumbidgee River	32	24	26	28
MRL26	Bundidgerry Creek	24	24	42	22
NCM45	Gara River	34	30	20	28
DM3	Gwydir River	38	28	22	26
MM24	Cooma Creek	42	NF	34	38

- Indicates 'rejected' site, not sampled during that survey.
NF - No Fish

Regional and river-type analyses

Table 6.5, Figure 6.5 and Figure 6.6 show the precision of the IBI scores at large temporal (season) and spatial (river-reach) scales, although variances of IBI scores bear no clear relationships to variances of biological characters, the magnitude being determined by arbitrarily chosen functions (Suter 1993). ANOVA results for the Darling region (Table 6.8) showed a significant effect for time of sampling, with higher scores from samples during the summer months (Figure 6.7). There were no significant differences in overall river-type scores for the Darling region. Murray region montane sites scored significantly higher on the biotic integrity scale than the two lowlands river types (Table 6.8, Figure 6.8) and there were also significantly higher scores in summer (Figure 6.7). Among North Coast sites (Table 6.8) there were also significant differences in both river type (Figure 6.7) and time of sampling (Figure 6.8), with montane sites and winter samples scoring lowest. Montane sites also scored lowest in South Coast sites (Table 6.8, Figure 6.8), and there was a significant year/sampling time interaction, with strong differences between years for winter-sample scores (1995 being higher), but similar summer scores.

Table 6.8 ANOVA results for each region. Significant probability values are shown bold.

	Darling		Murray		North Coast		South Coast	
Factor	**DF**	**Probability**	**DF**	**Probability**	**DF**	**Probability**	**DF**	**Probability**
River type	3	0.1077	3	**0.0002**	3	**<.0001**	3	**<.0001**
Year	1	0.6542	1	0.1248	1	0.5767	1	**0.0014**
River type * Year	3	0.7292	3	0.2273	3	0.4828	3	0.9383
Time	1	**0.0013**	1	**0.0145**	1	**0.0066**	1	**0.0002**
River type * Time	3	0.8313	3	0.2279	3	0.9442	3	0.4721
Year * Time	1	0.2762	1	0.6706	1	0.6322	1	**0.0018**
River type * Year * Time	3	0.1237	3	0.6154	3	0.2958	3	0.8060
Residual	64		63		64		63	

Unexpected results

Some unexpected biotic-integrity assessments were made, especially at SCRL68 in the Brogo River, the the only site to achieve a perfect IBI score of 60. This site is affected by dairy farming and some siltation. It was also classed as a regulated site, although Brogo Dam is seldom used to discharge regulated flows and almost all discharges are over the spillway following rainfall. Being only a few kilometres above the tidal limit, the fauna benefits from the estuarine faunal influences and catchability is high (see *Discussion*). Furthermore, all of the 15 native species recorded at the Brogo River site (Chapter 3) have been recorded as migrating upstream (McDowall 1996; Thorncraft and Harris 1996; Mallen-Cooper *et al.* 1995; Harris 1984; and NSW Fisheries unpublished data). Presumably the impassable barrier of a dam close upstream (25 km) has a concentrating effect by blocking these population movements, thus artificially increasing abundance, provided water quality is not affected.

Another unexpected result was that four of the ten highest-scoring sites are regulated sites. These four regulated sites are all in coastal drainages. They illustrate a difficulty with the design for the NSW Rivers Survey; it was difficult to identify sufficient replicate sites for regulated reaches in coastal rivers (see *Discussion*).

DISCUSSION

Effectivenesss of the IBI

The effectiveness of the IBI could have been strengthened if the measurement of sites to set metric values was the sole purpose of the NSW Rivers Survey, as least-disturbed sites could have been deliberately selected for sampling in all regions and river types. This would have improved the probability of sampling sites with highest levels of the fish-community attributes on which the metrics are based. It would probably have raised the MSRL for predictor metrics in the Murray region, where few sites are not extensively degraded, and thus would have distributed IBI scores more widely. Values for Murray-region predictor metrics appear to have been set too low because the randomly selected river reaches contain few if any sites that are of high environmental quality. Table 6.4 shows that values for all predictor metrics were well below those in other regions. Most Murray sites were regarded by sampling teams as having been affected by siltation, sewage effluent, alien plants and animals, catchment degradation, intensive grazing and/or river regulation.

A deliberate selection of better-quality sites would also have provided a stronger distinction between IBI results for the Murray region generally and those of the other ecological regions, where biological integrity is higher. But site selection for the NSW Rivers Survey was randomised to permit general conclusions on river health throughout the state. It is possible and desirable to add data from new, appropriately sampled sites to the graphs used for predictor metrics and thus to strengthen their predictive power, despite the warning of Karr *et al.* (1986) that incremental sampling at single sites can falsely inflate metric values where seasonal fluxes of species occur. Additional, higher-quality sites in the Murray region should now be sampled to add to the existing data, so that the predictor metrics can be re-calculated. In the same way, all four regional IBI indices can possibly be refined in the future by using additional sampling data to re-calculate the criteria for metric scores.

Ecological regions

The results showed the IBI is robust enough to be applied in different regions and river types. Distributions of scores were comparable across the four presumptive ecological regions despite the substantial differences in the fish communities of the four regions (Chapters 3 and 4). There was a similar level of site discrimination in each region. This adaptability was reflected in the strongly different predictor-metric ranges that were set for river catchment areas in each region Table 6.4).

As noted, it was difficult to identify sufficient regulated reaches in coastal regions. Large-scale irrigation industries do not occur in these eastern rivers as they do in the Murray and Darling regions, and there are few coastal rivers other than the Hawkesbury and Hunter where the 'regulated' criterion - '*rivers with flows that are substantially modified from the natural condition by the operation of a dam upstream'* (Chapter 1) - is well met. Furthermore, while coastal dams exert a strong barrier effect (Mallen-Cooper 1994; Harris 1984), and many also suppress high-flow components of the flow regime, their impact on downstream flows and habitats often differs from inland dams (Gehrke and Harris 1996). Coastal dams, in contrast to those for irrigation inland, commonly remain at or near full storage capacity and supply downstream flow by discharges over the spillway, providing both high-quality water and small-scale flow variability. These and other operational factors mean that the 'regulated' reaches in coastal regions are much less affected by disturbances of flow and quality of released water than those in the Murray and Darling regions.

Predictor metrics and river size

The process of setting the predictive levels for metrics 1-5 and 11 showed that the ranges of values for these fish-community attributes increase with catchment area (Figure 6.1), at least up to a maximum of about 10,000km^2. This validated a basic assumption: that the attributes measured by these metrics are related to river size, confirming earlier work in Victoria by Lake (1982). Several of the data distributions increased with catchment size only up to a maximum of about 10,000km^2, after which values of the independent variable ceased to increase and the distribution levelled off (Figure 6.1). Steedman (1988) found a similar effect in Southern Ontario was caused by the significant degradation of all larger streams. Eighteen of the Rivers Survey sites, 17 of them in inland regions, had catchments which exceeded 10,000km^2. All these sites were substantially impacted by various factors, mostly river regulation, so that the approximately asymptotic form of the predictive relationships may be due to degradation, or may simply illustrate a natural limit to species richness. This issue warrants further study but our data suggest it would be best to accept MSRL values at 10,000km^2 as the maximum expected levels for the six predictor variables.

Assumptions of linearity

As noted earlier, the IBI assumes that changes in communities are linearly related to degradation; that increasing disturbance results in proportional decreases in metric values and IBI scores. Further verification is needed for the linearity of some metrics when used with Australian fish communities, but there is supporting evidence in a number of cases. The catchment-scale declines in the proportions and numbers of native species (metrics 1 and 7) and individuals (Metric 6) in response to cumulatively increasing obstruction of stream-migration pathways by multiple migration barriers have been described (Harris and Mallen-Cooper 1994). Linear responses have been shown to river-flow diversion in streams of the Murray-Darling basin (Gehrke *et al.* 1995), with alien species representation (metrics 6 and 7) and loss of fish-community diversity (Metric 1) both increasing with the intensity of flow diversion. Increasing incidence of external disease lesions was observed in fish from more-eutrophic waters in the Hawkesbury system (Metric 12) (D. Pollard, NSW Fisheries, personal communication). The increasing success of alien species with increasing habitat degradation (metrics 6 and 7) has often been observed (Arthington 1985; Harris 1995). Steedman (1988) observed a curvilinear response to enrichment of warm Canadian streams in the numbers of tolerant alien fish, and used a non-linear function for this metric. But as yet there are no indications of such curvilinear responses among Australian fish. Thus the assumption of linear response has been supported for many aspects of the IBI but additional confirmation is still needed. In particular, the assumption of linear decline in fish abundance with increasing disturbance, such as higher nutrient concentrations, needs to be examined. Data from the Hawkesbury River (Gehrke *et al.* 1996) are available for such a test. Data are also recently available to examine responses to riparian degradation (Growns *et al.* 1997) and eutrophication (Pollard *et al.* 1994) in the Hawkesbury River system.

This consideration of the form of IBI responses to disturbance needs to include not only the fine-scale patterns of change within individual metrics, but also, more importantly, the response of the IBI as an integrated whole. Eutrophication is a good example; if the recognised non-linear pattern of response in stream invertebrates (whereby moderate eutrophication increases animal abundance but more-severe levels cause a decline (Hynes 1970)) is repeated in fish, then Metric 11 will not meet the assumption of linearity. Nevertheless, if this situation should arise it is predictable that values for the other metrics 1, 5, 6, 7, 9 and 10 would decrease in proportion with increasing eutrophication, thus providing the integrated index with considerable robustness.

Sampling season

Fish catchability is reduced in colder months (Chapter 10), limiting the value of winter IBI assessments. Our data and analyses clearly support the recommendation of Karr (1987) that sampling for IBI assessments should only be done in summer. For instance, site SCRL68 on the Brogo River was classified as 'Excellent' in Survey 4, during summer. But this same site was classed as 'Poor' in the preceding winter (Survey 3) when there was low total abundance and catadromous species were poorly represented. Seasonal patterns of native fish occurrence and abundance are also believed to be common, although they have not been well documented among Australian species. Catadromous fish such as *Potamalosa richmondia, Mugil cephalus* or *Macquaria novemaculeata* migrate downstream to estuarine spawning areas in winter (Harris 1988). Inland fish also appear to undergo seasonal fluxes, moving into and out of cooler, higher-altitude waters in summer and winter months respectively. Furthermore, mortality patterns among some abundant fish such as *Nematalosa erebi* show strong winter peaks (Puckridge 1986), and small, short-lived fish such as *Hypseleotris* spp.*, Galaxias olidus* or *Retropinna semoni* may have similar mortality peaks in winter, especially in headwater streams, although evidence for this is so far limited to casual observations. Clearly these various sources of summer increases in abundance and species richness mean that assessments of environmental quality based on fish-community measures are best confined to the warmer months of the year. Figure 6.7 reinforces this conclusion.

Sampling needs

Because fish catchability is variable temporally and spatially, it presents a source of error in measures of fish communities based on field sampling. Missed species or erroneous abundance estimates produce misleading metric scores, although the integration of 12 metrics based on four independent classes of attributes provides considerable robustness. The NSW Rivers Survey was designed to maximise the identification of fish species and sizes at a site through intensive sampling effort and the application of five different sampling methods (Chapter 1). These provisions also limit the inherent sampling error in the data. Nevertheless, variation in catchability is evident in the results, mostly associated with sampling season. Electrofishing methods contributed most to fish catches (Chapter 10), thus the values of the 12 metrics were influenced more by variation in electrofishing success than by variation in the other methods. Fish species and sizes show different catchability by electrofishing, and Cowx (1991) summarised the environmental sources of this variability. They include water temperature, conductivity, turbidity, velocity and depth. Catchability was believed by field teams to be relatively low in fast-flowing, deep river sites such as Site MRL27, Murray River, because there was little time for attracted and immobilised fish to be netted before being swept away from the electric field by the strong flow. This source of variability in sampling efficiency will be difficult to overcome completely, although sampling in a downstream direction in such sites is one potentially useful option. Despite predicted adverse effects of turbidity on catchability, through a decreased likelihood of seeing and netting fish below the surface, there was no detectable difference in relative efficiency of electrofishing and passive methods at turbid and clear sites (Chapter 10). High catchability is likely at sites that are shallow (<1.5m), warm, lacking dense cover and with good water clarity, as shown by the large catches in lowland reaches of small-medium coastal rivers. Furthermore, small-to-medium coastal streams such as the Tuross, Clyde, Brogo, Emigrant, Wilsons or Buckenbowra rivers are extremely well-suited to most fishing techniques, and especially boat electrofishing, just upstream of the estuary. Relative to 'higher-energy' sites, currents are slow, substrates fine and uniform, depths moderate and even, and habitat accessibility is good. All these features favour catchability. So the fauna of such near-tidal sites may be more completely sampled than those in higher altitudes. High catch rates and good IBI rankings of coastal reaches near the tidal limit are probably associated with good electrofishing catchability for these reasons, as well as the strength of their fish communities. From the general consistency of fish-community results between the two summer samples at most sites and the relatively low CV of the IBI grades (Table 6.5), it appears that variable catchability has not had a strong effect except between summer and winter samples.

Sampling is therefore a largely avoidable source of error in IBI assessments. The extensive experience gained in the NSW Rivers Survey provides a firm basis for designing sampling programs. Because of the large temporal and spatial scales over which fish integrate environmental conditions, infrequent sampling on a river-reach basis is suitable for many types of monitoring. Experience (Chapter 10) also shows that electrofishing can be used to replace passive sampling with multiple gear types, without adversely affecting data quality and with large improvements in speed of assessment and cost-effectiveness. Electrofishing effectively represents the full range of species in New South Wales rivers (Chapter 10), although backpack electrofishing may not give good estimates of relative abundance in streams with the deep pool habitats that are not suited to this method. Electrofishing techniques impose significant requirements for capital and staff training, implying that such sampling will continue to be largely the province of fisheries agencies, academic institutions and small numbers of specialist consultants.

The efficiency of assessment and monitoring methods is a significant issue for agencies responsible for river health. Cost calculations (Chapter 2) of the NSW Rivers Survey sampling program show that, while capital and labour costs of fish-community sampling are relatively high, large amounts of results are produced rapidly (Chapter 10). Although no good comparative data are available, costs per sample appear modest by comparison with other biological methods. This is because fish sorting, enumeration and species identification are comparatively straightforward, and data can be produced immediately at the end of fieldwork with no need for lengthy laboratory or analytical procedures once an IBI structure is available for use. Furthermore, the gear-efficiency assessments made possible by the Rivers Survey have enabled a rigorous comparison of the benefit/cost aspects of passive sampling methods and electrofishing (Chapters 2 and 10). Clear opportunities exist to create large improvements in speed, cost-effectiveness and efficiency by the sole use of well-designed electrofishing studies.

Scale issues

The IBI can function at various scales because of the mobility of fish. Over time-scales ranging from daily to seasonal levels, Australian fish occupy habitat areas ranging from only a few metres for small-bodied benthic species such as river blackfish and gudgeons, up to one or two kilometres for larger, free-swimming predators such as Australian bass and Murray cod. Therefore, the fish-based IBI can be used to detect local, point-source disturbances and brief events. At larger scales, the life-cycles of many fish extend for decades, and few are short-lived (McDowall 1996). Similarly, most Australian fish are migratory over substantial distances (Harris and Mallen-Cooper 1994), with some travelling through entire river systems. These large-scale attributes mean that the fish-based index can provide useful IBI assessments at equally large spatial and temporal scales. This illustrates the need to understand the ways in which individual metrics respond to fish-community disturbances, so that the significance of these scale issues can also be understood. For example, at one extreme, building a dam which creates a major barrier to the passage of migratory species can influence IBI metrics throughout most of the system upstream. Alternatively, the effects on IBI assessments of an event causing extensive mortality among long-lived, slow-maturing fish - especially if they are not able to recolonise freely from neighbouring areas - may disappear slowly, long after the event has passed. At the other extreme, localised pollution incidents may require frequent IBI assessments to be detectable, as fish can rapidly recolonise the area when water quality is restored.

CONCLUSIONS

This test of the IBI based on fish communities of New South Wales rivers has shown that the index can repeatably discriminate relative environmental quality among widely different river reaches. The index's metrics responded to a broad range of environmental conditions and reflected known environmental impacts. The structure of the index successfully permitted standardised assessments across very different river types, river sizes and ecological regions and further refinement of the metrics can enhance this consistency. IBI assessments, like other river-health measures, plainly need to be interpreted in the light of knowledge about the site and its fauna. Having ready access to the individual metrics and data which support them provides the means to do this.

The index can be improved in future by collecting more data from high-quality sites to strengthen predictor metrics 1-5 and 11 for the Murray region. Sampling for IBI assessments should be limited to the warmer months. Metric 8 performed poorly and should be deleted, leaving an 11-metric index. This will require adjustment in future to IBI scores at any river reaches or sites where the present work is to be used as a baseline for monitoring. Appendix 1 lists all of the metric scores to permit such adjustments. Qualitative rankings for IBI scores derived from the remaining 11 metrics should follow a modification of the original scheme, giving values of Excellent 55-53, Good 47-43, Fair 39-35, Poor 29-23, and Very poor 17-11. The value of a suitable data transformation for Metric 11 should also be investigated, with the aim of reducing error in the prediction criteria caused by skewed data distributions largely caused by the need to include 'observed' abundance data. Better knowledge is needed on the tolerances and intolerances of the fauna to strengthen Metric 5. For coastal rivers, elimination from calculations of those species classed as essentially marine or estuarine would remove the tendency of Metric 1 to be biased towards such sites. For all of the six predictor metrics, it appears preferable to limit the criteria for IBI scores to those estimated for a catchment area of 10,000 km^2.

To complete testing the performance of the IBI in Australia, additional knowledge is needed on the accuracy, sensitivity and precision of the results of the IBI as related to other river-health assessment systems. The index needs to be more fully validated. The rapid biomonitoring approach of Chessman (1995) and others (e.g. Norris and Norris 1995), and the AUSRIVAS project based on the British RIVPACS model (Wright 1995) using macroinvertebrates as indicators of water quality, provide opportunities for comparisons. Work is planned to make large-scale direct and indirect comparisons between assessment results from AUSRIVAS and the IBI at related sites (river-reach level) and times (seasons) to determine the correspondence between broad patterns of relative environmental health. These large spatial and temporal scales have been shown to be sometimes more influential than local, short-term ones (Roth *et al.* 1996; Allen *et al.* 1997). This comparison will be based on the assumption that river-health changes illustrated by such indicators are likely to occur at relatively large scales, so that small-to-moderate-scale differences in the exact time and place of sampling may not significantly alter assessments. This assumption will be specifically tested by making replicated IBI assessments at small spatial and temporal scales in a set of New South Wales rivers, and by direct assessment of IBI and RIVPACS results at consistent sites and times.

IBI assessments also need to be compared with results from independent tests of river or catchment health, and this can be done for NSW Rivers Survey sites. Such measures as the cultural pollution index (Leonard and Orth 1986), human population density, or agricultural development indicators at the catchment scale (e.g. Allen *et al.* 1997) need to be assessed for the particular catchments, and their results related to IBI outputs to provide more knowledge on the accuracy and sensitivity of the IBI using Australian fish communities. Comparisons can also be made with the results of multivariate assessments using fish and habitat data from the Rivers Survey (Chapter 3) to evaluate the levels of agreement and understanding achieved by the two different approaches. The general repeatability of IBI assessments over longer time scales provides another available test of validity, as does the use of assessments at known-impact sites.

IBI scores for given sites are always relative to one another and have no absolute meaning (Karr *et al.* 1986). The value of the present work lies in the capacity to make judgements of spatial and temporal changes in the relative quality of rivers within and among the four regions. The results show that rivers in the Murray region are more severely degraded than those of the Darling, North Coast and South Coast regions. A large proportion of montane rivers in New South Wales coastal regions are also in degraded condition relative to other river types in the same regions. Future cross-calibration of the IBI with independent measures of catchment condition and river health, together with tests of repeatability and response to known impacts, will enable validation of the index's accuracy, sensitivity and robustness in assessing river health. The main immediate values of the present IBI results lie in their capacity to provide a baseline for monitoring river health, and an efficient and sensitive means of checking the relative condition of any other river site in New South Wales.

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REFERENCES

Allen, J.D., Erickson D.L., and Fay, J. (1997). The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* **37**, 149-161.

ANZECC (1992). *National Water Quality Management Strategy. Australian Water Quality Guidelines for Fresh and Marine Waters.* Australian and New Zealand Environmental and Conservation Council, Canberra.

Arthington, A. H. (1985). The biological resources of urban creeks. *Australian Society for Limnology Bulletin,* **10**, 33-40.

Australia State of the Environment (1996). Report to the Commonwealth Minister for the Environment by the State of The Environment Council. CSIRO, Melbourne.

Callinan, R.B and Fraser, G.C. (1989). Pathology of red spot disease in sea mullet, *Mugil cephalus* L., from eastern Australia. *Journal of Fish Diseases,* **12,** 467-479.

Chessman, B.C. (1995). Rapid assessment of rivers using macroinvertebrates: A procedure based on habitat-specific sampling, family level identification and biotic index. *Australian Journal of Ecology*. **20**, 122- 129.

Cowx, I. G. (ed.) (1991). *Catch Effort Sampling Strategies*. Fishing News Books, Oxford.

Cranston, P.S., Fairweather, P. and Clarke, G. (1996) Biological indicators of water quality. In: *Indicators of Catchment Health. A Technical Perspective*. (J. Walker and D.J. Reuter, Eds.) CSIRO, Australia.

CSIRO (1992). *Towards Healthy Rivers*. CSIRO Division of Water Resources Consultancy Report No. 92/44.

Fausch, K.D., Karr, J.R. and Yant, P.R. (1984) Regional application of an index of biotic integrity based on stream-fish communities. *Transactions of the American Fisheries Society,* **113**, 39-55.

Fore, L.S., Karr, J.R. and Conquest, L.L. (1994). Statistical properties of an index of biotic integrity used to evaluate water resources. *Canadian Journal of Fisheries and Aquatic Sciences,* **51** (5), 1077-1087.

Gehrke, P.C., Brown, P., Schiller, C.B., Moffatt, D.B., and Bruce, A.M. (1995). River regulation and fish communities in the Murray-Darling River system, Australia. *Regulated Rivers: Research and Management.* **11**, 363-375.

Gehrke, P. C. and Harris, J.H. (1996). *Fish and Fisheries of the Hawkesbury- Nepean River System*. Final Report to the Sydney Water Cooperation, July 1996. NSW Fisheries and Cooperative Research Centre for Freshwater Ecology, Sydney.

Growns, I.O., Pollard, D.A, and Gehrke, P.C. (1997). Changes in riverine fish associated with vegetated degraded banks, upstream of and within nutrient enriched zones. *Fisheries Management and Ecology* (in press).

Harris, J.H. (1995). The use of fish in ecological assessments. *Australian Journal of Ecology,* **20** (1), 65-80.

Harris, J.H., and 25 others. (1996). The NSW Rivers Survey. Final report, Part 1 - Data, June 1996. NSW Resource and Conservation Assessment Council - Project G8. NSW Fisheries Research Institute and The Cooperative Research Centre for Freshwater Ecology.

Harris, J. H. and Mallen-Cooper, M. (1994). Fish-passage development in rehabilitation of fisheries in mainland south-eastern Australia. In: *Rehabilitation of Freshwater Fisheries.* (Cowx, I.G. Ed.). (Fishing News Books: Oxford). pp 185-193.

Harris, J.H. (1984). Impoundment of coastal drainages of south-eastern Australia and a review of its relevance to fish migration. *Australian Zoologist,* **21**(3), 235-50.

Harris, J.H. (1988). Demography of Australian bass, *Macquaria novemaculeata* (Perciformes, Percichthyidae) in the Sydney Basin. *Australian Journal of Marine and Freshwater Research* **39**, 355-69.

Hynes, H.B.N. (1970) *The Ecology of Running Waters.* Liverpool University Press, Liverpool.

Karr, J. R. (1981). Assessments of biotic integrity using fish communities. *Fisheries (Bethesda).* **6**, 21-7.

Karr, J.R. (1987). Biological monitoring and environmental assessment: a conceptual framework. *Environmental Management* **11**, 249-56.

Karr, J. R. , Fausch, K.D., Angermeir,, P.L, Yant, P.R. and Schlosser, I.J. (1986). Assessing biological integrity in running waters. A method and its rationale. *Illinois Natural History Survey Special Publication.* **5**,28

Lake, P.S. (1982). The relationships between freshwater fish distribution, stream drainage area and stream length in some streams of south- eastern Australia. *Australian Society for Limnology Bulletin,* **8**, 31-7.

Leonard, P.M. and Orth, D.J. (1986). Application and testing of an Index of Biotic Integrity in small, coolwater streams. *Transactions of the American Fisheries Societyy,* **115,** 401-14.

McDowall, R.M. (1981) The relationships of Australian freshwater fishes. In: *Ecological Biogeography of Australia.* (A. Keast, Ed.) Junk, The Hague.

McDowall, R.M. (Ed.) (1996). *Freshwater Fishes of South-eastern Australia.* (2nd edition) Reed Books, Sydney Australia.

Mallen-Cooper, M. (1994) How high can a fish jump? *New Scientist.* 142 (1921), 32-37.

Mallen-Cooper, M., Stuart I., Hides-Pearson, F. and Harris, J. (1995). *Fish Migration in the Murray River and Assessment of the Torrumbarry Fishway.* Final Report for MDBC. NSW Fisheries.

Norris, R.H. and Norris, K.R. (1995) The need for biological assessment of water quality: Australian perspective. *Australian Journal of Ecology*, **20**, 1-6.

Paxton, J. R. , Hoese, D.F., Allen, G.R. and Hanley, J.E. (1989). *Zoological Catalogue of Australia. Vol. 7, Pisces.* AGPS, Canberra.

Pollard, D.A., Growns, I.O. and Harris, J.H. (1994). The effects of treated sewage-related water quality changes on the distributions and abundances of fish in the middle reaches of the Hawkesbury-Nepean River system. In: Pollard D.A., Growns I.O., Pethebridge R.L. and Marsden T.J. Hawkesbury-Nepean Fish Ecology Study Six-monthly report to the Sydney Water Board. (N.S.W. Fisheries Research Institute, Sydney.)

Puckridge, J. (1986). The reproductive biology of bony bream in the lower Murray. *Proceedings 13th Annual ASFB Conference, Darwin 1986.* p 42.

Roberts, J. and Sainty, G. (1996). *Listening to the Lachlan.* Murray-Darling Basin Commission, Canberra.

Roth, N.E., Allen, J.D., and Erickson, D. (1996). Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology*, **11**, 141-156.

Steedman, R. J. (1988). Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. *Canadian Journal of Fisheries and Aquatic Sciiences,* **45**, 492-501.

Suter, II G.W. (1993). A critique of ecosystem health concepts and indexes. *Environmental Toxicology and Chemistry.* **12**, 1533-1539.

Thorncraft, G.A. and Harris, J.H. (1996). *Assessment of Rock-Ramp Fishways.* May 1996, 39 pp. (NSW Fisheries Research Institute and Cooperative Research Centre for Freshwater Ecology.

Underwood, A.J. (1981). Techniques of analysis of variance in experimental marine biology and ecology. *Oceanography and Marine Biology Annual Reviews,* **19**, 513- 605.

Wright, J.F (1995). Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. *Australian Journal of Ecology,* **20** (1), 181-197.