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Do protected areas benefit freshwater species? A broad-scale assessment for fish in Australia's Murray–Darling Basin

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Summary

1. Assessment of the performance of protected areas in conserving freshwater biodiversity has been limited, has mostly involved small-scale studies and has produced mixed findings.

2. I analysed monitoring data from Australia's Murray–Darling Basin ($>10^6$ km^2 and mostly arid or semi-arid) to compare fish assemblages between rivers inside and outside of protected areas.

3. The average richness and abundance of native species were significantly lower at sites within protected areas, but these sites were weighted towards steeper terrain and colder climates.

4. When analysis was confined to a subset of geographically and environmentally matched sites, the richness and abundance of native species did not differ significantly between protected and unprotected areas, and only two native species were significantly more abundant within protected areas, whereas another two were significantly more abundant outside.

5. *Synthesis and applications*. Protected status probably has little effect on native fish in the Murray–Darling Basin because it does not, by itself, exclude threats such as alien fish and alteration of water regimes. My findings therefore support the view that reserves need to be designed and managed specifically for freshwater species if they are to be most effective in their conservation. In the Murray–Darling Basin, and in similar regions around the world, actions such as greater control of alien species and allocation of adequate environmental flows will likely be critical to enabling protected areas to realize their potential for aquatic conservation.

Key-words: conservation, effectiveness, performance, propensity score, reserve, river

Introduction

Protected areas are a mainstay of biodiversity conservation throughout the world, but their performance has not been adequately assessed (Gaston *et al*. 2008). Uncertainty is particularly high in the case of freshwater biodiversity, with even the representation of freshwater features in protected areas being poorly known (Herbert *et al.* 2010). Doubts about the benefits of existing protected areas for freshwater conservation arise because few such areas have been designed or managed with freshwater biodiversity specifically in mind (Saunders, Meeuwig & Vincent 2002; Abell, Allan & Lehner 2007). For example, terrestrial protected areas often include only part of a river's catchment or use a river as a boundary rather than fully including it (Nel *et al.* 2007). Consequently, rivers within protected areas are often vulnerable to transmission of impacts from land and water use beyond reserve boundaries (Pringle 2001). In addition, fresh waters within protected areas may be stocked with or invaded by alien fish (Pittock, Hansen & Abell 2008).

Declines and losses of freshwater biodiversity within protected areas have been widely documented. For example, Barendregt, Wassen & Schot (1995) attributed a loss of wetland vegetation diversity in a Dutch reserve to extrinsic influences such as altered hydrology and nutrient enrichment. Disappearances of frogs ascribed to chytrid fungus infections and species losses associated with fish introductions have also been reported within protected areas (Schindler 2000; Skerratt *et al*. 2007).

Do protected areas nonetheless perform better than comparable unprotected areas in maintaining freshwater species? Only a few, mostly small-scale, studies have tackled

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*Correspondence author. E-mail: bruce.chessman@environment.nsw.gov.au this question, and they have produced mixed results. Positive findings include those of Baird & Flaherty (2005), who observed that villagers in Lao PDR reported increased fish abundance after protection zones were established, and Cucherousset *et al.* (2007), who found that eels were more abundant and larger in protected portions of a French wetland than in fished areas. Similarly, Sanyanga, Machena & Kautsky (1995) reported that the mean body size of commercial species was larger in protected than in fished areas of Lake Kariba, Zimbabwe. On the other hand, Mancini *et al.* (2005) observed that a macroinvertebrate index did not differ significantly between stream reaches inside and outside of Italian protected areas, and Srinoparatwatana & Hyndes (2011) found no consistent differences in abundance or biomass of wetland fish species between a protected area and an adjacent fished area in Thailand.

Comparisons of biodiversity within and beyond protected areas are a logical way to assess performance, but may be confounded if the places being compared differ in natural environmental features (Mas 2005). Such confounding is unfortunately likely because the creation of reserves often favours land with little potential for agricultural, urban or industrial use, such as steeply sloping terrain with low soil fertility (Scott *et al.* 2001; Pressey *et al.* 2002). Confounding by extraneous variables that covary with those of concern is a widespread problem when observational studies of the natural environment attempt to assign causes to observed differences or changes (Beyers 1998). In such studies, the role of potentially confounding factors cannot be balanced or minimized as it would in controlled experiments, where treatments are assigned randomly to experimental units and extraneous factors are held constant while those of interest are varied. However, propensity scores (Rosenbaum & Rubin 1983) provide a way to adjust comparisons for the influence of confounding factors in observational studies. A propensity score is the probability that an observational unit has received a specified treatment given the unit's values of potentially confounding factors. A comparison that is not biased by the confounding factors can be achieved by choosing observational units such that propensity scores are balanced among treatments.

I used propensity scores to evaluate how protected area status affects riverine fish assemblages over a large spatial extent: Australia's Murray–Darling Basin, which covers 1 061 469 km^2 or about one-seventh of the continent. Fish are an appropriate focus for a study of protected area performance because they are prominent in the precipitous decline of freshwater biodiversity that has been reported around the globe (Dudgeon *et al.* 2006; Strayer & Dudgeon 2010). Nearly half of the world's known fish species live in fresh waters (Lévêque *et al.* 2008), and 37% of those evaluated by the IUCN in 2008 were assessed as threatened (Vié, Hilton-Taylor & Stuart 2009). The Murray–Darling river system is also a salient case study, because many of its indigenous fish species have suffered severe historical declines (Cadwallader 1978) and several are formally listed as threatened (Koehn & Lintermans 2012). Current efforts to sustain the river system's biodiversity while accommodating social and economic needs for water illustrate challenges that other parts of the world are also likely to face (Pittock & Connell 2010).

Using data from a basin-wide monitoring programme, I first compared fish assemblages between all sampling sites located within protected areas (hereafter 'protected sites') and all sites outside of such areas ('unprotected sites'). I then used propensity scores to select a subset of protected and unprotected sites that were matched for natural environmental features and repeated the comparison. I hypothesized that once the effect of confounding factors was controlled, native fish would be more diverse and abundant in protected areas because such areas would provide at least some shelter from threatening processes associated with anthropogenic disturbance of rivers and their watersheds.

Materials and methods

STUDY AREA

The Murray–Darling Basin (Fig. 1) extends between 24 and 38°S, ranges in elevation from sea level to 2228 m and includes Australia's three longest rivers: the Darling (2740 km), Murray (2530 km) and Murrumbidgee (1690 km). Most of the basin is arid or semi-arid, so that despite its large size, it has an annual average discharge of only 4700 GL, which would be 12 200 GL if water diversions were absent (CSIRO 2008). The basin is Australia's premier agricultural region, accounting for about

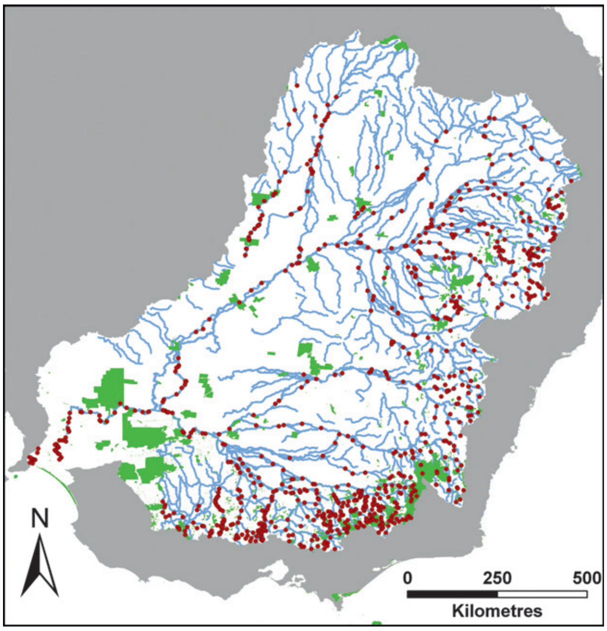

Fig. 1. Map of the Murray–Darling Basin showing protected areas (green), main rivers (blue) and fish sites sampled in the Sustainable Rivers Audit (red).

65% of the nation's total area of irrigated crops and pastures and about 39% of the national value of agricultural production (ABS 2008). Its fish fauna includes about 70 freshwater, estuarine and marine species, 12 of which are not native to Australia (Hardy *et al.* 2011).

DATA SOURCES

I analysed fish data collected between November 2004 and June 2010 as part of the Sustainable Rivers Audit, a monitoring programme for rivers throughout the basin (Davies *et al.* 2010). The data comprised geographical co-ordinates of each sampling site and the number of individuals of each fish taxon (mostly species level) collected at each site on each sampling occasion. Sampling sites were selected by stratifying the basin into geographical zones and choosing randomly from a defined stream network within each zone, subject to restrictions on the proximity of sites to one another. The data were from 839 sites (Fig. 1), of which 695 were sampled once and 144 twice. Sampling was spread over all calendar months except July, August, September and October, and fish were collected in box traps and with boat-mounted, bank-mounted or backpack electrofishing gear and recorded if longer than 15 mm.

Using a geographical information system (ArcGIS®), I assigned each site to a segment of a national stream layer (www.ga.gov.au/products/servlet/controller?event = GEOCAT_DETAILSandcatno = 73082; accessed January 2012) derived from a 9″ digital elevation model and attributed with environmental variables (Stein & Nevill 2011). I used satellite imagery in Google Maps® to assist in segment assignment where ambiguities arose from limitations in the accuracy of site co-ordinates and the stream layer (e.g. near tributary junctions). I assessed the protection status of each site by overlaying a GIS layer from the Collaborative Australian Protected Areas Database (www.environment.gov.au/parks.nrs/science/capad; accessed January 2012), which maps all reserves identified by Australian conservation agencies in mid-2008 as meeting the IUCN definition of a protected area (all management categories). I inspected each site and defined it as being either within or outside of a protected area, making allowance for minor discrepancies between site co-ordinates and actual river positions. In cases where a site lay on a river that separated protected and unprotected land, I treated the site as unprotected. I obtained climatic data for each site, averaged over the period 1950–2000, from the WorldClim global climate grids (www.worldclim.org; accessed December 2010), which interpolate between weather stations at a resolution of 30 arc seconds or approximately 1 km (Hijmans *et al.* 2005).

STATISTICAL ANALYSES

I first tested whether protected and unprotected sites differed in environmental variables that are likely to have a major bearing on fish assemblages (Jackson, Peres-Neto & Olden 2001). I chose site latitude, site longitude, the catchment area upstream of the site, the downstream distance from the site to the ocean (or to a terminal lake in the case of endorheic streams), the slope of the stream segment containing the site and the average air temperature at the site. Latitude and longitude were included to account for fish biogeography, catchment area to represent habitat variation from small headwater streams to large lowland rivers, distance to the mouth because of its likely importance to diadromous fish species, and slope because of its influence on stream morphology and hence fish habitat. Air temperature was included as a surrogate for water temperature, which was not available, and in recognition of the varying thermal requirements of fish species. Latitude and longitude were obtained from the fish data base, catchment area, distance to the mouth and slope from the national stream layer and temperature from WorldClim.

I used *t*-tests to compare protected and unprotected sites for each environmental variable, after transforming catchment area and slope to logarithms to better approximate normal distributions. A few segments had slope values of zero that could not be logarithmically transformed, so I substituted the lowest nonzero values. If *F*-tests revealed significant ($P < 0{\cdot}05$) differences in variances between protected and unprotected sites, I used separate-variance *t*-tests, otherwise I used pooled-variance tests.

I also compared fish assemblages between protected and unprotected sites, averaging data for sites that were sampled twice. I tested for differences in richness (number of taxa recorded per site per sampling occasion) of all native species combined and all alien species combined, differences in abundance (number of individuals per site per occasion) of all native species combined and all alien species combined, and differences in abundances of individual taxa. For richness, I used Poisson tests to compare protected and unprotected sites, and for combined abundance, I used *t*-tests as described above after transforming data to log $(x + 1)$. For abundances of individual taxa that contained many zero values, I used nonparametric Mann–Whitney *U*-tests.

Next, I selected subsets of protected and unprotected sites that were matched for the environmental variables. I first fitted a logistic regression model to estimate the probability that a site would be in a protected area given its environmental data, that is, its propensity score. The model took the form $\log(p_i/(1 - p_i)) = \beta_0 + \beta_1 x_{1,i} + \beta_2 x_{2,i} + \ldots + \beta_6 x_{6,i} + e$, where p_i was the propensity score of site i, β_0 was a constant, $\beta_1 \ldots \beta_6$ were regression coefficients, $x_{1,i} \ldots x_{6,i}$ were values of the six environmental variables for site i and e was an error term. After calculating each site's propensity score, I arranged the scores from highest to lowest and classified each site as either matched or unmatched according to where it fell in the sequence. I examined each pair of sites with adjacent scores and assessed the pair as matched if the sites were of different status (i.e. one protected and one unprotected) and unmatched if they were of the same status. Apart from the two sites at the extremes of the sequence, each site was involved in two comparisons – one with the site above it and one with the site below. Consequently, a site could be assessed as both matched and unmatched, in which case I treated it as matched. I then repeated all statistical comparisons of protected and unprotected sites for the matched subset.

Results

The sampling sites were most concentrated in the south and east of the basin where stream density is highest, and 69 (8·2%) were assessed as being within protected areas (Fig. 1). Protected sites were distributed across all IUCN protected area management categories apart from category VI (Table 1), but most fell in category II, that is, large natural or near-natural tracts reserved to protect characteristic species and ecosystems and broad-scale ecological processes, while allowing for compatible human

Table 1. Percentage of protected sites within each IUCN protected area management category

IUCN category	% of sites
Ia	9
Ib	3
II	70
III	14
IV	1
V	3
VI	0

uses (Dudley 2008). Almost half were within extensive, mountainous national parks: Kosciuszko National Park in New South Wales (6900 km^2) and the adjoining Alpine National Park in Victoria (6600 km^2) and Namadgi National Park in the Australian Capital Territory (1060 km^2). Nearly a quarter were in strip reserves along riverine corridors. On average, protected sites lay farther south, had smaller catchments and steeper slopes and were colder than unprotected sites (*t*-tests; $P \leq 0{\cdot}001$). However, the two site types did not differ significantly in mean longitude ($P = 0{\cdot}160$) or distance from the mouth ($P = 0{\cdot}113$; Fig. 2).

The fish data included 36 described species, two undescribed species of *Galaxias* and the genus *Hypseleotris*, comprising *H. klunzingeri* plus undescribed species (Table 2). On average, protected sites had significantly lower native fish richness (Poisson test; $P < 0{\cdot}001$) and abundance (*t*-test; $P = 0{\cdot}015$) than unprotected sites, but the two site types did not differ significantly in richness or abundance of alien fish ($P > 0{\cdot}05$; Fig. 3). Six fish taxa were significantly more abundant at protected sites, and another six were significantly more abundant at unprotected sites (Mann–Whitney *U*-tests; Table 2).

The logistic regression model was highly significant ($P \ll 0{\cdot}001$) and assigned mean propensity scores of 0·18 for sites that actually were within reserves and 0·07 for sites that were not. The matching process selected 65 of the protected sites and 100 of the unprotected ones. The environmental differences between protected and unprotected sites were much smaller for this matched subset than for all of the sites (Fig. 2), and none of them was statistically significant (*t*-tests; $P > 0{\cdot}05$). Native richness, native abundance and alien abundance did not differ significantly for the matched subset (Poisson and *t*-tests; $P > 0{\cdot}05$), but alien richness was significantly higher in protected sites ($P = 0{\cdot}033$; Fig. 3). Significant differences in abundance remained after matching for six fish taxa, four of which were more abundant within protected areas (two of them native), while two were more abundant outside (both native; Mann–Whitney *U*-tests; Table 2).

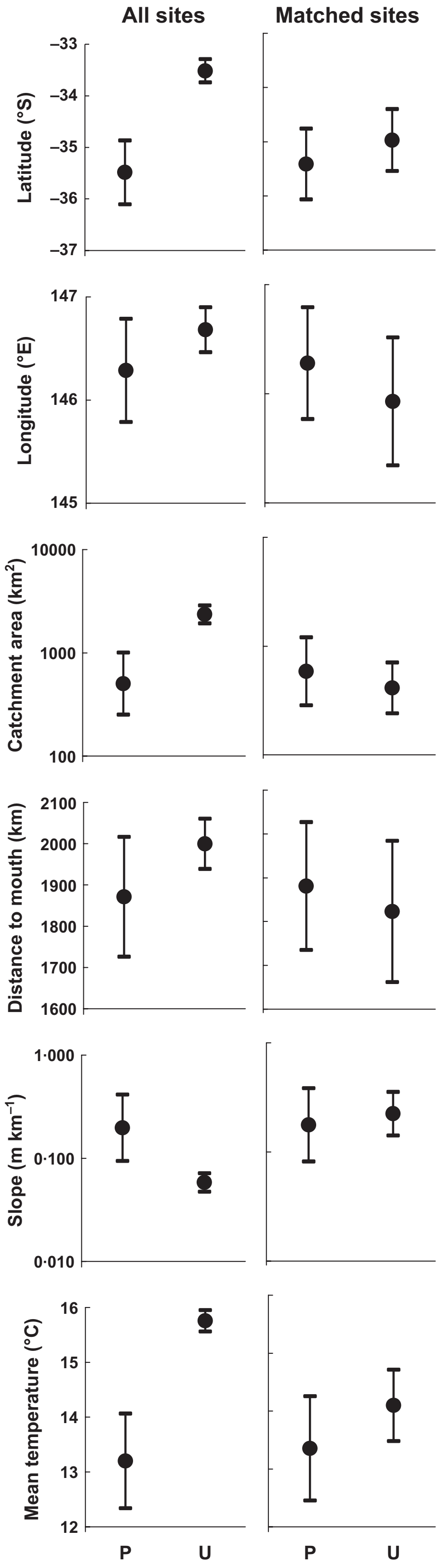

Fig. 2. Mean values of environmental variables (±95% confidence limits) for all protected (P) and unprotected (U) sites and for an environmentally matched subset of sites. Values for catchment area and slope were calculated from logarithmically transformed data.

Discussion

On average, fish sampling sites within protected areas were on smaller, colder and steeper streams than

Table 2. Mean number of individuals of each fish taxon collected at all protected and unprotected sites and at an environmentally matched subset of sites

	All sites		Matched sites	
Taxon	Protected	Unprotected	Protected	Unprotected
Ambassis agassizii (T)	0·043	0·006	0·046	0·005
Bidyanus bidyanus (T)	0·022	0·073	0·023	0·015
Carassius auratus (A)	1·109	2·134	1·177	1·070
Carassius carassius (A)	<0·001	0·002	<0·001	<0·001
Craterocephalus amniculus (T)	<0·001	0·103	<0·001	0·160
Craterocephalus fluviatilis (T)	<0·001	0·001	<0·001	0·010
Craterocephalus stercusmuscarum fulvus (T)	1·000	3·471	1·062	2·955
Cyprinus carpio (A)	12·261***	9·173***	13·015	4·535
Gadopsis bispinosus (T)	5·152***	1·458***	5·469**	1·980**
Gadopsis marmoratus (T)	0·261*	1·129*	0·246*	0·885*
Galaxias brevipinnis	0·058	0·017	0·062	0·090
Galaxias fuscus (T)	<0·001	0·103	<0·001	0·790
Galaxias maculatus	0·681*	0·078*	0·246	0·310
Galaxias olidus (T)	3·341	6·019	2·992	8·865
Galaxias sp. 1	<0·001*	1·793*	<0·001**	2·295**
Galaxias sp. 2	0·188	0·231	0·200	0·115
Galaxias truttaceus (T)	<0·001	0·003	<0·001	<0·001
Gambusia holbrooki (A)	3·145***	35·348***	3·338	18·225
Hypseleotris spp.	7·312**	20·268**	7·762	16·710
Leiopotherapon unicolor	10·862	1·370	11·531	0·845
Maccullochella macquariensis (T)	0·116	0·052	0·123	0·090
Maccullochella peelii (T)	0·630	0·938	0·669	0·485
Macquaria ambigua (T)	1·442	1·386	1·531	0·565
Macquaria australasica (T)	0·014	0·012	0·015	0·010
Melanotaenia fluviatilis (T)	0·884	1·821	0·938	1·355
Misgurnus anguillicaudatus (A)	0·014	0·073	0·015	<0·001
Mogurnda adspersa (T)	<0·001	0·008	<0·001	<0·001
Nannoperca australis (T)	0·043	1·856	0·046	4·130
Nematalosa erebi	11·543**	26·612**	12·254	8·130
Neosilurus hyrtlii	0·159***	0·027***	0·169*	0·060*
Onchorhynchus mykiss (A)	8·174***	0·949***	8·231***	3·450***
Perca fluviatilis (A)	0·957	4·531	1·015	1·750
Philypnodon grandiceps	2·703	1·473	2·869	2·285
Philypnodon macrostomus	0·014	0·009	0·015	<0·001
Retropinna semoni	2·399*	5·772*	2·546	3·210
Rutilus rutilus (A)	<0·001	0·038	<0·001	0·010
Salmo trutta (A)	7·732***	1·916***	6·885*	4·070*
Tandanus tandanus (T)	0·080	0·213	0·085	0·010
Tinca tinca (A)	<0·001	0·147	<0·001	0·785

Species listed as threatened at the state, territory or national level (Koehn & Lintermans 2012) are designated as (T) and those alien to Australia as (A).
Asterisks denote significant differences between site types: *0·01 < P < 0·05; **0·001 < P < 0·01; ***P < 0·001.

unprotected sites. A bias in allocation of protected status towards higher elevations has been observed in many parts of the world (Keith 2000; Scott *et al.* 2001; Januchowski-Hartley *et al.* 2011) and may reflect the scenic value of mountainous terrain and its lower suitability for competing land uses such as agriculture. The Murray–Darling Basin is straddled by extensive alpine and subalpine reserves but also has some large protected areas at low elevations. However, the latter lie mainly in arid and semi-arid tracts where non-ephemeral streams are scarce. Consequently, few of the lowland sites but many of the upland ones fell within protected areas.

The existence of statistically significant natural environmental differences between sites inside and outside of reserves precluded inferring benefits of protected status from a universal comparison of protected and unprotected sites. However, the use of propensity scores enabled a comparison that was geographically and environmentally balanced and suggested that protected status had little overall effect on fish assemblages. Two native species (plus two alien species) were significantly more abundant in protected sites than in matched unprotected sites, but two native species were significantly less abundant, and given that 39 taxa were considered, two significant differences are expected by chance at $\alpha = 0{\cdot}05$.

Beneficial effects of the reserve system were perhaps under-estimated by this analysis because movements of fish out of protected areas could have spread benefits

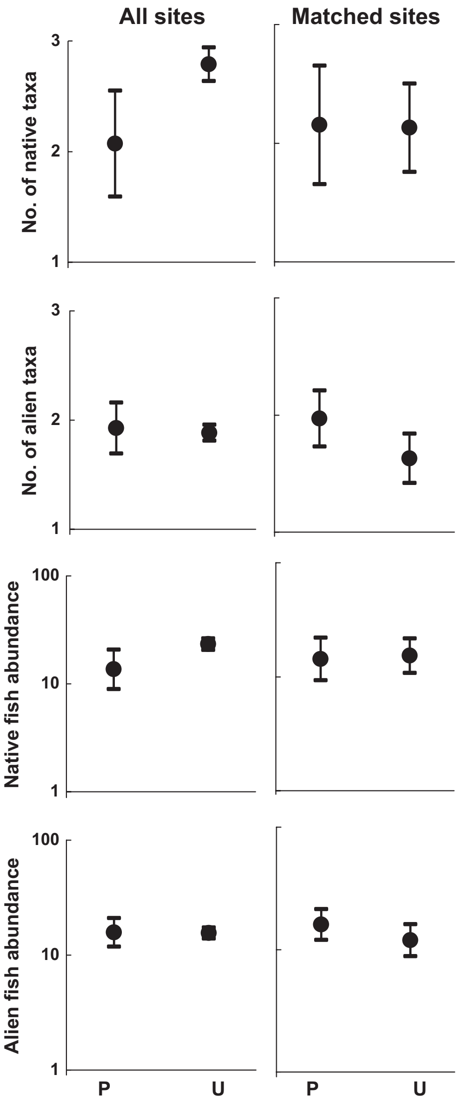

Fig. 3. Mean values of fish assemblage attributes (±95% confidence limits) for all protected (P) and unprotected (U) sites and for an environmentally matched subset of sites. Values for abundance of native and alien species were calculated from logarithmically transformed data.

from protected to unprotected areas. However, any fish that moved randomly among the sampling sites in this study would have spent only about 8% of its time in reserves, and consequently, any benefit received from reserves by highly mobile species is likely to be limited. In addition, protected areas might have provided benefits to fish living in downstream, unprotected areas, for example by influencing water quality. However, if this effect were large, it should have been manifest in assemblage differences between protected and unprotected sites, because on average, the proportion of a site's catchment that was within protected areas was about twice as great for protected as for unprotected sites.

Protected area status probably has little effect on fish assemblages in the Murray–Darling Basin because it does not, by itself, exclude some major anthropogenic influences. For example, reductions in river flows and the inundation of floodplain wetlands, caused by impoundment and water abstraction, have been implicated as a principal cause of native fish decline in the Murray–Darling Basin (Cadwallader 1978). The flow (and temperature) regimes of rivers in the basin's protected areas can be altered by impoundment and abstraction in upstream, unprotected river reaches or even by within-reserve infrastructure such as hydro-electric developments (Kingsford 2000). Water chemistry within protected areas can be affected by upstream land use or within-reserve tourism developments (Pickering, Harrington & Worboys 2003). Protected status also does not prevent fish harvesting, because angling is generally permitted within reserves, or exclude alien fishes, which impose a great constraint on native fish conservation. Introduced salmonids, especially *Salmo trutta* and *Oncorhynchus mykiss*, have severely reduced the abundance and geographical ranges of native *Galaxias* species in south-eastern Australia (Crowl, Townsend & McIntosh 1992; McDowall 2006), limiting the galaxiids of some protected areas to fragmented populations with a high risk of genetic diversity loss through local extinction (Green 2008). Other alien fishes in the basin with well-documented adverse impacts include *Cyprinus carpio* (Koehn 2004) and *Gambusia holbrooki* (Pyke 2008). The results presented here confirm that alien fishes are plentiful in protected areas and even suggest that some alien species are more abundant within such areas than outside of them.

Studies reporting benefits of protected areas for freshwater fish species have typically involved specific provisions for fish conservation such as 'no-take' zones (Sanyanga, Machena & Kautsky 1995; Baird & Flaherty 2005; Cucherousset *et al.* 2007). In the case of the Murray–Darling Basin, environmental water allocations are already being used to enhance spawning and recruitment of native fish in some protected areas (Tonkin, King & Mahoney 2008; Rayner, Jenkins & Kingsford 2009; King *et al.* 2010) and might become a widespread and routine management tool in protected areas where river flows and wetland inundation are affected by water resource development or climate change. Opportunities also exist to reduce alien fish populations within the basin's protected areas. Broad-scale eradication of trout would not be socially acceptable because of their recreational popularity and economic value (Jackson *et al.* 2004), but local removal is technically feasible (Lintermans 2000) and can be done selectively to protect threatened native species (Raadik, Saddlier & Koehn 1996). Some other alien fishes such as *C. carpio* and *G. holbrooki* are more difficult than trout to control or eradicate, but have little social or economic value and therefore might be widely targeted within protected areas.

Many authors have advocated the creation of new types of protected areas dedicated specifically to freshwater conservation (Moyle & Yoshiyama 1994; Saunders, Meeuwig & Vincent 2002; Kingsford & Nevill 2005;

Abell, Allan & Lehner 2007; Suski & Cooke 2007; Humphries & Winemiller 2009; Williams *et al.* 2011). In the Murray–Darling Basin, several intensively managed 'demonstration reaches' have been established as part of a Native Fish Strategy that aims to rehabilitate indigenous fish populations (Koehn & Lintermans 2012). These reaches do not have formal reserve status but might serve as a precursor to freshwater protected areas by fostering community awareness and support (Barrett & Ansell 2003). Other potential candidates for freshwater protected areas in the Murray–Darling Basin include habitats of threatened fish species that are poorly represented in existing reserves and drought refuges, which are likely to become increasingly important in the light of projected long-term climatic drying (Leblanc *et al.* 2012).

CONCLUSION

Much remains to be done to develop an adequate system of freshwater protected areas for Australia (Fitzsimons & Robertson 2005; Stein & Nevill 2011) and the world (Abell, Allan & Lehner 2007). The actions necessary to achieve this aim include the creation of new reserves to fill gaps in coverage of the full spectrum of freshwater species and ecosystems and to embrace critical habitats such as refuges. However, the present study provides broad-scale evidence to reinforce concerns that protected status *per se* will not adequately conserve freshwater biodiversity, especially where the principal threats are not habitat destruction or direct harvesting but more insidious and pervasive factors such as alien species and climate change (Abell, Allan & Lehner 2007; Pittock, Hansen & Abell 2008; Rahel, Bierwagen & Taniguchi 2008; Williams *et al.* 2011). Area protection will achieve its potential for freshwater conservation only if coupled with intensive management to abate threats. For the Murray–Darling Basin, technical advances in control of alien fishes (Britton, Gozlan & Copp 2010) and resolution of competing demands for water (Connell & Grafton 2011) are likely to be critical to enabling protected areas to maximize their contribution to freshwater conservation. The same is likely to hold true in other parts of the world where invasive fishes are rife or dry climates intensify pressures on water resources.

Acknowledgements

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