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Estimating species richness and catch per unit effort from boat electro-fishing in a lowland river in temperate Australia

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Abstract Biodiversity estimates are typically a function of sampling effort and in this regard it is important to develop an understanding of taxon-specific sampling requirements. Northern hemisphere studies have shown that estimates of riverine fish diversity are related to sampling effort, but such studies are lacking in the southern hemisphere. We used a dataset obtained from boat electro-fishing the fish community along an essentially continuous 13-km reach of the Murrumbidgee River, Australia, to investigate sampling effort effects on fish diversity estimates. This represents the first attempt to investigate relationships between sampling effort and the detection of fish species in a large lowland river in Australia. Seven species were recorded. Species-specific patterns in catch per unit effort were evident and are discussed in terms of solitary and gregarious species, recreational fishing and the monitoring of rare and threatened species. There was a requirement to sample substantial lengths of river to describe total species richness of the fish community in this river reach. To this end, randomly allocated sampling effort and use of species richness estimators produced accurate estimates of species richness without the requirement for excessive levels of effort. Twenty operations were required to estimate species richness at this site, highlighting the need for comparable studies of river fish communities in lowland rivers elsewhere in Australia and the southern hemisphere.

Key words: Australia, electro-fishing, lowland river, sampling effort, species richness, species richness estimator.

INTRODUCTION

Species diversity estimates are typically a function of sampling effort and it is important to develop an understanding of taxon-specific sampling requirements (Gotelli & Colwell 2001; Chao *et al.* 2005). Fishes inhabit a range of different ecosystems and habitats, requiring a variety of sampling strategies to facilitate assessment of diversity. Accurate estimates of the diversity and abundance of fishes can be difficult to obtain in large lowland rivers where habitat characteristics including variable depth, complex woody structure, heterogeneous habitat, flow and turbidity limit the effectiveness of sampling gear (Thévenet & Statzner 1999). Species-specific traits that affect detection and the spatial distribution of different species compound the problem (Fausch *et al.* 2002). Currently, boat electro-fishing represents one of the more useful means of surveying fishes in these habitats (e.g. Harvey & Cowx 1996). Single-pass sampling often forms the basis of routine monitoring with this technique (e.g. Odenkirk & Smith 2005). Priorities in developing a methodology for monitoring programmes based on single-pass electro-fishing include determining the length of river to survey at a site (Lyons 1992; Patton *et al.* 2000; Hughes *et al.* 2002), an issue intimately related to representative sampling (Cao *et al.* 2002). Consideration of the spatial scale of sampling is a major issue for monitoring studies and has only recently been addressed in stream ecology (Downes *et al.* 2002). The length of stream sampled has a strong bearing on levels of precision and accuracy (Downes *et al.* 2002). We know little about the physical distances at which fish community data are auto-correlated and have no data on which to determine how far apart fish community samples need to be to ensure statistical independence in ecological studies

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(Downes *et al.* 2002). As a result, the spatial scale at which fish communities are studied in this region are often arbitrary or based entirely on logistical considerations.

It has been demonstrated that both the length of stream sampled and the sampling effort applied at a site influence the accuracy of fish diversity estimates (Lyons 1992; Hughes *et al.* 2002; Smith & Jones 2005) as a function of: (i) the longitudinal accumulation of microhabitats encountered with increasing stream length; and (ii) the distribution of rare species (Cao *et al.* 2001; Kennard *et al.* 2006). Therefore, the relationship between species accumulation and the length of stream surveyed (and/or sampling effort) can be used to determine the minimum stream length to survey at a site in achieving the desired objectives of particular monitoring applications (Cao *et al.* 2001). The minimum length of stream required to obtain accurate estimates of fish species richness at a site using boat and backpack electro-fishing has been investigated, particularly in temperate North America (Lyons 1992; Hughes *et al.* 2002; Meador 2005; Smith & Jones 2005). In many instances, substantial stream length must be surveyed to estimate species richness (Lyons 1992; Cao *et al.* 2001; Hughes *et al.* 2002; Smith & Jones 2005). Notably, Hughes *et al.* (2002) concluded that reliable estimates of 95% of the species richness required sampling from a river reach equalling 85 times the mean wetted channel width, while 100% of species were detected within 300 channel widths.

In Australia, sampling comparable lengths of stream to that proposed by Hughes *et al.* (2002) has not been undertaken, although, the effects of sampling effort and stream length have been examined for shorter lengths of stream (Faragher & Rodgers 1997; MDBC 2004; Kennard *et al.* 2006) for stream lengths of <1 km. In temperate Australia, riverine fish communities frequently contain low species richness (Figs 2,3 in Gehrke & Harris 2000). Meador (2005) commented that surveying 500–1000 m of stream might produce better estimates of species richness in low diversity fish communities (<10 species) compared with species-rich communities. Routine monitoring (e.g. by fisheries agencies) is frequently conducted over river lengths measuring hundreds of metres to 1 km (http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html). Building on the approach of Harris and Gehrke (1997), the Sustainable Rivers Audit in the Murray–Darling Basin (MDBC 2004; Lintermans *et al.* 2005) is starting to develop approaches to monitoring riverine fish communities at the scale of catchments and basins but based on data collected at relatively small scales (i.e. sites of up to 1 km in length). Data collected in the current study provide a unique opportunity to assess species richness estimates and catch per unit effort (CPUE) at a scale one order of magnitude greater than that typically used in fish monitoring studies in Australia. Subsequently, the current study provides an assessment of the appropriate extent of a site for fish community sampling design.

In this study we aimed to: (i) develop an understanding of the relationship between sampling effort and representative samples of fish species richness; (ii) determine the sampling effort required to obtain precise estimates of the CPUE for individual species; and (iii) examine the spatial distribution of the fish community at a site, within a lowland river reach in temperate Australia.

METHODS

Study site

The study was undertaken in a lowland reach of the Murrumbidgee River, near Narrandera, in southern inland New South Wales, Australia (Fig. 1). River redgum (*Eucalyptus camaldulensis* Dehnh.) was conspicuous along much of the riparian zone of the river and channel widths were usually in the order of 70 m (Growns *et al.* 2004). River depths of 1–2 m were typical throughout the study area, with depths of 3–5 m commonly encountered on outside bends. The dominant in-stream habitats were bare sand or mud interspersed with fallen trees or branches (particularly of *E. camaldulensis*).

Fish sampling

We surveyed the fish community from 11–24 September 2003 over an almost continuous 13-km length of river (excluding three short sections of shallow, sand bar that could not be navigated), involving 157 replicate boat electro-fishing operations. We used a 4.5-m aluminium hull electro-fishing boat fitted with a 7.5 kW Smith-Root© Model GPP 7.5 H/L electro-fishing unit. Two anodes were suspended from booms mounted on the bow of the boat and a cathode was mounted along each side of the hull. The electro-fisher was operated at 1000 volts DC, 60 Hz, 35% of range with an average current draw of 4.5 amps. Each replicate operation was undertaken for 5 min (elapsed time), equating to a mean (±SE) of 92 ± 0.35 s of applied power. All navigable habitats were surveyed in proportion to availability by criss-crossing the channel and continuing in an overall upstream direction (to prevent re-sampling habitats). Each operation covered a distance of approximately 80 m of river. Two operators collected immobilized fish from the bow of the boat by dip net. A support boat followed approxi-

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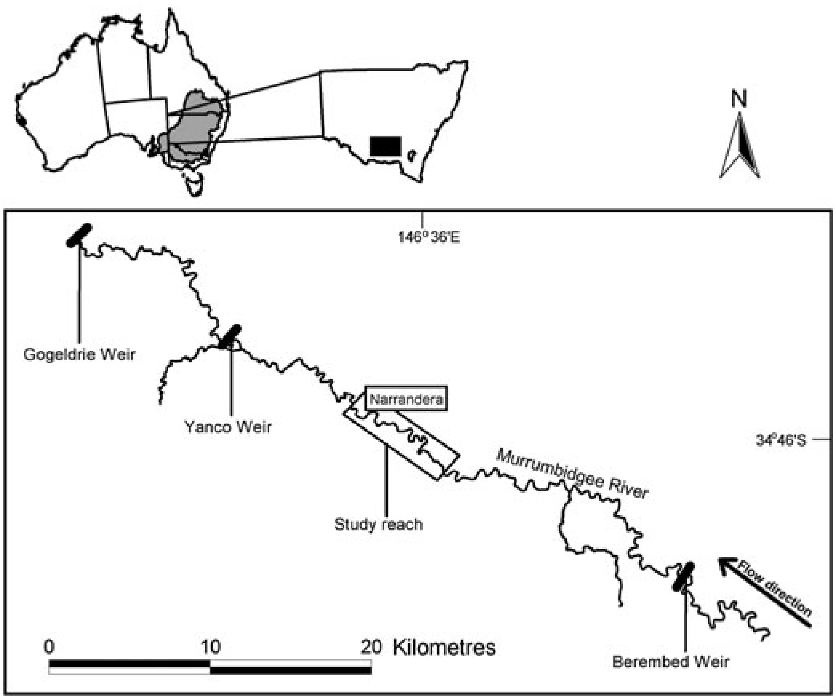

Fig. 1. Study reach in the Murrumbidgee River near Narrandera, New South Wales, Australia.

Table 1. Summary of the fishes surveyed in the study (the number of caught and observed individuals was combined)

Fish species	Catch
Carp *Cyprinus carpio* Linnaeus	204
Trout cod *Maccullochella macquariensis* (Cuvier)	131
Golden perch *Macquaria ambigua* (Richardson)	45
Australian smelt *Retropinna semoni* (Weber)	41
Murray cod *Maccullochella peelii* (Mitchell)	15
Unidentified cod *Maccullochella* spp.	6
Silver perch *Bidyanus bidyanus* (Mitchell)	4
River blackfish *Gadopsis marmoratus* (Richardson)	4

mately 50 m downstream of the electro-fishing boat to ensure that stunned fishes and particularly the endangered trout cod *Maccullochella macquariensis* (Cuvier) that were slow to surface were collected and fully recovered.

We identified all captured individuals to species and returned them to the river following recovery. The number of fish that were observed while electro-fishing, but not captured was also recorded. These individuals were identified to species except on six occasions when individuals were identified to genus but could not be distinguished as being either *M. macquariensis* or Murray cod *Maccullochella peelii* (Mitchell). Unidentified individuals of *Maccullochella* are included in a summary of the survey (Table 1) but were not included in analyses. Data from observed and caught fishes were pooled.

Water temperature, conductivity and turbidity were recorded at 15-min intervals from a single location in the middle of the study reach using a Hydrolab (Loveland, CO).

Data analysis

A number of approaches were used to investigate spatial patterns in species richness and CPUE of each species. This involved: (i) examining sequences of operations in which species were not detected; (ii) calculating confidence limits associated with mean CPUE of each species based on random and consecutive re-sampling of data; (iii) checking for correlation between species; (iv) testing for similarity in species richness among sub-reaches (spatial auto-correlation); and (v) assessing the performance of species richness estimators in relation to sampling effort.

Record was made of all sequences of consecutive operations where a species was not detected. These data were summarized as mean, minimum and maximum sequences in terms of number of operations. Where a sequence included the beginning or end of the 13-km reach (i.e. operation 1 or operation 157), a potential underestimate occurred (since potential catches associated with operations beyond the 13-km reach were unknown). Nevertheless, these sequences were retained in calculating summary statistics as their removal caused a greater bias in the

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output (lengthy sequences with nil capture of some species were associated with the extremities of the study reach).

To determine the minimum sampling effort required to produce CPUE (of each species) within narrow confidence limits, two practical sampling strategies were simulated in this study. In the first strategy, samples were taken by drawing groups of consecutive operations – for example, a five-operation sample could consist of operation 83, 84, 85, 86 and 87. This mimicked a common method of survey (see references in http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html). The second strategy was based on drawing random samples of operations collected over the full 13-km study reach – for example, a five-operation sample comprising operation 3, 9, 41, 83 and 129. A macro written in Microsoft Excel 2000 sub-sampled data at one-operation increments either based on the consecutive field sampling order or based on a random order, repeating this process 1000 times for sample sizes ranging from 2 to 75 (i.e. up to approximately half the sample). Ninety-five percent confidence limits were generated for each sample size by taking the 2.5 and 97.5 percentiles. These confidence limits were not spaced evenly from the mean, as data did not follow a normal distribution owing to the large number of zero values in the data set.

We investigated possible interrelationships between species by correlating abundance within operations, based on data standardized to catch-per-minute of electro-fishing (power-on-time). Due to the non-normal distribution of data, Spearman Rank correlation (Sokal & Rohlf 1995) was used and the abundance of each species ranked across operations. Given that multiple comparisons were undertaken, a Bonferroni correction (Sokal & Rohlf 1995) was applied to reduce the experiment-wise Type I error rate. With $k = 21$ comparisons, the significance level equivalent to $\alpha = 0.05$ was $P = 0.002$.

Analysis of fish community structure was undertaken to determine whether fish communities within neighbouring parts of the reach were more similar than those from more distant parts of the 13-km reach (equivalent to an auto-correlation). Two analyses were undertaken. The first used the average of five consecutive operations (i.e. operations 1–5, 6–10, 11–15 etc.) and the second used the average of 10 consecutive operations (i.e. operations 1–10, 11–20, 21–30 etc.). Analysis based at a resolution of individual operations could not be undertaken owing to the substantial variability among operation-specific data, and the predominance of zeros in the data set. Analyses were performed on standardized catch-per-minute of electro-fishing on-time. Similarity matrices were created using Primer 5.1.2 (Plymouth Marine Laboratory). Data were square root transformed to make the influence of common and rare species more similar within the analysis. Similarities between fish assemblages within each group of five or 10 consecutive operations were calculated using the Bray-Curtis similarity measure (Bray & Curtis 1957).

A second matrix was created which represented the spatial proximity of the 31 grouped samples (each the average of five consecutive operations) and 15 grouped samples (10 consecutive operations) (adjacent grouped samples having a proximity value of 1, grouped samples separated by one reach having a proximity value of 2, grouped samples separated by two reaches having a proximity of 3 etc.). The 'proximity' matrix was correlated with the Bray-Curtis similarity matrix of sampling data using the RELATE function in Primer, with 999 permutations. This function uses the Spearman Rank correlation to relate corresponding values in the two matrices.

To determine an appropriate level of sampling for estimating species richness, we calculated three estimators: Jackknife 1 and 2 (Burnham & Overton 1979) and Chao 2 (Chao 1987). These estimators were calculated based on simulated consecutive and random sampling strategies. Data were re-sampled with replacement at five-sample increments, from 5 to 75 samples, with this process repeated 1000 times for each sample size. An additional comparison, the average species richness from re-sampled data, was used. The average species richness was not based on an estimator (it was the average number of species recorded in relation to sampling effort) and provided a reference against which the three estimators could be compared.

RESULTS

During the sampling period daily water temperature was 14.4 ± 0.05°C (mean ± SE), conductivity was 180 ± 0.4 μS cm^{-1} and turbidity was 15.0 ± 0.3 NTU. Seven fish species were captured or observed in this study (Table 1). The alien species carp *Cyprinus carpio* (204 individuals) and the native trout cod, *M. macquariensis* (131 individuals) were frequently encountered. The former was schooling or solitary, whereas, *M. macquariensis* was usually solitary and only occasionally found with more than a single individual associated with a single wood structure. Six *Maccullochella* spp. were observed though not identified to species level. Two species (river blackfish *Gadopsis marmoratus* and silver perch *Bidyanus bidyanus*) were rarely encountered (Table 1).

Detection of species

The sampling effort required to detect different species was variable, species-specific and generally a

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Table 2. Species-specific non-detection over 157 boat electro-fishing operations in a 13 km reach of the Murrumbidgee River

Species	Sequence (consecutive operations of non-detection)			Number of sequences
	Mean (±SE)	Min	Max	
Cyprinus carpio	2 ± 0.2	1	7	39
Maccullochella macquariensis	2 ± 0.3	1	8	33
Macquaria ambigua	4 ± 0.9	1	18	28
Maccullochella peelii	10 ± 4	2	49	14
Retropinna semoni	15 ± 5	1	44	10
Bidyanus bidyanus	31 ± 7	9	53	5
Gadopsis marmoratus	39 ± 17	7	74	4

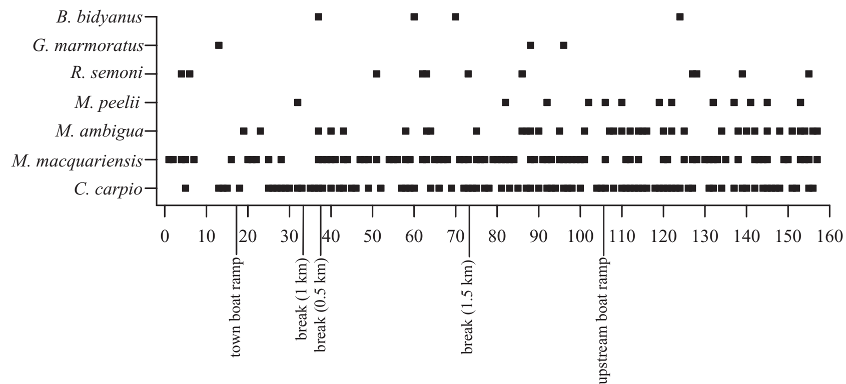

Fig. 2. The presence of each species (■) detected in the study reach from each electro-fishing operation. Operations occurred from downstream (operation 1) to upstream (operation 157) in an almost continuous reach of river, except for three breaks that could not be navigated by boat involving 1, 0.5 and 1.5 km of river, respectively.

function of the frequency of capture (Table 2, Fig. 2). In regard to the least common species, the mean number of consecutive operations (±1 SE) where *G. marmoratus* and *B. bidyanus* were not detected was 39 ± 17 and 31 ± 7, respectively. The maximum number of operations between detection was substantially greater than that used in standard surveys (44–74 operations) for four of the seven species (Table 2, Fig. 2; http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html).

Catch per unit effort

Data re-sampling revealed that the confidence limits of the mean catch per operation of each species differed according to whether a consecutive or random strategy was used. Convergence of confidence limits began to occur for each of the seven species within the first 10 operations for both consecutive and random re-sampling (Fig. 3). In the case of *C. carpio* and *M. macquariensis*, confidence limits converged quicker at smaller sample sizes for the random compared with the consecutive re-sampling strategy (Fig. 3(a) and (b), respectively). Beyond 30 consecutive operations little convergence in confidence limits occurred, whereas confidence limits continued to converge after 70 operations based on random re-sampling. For *Macquaria ambigua* and *M. peelii* (Fig. 3(c) and (f), respectively) convergence of confidence limits generally occurred at comparable sample sizes for both consecutive and random re-sampling strategies. However, in the case of *G. marmoratus*, *Retropinna semoni* and *B. bidyanus* (Fig. 3(d), (e) and (g), respectively), the convergence of confidence limits occurred at smaller sample sizes when data were re-sampled using the consecutive strategy.

Correlation of species within operations

Of the 21 possible inter-species correlations, none identified significant relationships between any two species. Only one pair of species, *C. carpio* and *M. ambigua* had a significant association ($r = 0.21$, $P = 0.008$); however, this relationship was not significant when Bonferroni correction was applied.

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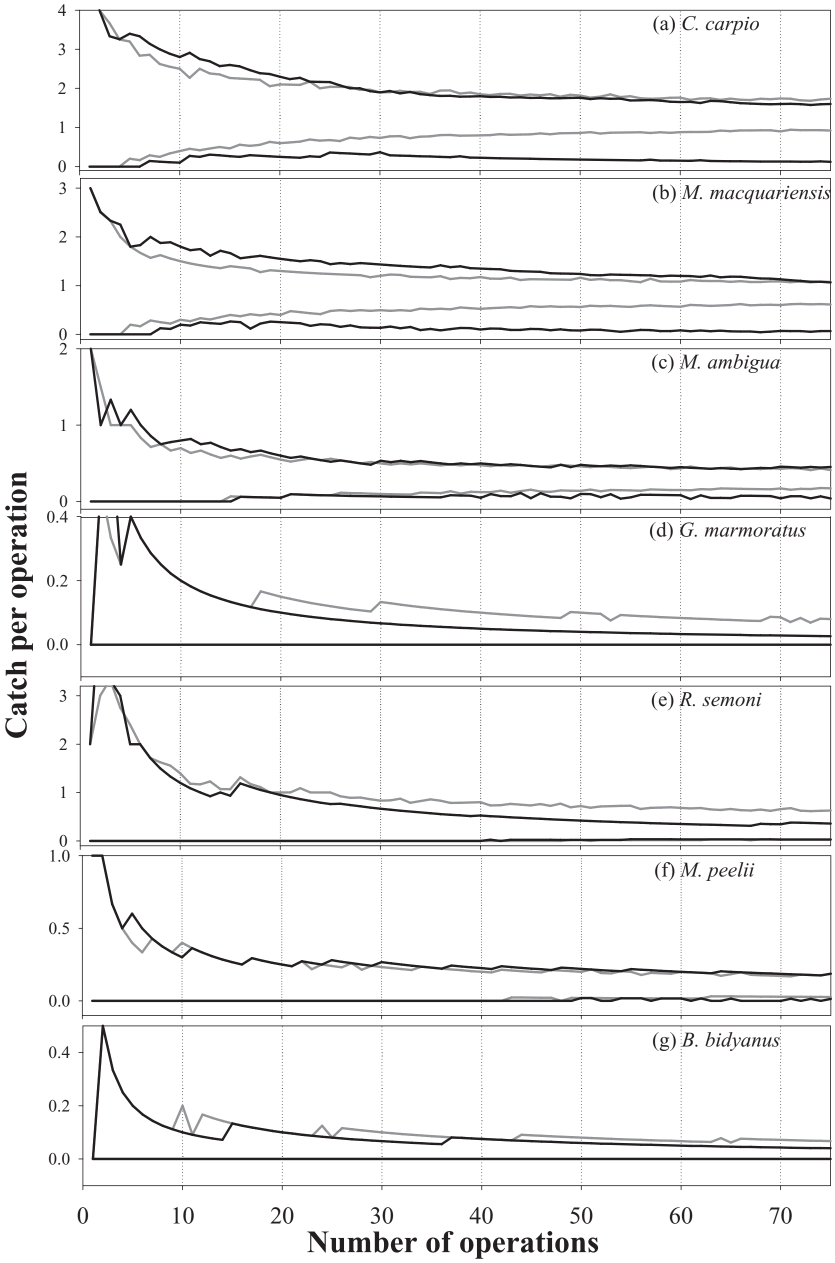

Fig. 3. The effect of sample size and re-sampling strategy on 95% confidence limits of mean catch for (a) *Cyprinus carpio*; (b) *Maccullochella macquariensis*; (c) *Macquaria ambigua*; (d) *Gadopsis marmoratus*; (e) *Retropinna semoni*; (f) *Maccullochella peelii*; and (g) *Bidyanus bidyanus*. The upper and lower confidence limits are plotted as consecutive (black) and random (grey) re-sampling strategies.

Analysis of fish community structure among grouped samples

There were significant relationships between fish community composition and proximity of grouped samples when analysed at the scale of five consecutive operations (Global $R = -0.196$, $P = 0.001$) and 10 consecutive operations (Global $R = -0.279$, $P = 0.010$). This suggests that there was significant clustering among grouped samples (i.e. the fish assemblage in neighbouring parts of the reach were more similar than those across distant parts of the reach).

Estimating species richness

Based on re-sampling of the entire dataset, 25–35 consecutive operations were required to estimate the total

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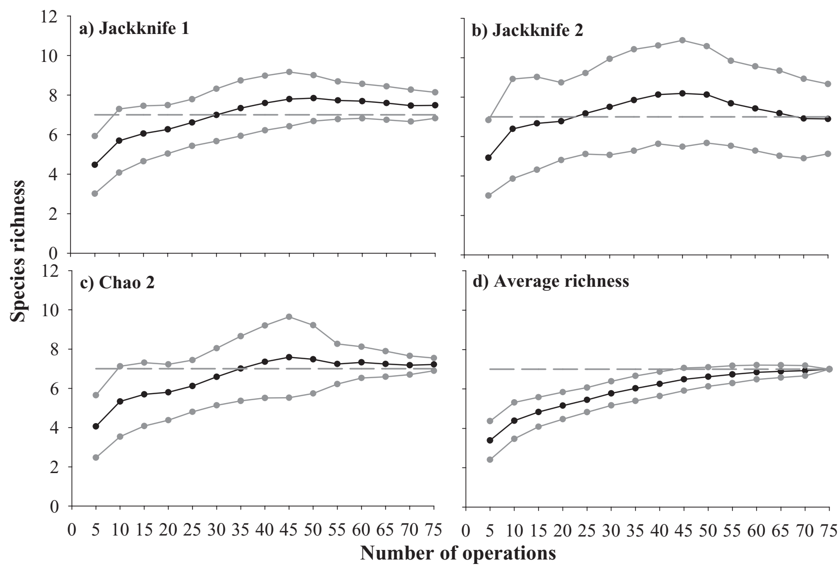

Fig. 4. The estimated species richness of the study reach based on a consecutive re-sampling strategy (use of samples from spatially contiguous operations) using three species richness estimators (a–c). The average richness shown in (d) represents the average number of species against sampling effort without using an estimator. The means (±SD) are calculated based on 1000 iterations (from *n* samples) and are represented by black and grey lines, respectively. The dashed line represents the total species richness detected in this study from 157 operations.

species richness (i.e. to approach the asymptotic value of that achieved from 157 operations) (Fig. 4). Sampling based on consecutive operations beyond this level of effort led to overestimation of total species richness with all estimators (>35 Jackknife 1, 30–60 Jackknife 2, 40–50 Chao 2), although, use of Jackknife 2 and Chao 2 resulted in comparable estimates of the true species richness by 65 and 55 operations, respectively (see asymptote of species richness curves in Fig. 4). In comparison, the random re-sampling strategy estimated total species richness from 20 operations and thereafter consistently tracked the total species richness (Fig. 5).

DISCUSSION

Estimating species richness at a site

Length of stream surveyed and survey effort were shown to affect the detection of rare species and therefore the estimates of total species richness in this study. The finding parallels that of North American studies in revealing that a substantial level of sampling effort must be applied within a threshold extent of river to obtain representative estimates of fish diversity (Cao *et al.* 2001; Hughes *et al.* 2002; Smith & Jones 2005).

The current study investigated this issue under Australian conditions where boat electro-fishing has typically been applied at river sites for watershed-level sampling (Smith & Jones 2005) for the purpose of making spatial comparisons or comparing representative samples of Australian fish communities through time or for estimating river health (Harris & Gehrke 1997; MDBC 2004). In these programmes, effort in the order of 8–15 replicate boat electro-fishing operations represents standard practice at a site, with operations ranging from 2–5 min of elapsed-time (http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html). This level of effort is usually applied to up to 1 km of river (http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html) and species richness estimators have not been used. While this approach is used for reporting on the health of fish communities over a large extent through time or for identifying the possible effects of large scale impact (e.g. river regulation), results from the current study indicate that this level of effort is unlikely to provide an accurate estimate of species richness or a precise estimate of CPUE at the smaller scale of a river reach (comparable with a site).

In the current study, we achieved an accurate assessment of species richness at a site by: (i) increasing the standard electro-fishing effort to at least 20 operations positioned randomly and comprising operations in the order of 90 s on-time (comparable with 5 min elapsed-time); and (ii) applying a species richness estimator. Data sub-sampling demonstrated that an accurate estimate of species richness could be obtained from a moderate increase in sampling effort compared with

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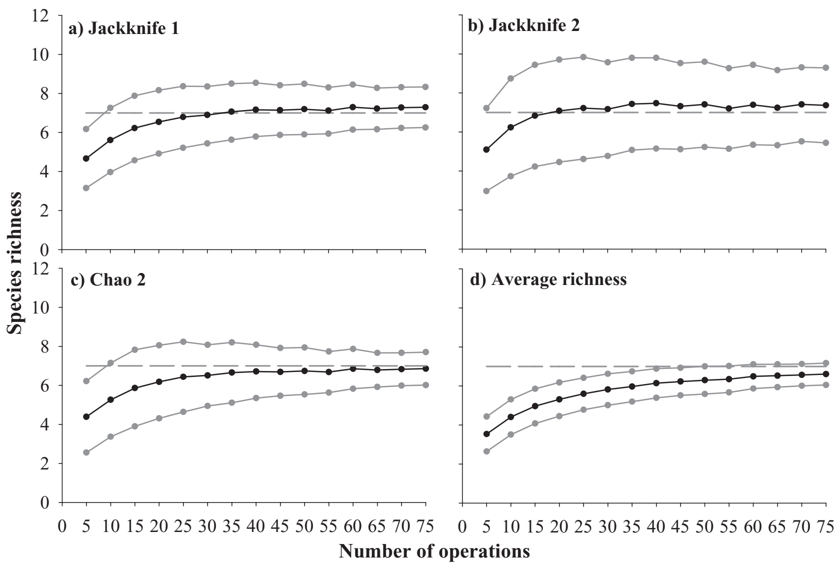

Fig. 5. The estimated species richness of the study reach based on a random re-sampling strategy using three species richness estimators. The means (±SD) are calculated based on 1000 iterations (from *n* samples) and are represented by black and grey lines, respectively. The dashed line represents the total species richness detected in this study from 157 operations.

that more commonly used in watershed-level sampling in the Murray–Darling Basin (http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html). A secondary issue (besides increasing sampling effort) was the spacing of operations at a site. Where an accurate estimate of species diversity is required, our comparison of consecutive (current practice by state agencies) and random sampling strategies showed that the latter provides a superior basis for accurately estimating total species richness in a reach (compare Fig. 4 and Fig. 5). In particular, estimates levelled more rapidly with random as opposed to consecutive sampling (Figs 4,5). This reflected a level of auto-correlation in the data. Specifically, sampling in adjacent areas of a study reach (represented by analysis of consecutive operations) was found to increase the likelihood of repeatedly sampling the same subset of the fish community. By positioning each replicate operation within a study area randomly, the likelihood of sampling alternative representations of the fish community present within the study area is increased. As a result, randomly distributing the sampling locations of individual replicate operations within a study area is more likely to collect a broader representation of the structure of a fish assemblage present, and consequently, is more likely to optimize the number of taxa collected.

While Jackknife 1 and Chao 2 required slightly more operations to provide an accurate species richness estimate (about 25), the Jackknife 2 estimator attained this earlier, but showed consistently higher variation in relation to increasing sampling effort (Fig. 5). The optimal estimator is likely to be river-specific, as the number or proportion of species with a total catch of 1 or 2 or that occur in only one or two operations leads to estimator specific bias (Cao *et al.* 2004). The transferability of these findings to other sites, rivers and times should be resolved in developing an appreciation of the effect of sampling effort on estimates of fish species richness in Australian lowland rivers. However, the usefulness of species richness estimators is not in dispute. Estimators outperformed raw data in this investigation of a low diversity fish community (Fig. 5), and we join other ecologists (e.g. Gotelli & Colwell 2001; Hughes *et al.* 2002) in recommending estimators to maximize benefit from community survey data.

To determine the composition of a fish community at a site, a level of survey effort in excess of that recommended here for estimating total species richness is likely to be required. This is typical in community surveys (Cao *et al.* 2001, 2002; Hughes *et al.* 2002). In this regard, the maximum number of consecutive operations required to detect each of four species (*G. marmoratus*, *B. bidyanus*, *M. peelii* and *R. semoni*) ranged from 44 to 74 (Table 2). This highlights the risk of not detecting species at a site by boat electro-fishing when applying typical levels of survey effort in Australian rivers (http://www.ecolsoc.org.au/What%20we%20do/Publications/Austral%20Ecology/AE.html). The amount of effort required to obtain an estimate of true species richness at a site in Australian systems is likely to be river-specific, habitat-specific and dependent on temporal scale if North American studies serve as a guide (e.g. Hughes *et al.* 2002). Species richness estimates from a number of riverine sites are required before the transferability of

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this finding can be appreciated in the Australian context. In turn, this has the capacity to influence the design of watershed-level fish community sampling programmes (e.g. MDBC 2004), in addition to improving localized studies within river reaches (e.g. where a threatened species exists as a small population fragment).

Fish community distribution

Inter-species correlations revealed no significant relationships between any two species when analysed at the scale of the smallest sampling unit in this study (operation). However, lowland Murray–Darling Basin fishes are known to exhibit species specific differences in habitat-use at finer scales than was examined in the current study (Koehn 1997; Koehn & Nicol 1998; Crook *et al.* 2001; Boys & Thoms 2006; Nicol *et al.* 2007) and this may be partly explained by inter-specific interactions and/or habitat preferences (Schlosser 1982). In the current study, either these mechanisms were unimportant or were occurring at scales that we did not investigate.

Catch per unit effort

The amount of sampling effort required to accurately estimate CPUE was species-specific in the current study. Mean CPUE of species that were infrequently caught (*B. bidyanus*, *G. marmoratus*, *M. peelii*) could not be distinguished from nil catch based on lower confidence limits (Fig. 3). In contrast, the lower confidence limits were above zero for the three most abundant species (Fig. 3). Variation in estimates of CPUE was a function of the total count of a species, and was consistent with the findings of others (Cao *et al.* 2002). Patterns of aggregation can also affect estimates of CPUE (Pennington & Volstad 1994). For instance, despite similar total catch of *M. ambigua* ($n = 45$) and *R. semoni* ($n = 41$), the CPUE of the latter was less precise (Fig. 3). *Retropinna semoni* was only detected in 11 operations in the current survey and was abundant when found, leading to high variation in estimates of CPUE (Fig. 2). In contrast, *M. ambigua* was found in 36 operations and occurred in low abundance within operations, leading to relatively less variation in estimates of CPUE (Fig. 2).

Of the three species for which CPUE was estimated with a high level of precision, two species (*C. carpio* and *M. ambigua*) are widespread and abundant in the Murray–Darling Basin (Lintermans 2007). Widespread monitoring of the latter two fishes, comprising an endemic and an introduced species, potentially represents a useful and reliable means of quantifying changes in aquatic ecosystems within the Basin in relation to adaptive management exercises. In contrast, CPUE of the recreational angling species *M. peelii* reflects: (i) low density of the species in this reach of the Murrumbidgee River; and/or (ii) that boat electro-fishing is inefficient for detecting this species based on even large amounts of sampling effort.

Substantial sampling effort is required to estimate CPUE of rare species with precision let alone to estimate population size, and represents a major obstacle in managing threatened fishes. The *B. bidyanus* population in the Murrumbidgee River provides an example (Gehrke *et al.* 1995; Gehrke & Harris 2000; Gilligan 2005b; this study). Conversely, the endangered *M. macquariensis* is at sufficient density in this reach of the Murrumbidgee River for CPUE to be measured with precision, therefore affording the opportunity to conduct adaptive management (Walters & Holling 1990) at the within-reach scale. In river systems containing numerous threatened fishes, one of the major functions of surveys is detecting remnant populations. A major challenge in the Murray–Darling Basin is then conducting focussed management on these remnants where success and failure can be measured at the population level. Furthermore, single-pass electro-fishing is likely to be a particularly important method for surveying these remnant populations, since the increased potential for injury and mortality as a function of multiple-pass electro-fishing becomes unacceptable from a conservation perspective (Kennard *et al.* 2006).

Limitations of this study

The current survey design had a number of limitations. Shallow water posed the greatest problem in this regard and certain large and complex woody habitat could only be accessed at the exposed edges. We also used only a single gear type and use of multiple gear types would likely have sampled particular species more effectively and may have increased the number of species detected (Faragher & Rodgers 1997; MDBC 2004; Lintermans *et al.* 2005; Kennard *et al.* 2006).

Fishes can also be absent from the river channel in time and space. Native species including Murray River rainbowfish *Melanotaenia fluviatilis* (Castelnau), *Gadopsis marmoratus,* bony herring *Nematolosa erebi* (Günther), and *Hypseleotris* spp. and the alien species redfin perch *Perca fluviatilis* Linnaeus and goldfish *Carassius auratus* Linnaeus are detected periodically in low abundances (Gilligan 2005b; Baumgartner 2007; Wooden unpublished data 2000) in the vicinity of this study reach. Baumgartner (2007) recorded 12 species from a section of river immediately downstream of our study area. However, in that study, *M. fluviatilis, Hypseleotris* spp. and *N. erebi* were only captured within the

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Yanco Weir pool (see location in Fig. 1) in lentic waters. These species are common in the lower reaches but are typically rare in the middle reaches of the Murrumbidgee River possibly as a function of human impacts, including barriers to migration and thermal pollution (Gilligan 2005b; Baumgartner 2007).

Recommendations

The current study presents an approach for effectively estimating fish diversity based on application of a single sampling method at a site. We recommend this approach be transferred to a number of sites to facilitate effective and efficient assessment of fish communities at the watershed scale and to develop robust reach specific monitoring programmes. Specifically we suggest: (i) conducting sampling of fish communities within a continuous length of lowland river at a number of sites; (ii) re-sampling of the data with different strategies; and (iii) application of species richness estimators (Hughes *et al*. 2002; Chao *et al*. 2005). In this way, an appreciation of the trade-offs in allocating sampling effort within and among sites can be obtained, and the costly methods of surveying fish species richness further refined (cf. Pennington & Volstad 1994). In the Australian context, it may be beneficial to focus on the issue of river-specific sampling-effort as identified in studies of fish communities in temperate North American rivers (Cao *et al*. 2001, 2002; Hughes *et al*. 2002; Smith & Jones 2005).

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