Identifying conservation priorities for aquatic refugia in an arid zone, ephemeral catchment: a hydrological approach

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ABSTRACT

The conservation priorities for protecting aquatic habitat of arid zone rivers requires an understanding of the drivers of distribution, characteristics and persistence of aquatic refugia in ephemeral to intermittent arid zone rivers. We describe and analyse the aquatic refugia that persist during drought periods in a large, arid, ephemeral river, the Neales River of central Australia, concentrating on refugia for fish and the hydrological processes that influence their location, characteristics and persistence. Field data were collected on flow events, water-level fluctuations in waterbodies during periods of no flow, waterbody dimensions and salinity. Modelling of open-water evaporation rates, analysis of salinity changes and frequency of flow events were used to investigate the main processes determining the persistence of waterbodies. The catchment contained both fluvial waterholes and artesian springs providing aquatic refugia. In addition, anthropogenic refugia, in the form of small dams situated on the floodplain and uncontrolled flow from artesian bores, also provided potential refuge. The catchment only contained one waterhole that had a cease-to-flow depth sufficient to persist over a no-flow period of 24 months, but this would also dry within 24–30 months. The capacity of the artesian springs to act as refugia is limited by their shallow depths and limited connectivity with the main river system. These data illustrate the vulnerability of the fluvial ecology of ephemeral, arid zone rivers to long-term droughts and water resource use and assist in identifying specific refugia that require conservation management.

KEY WORDS

Neales River; Lake Eyre Basin; salinity; refugia; springs; evaporation modelling

INTRODUCTION

Conservation priorities in arid zone rivers

The conservation priorities of arid zone rivers can be masked by a paucity of data on the hydrological and geomorphological framework supporting habitat requirements of aquatic fauna. Refugial waterbodies are critical for sustaining populations of many aquatic fauna during periods of no flow (Arthington et al., 2005; Sheldon et al., 2010), including endemic species (Fensham et al., 2011), and their distribution can have a large influence on the recolonization movement of aquatic fauna during flow events (Magoulick and Kobza, 2003; Sheldon et al., 2010). In arid zone catchments, the existence of long-term aquatic refuges influences the persistence of much of the fish assemblage at a catchment or even regional scale (Fensham et al., 2011). However, knowledge of the drivers of distribution,

characteristics and persistence of aquatic refugia at the catchment scale is commonly unavailable (Magoulick and Kobza, 2003; Bunn et al., 2006; Larned et al., 2010).

Conservation priorities for wetlands typically focus on high biodiversity (Mittermeier et al., 1998; Junk et al., 2006) to identify regionally important sites and with a particular focus on endemism for arid zone wetlands (Fensham et al., 2011). However, arid zone rivers are commonly typified by only moderate diversity in their aquatic fauna and favour generalist species with high dispersal capacity (Hoagstrom and Berry, 2006; Maltchik and Medeiros, 2006; Balcombe et al., 2007). As a result, an approach concentrating on biodiversity and endemism can ignore conservation requirements of large catchments (i.e. >10 000 km²) with moderate and non-endemic biodiversity but which are vulnerable to anthropogenic pressures on key refugia. The vulnerability of such catchments is heightened if recolonization pathways from neighbouring catchments for aquatic fauna are poorly understood or unlikely under current climatic conditions (Faulks et al., 2010; Fensham et al., 2011). Long drought periods with no streamflow can result in the available aquatic habitats contracting to a small handful (e.g. Bunn et al., 2006), or even a single waterbody. Even though the aquatic fauna have evolved under these variable conditions, during drought periods, these aquatic refugia then become highly vulnerable to anthropogenic disturbances; ranging from broad-scale changes in the natural flow regime through regulation (Bunn and Arthington, 2002), to more local scale effects that influence individual refuges, such as water extraction, local diversion of flow altering the frequency of inundation, pollution and over-fishing (Sheldon et al., 2010). Therefore, the identification and protection of these refugia is an important conservation priority at the catchment scale and potentially at a larger regional scale.

Types of aquatic refugia and persistence characteristics

The characteristics that define aquatic refugia are taxon-dependent and so display considerable spatial and temporal variability (Lancaster and Belyea, 1997). In the context of the requirements of fish and fully aquatic invertebrates in intermittent to ephemeral rivers, this simplifies to the provision of water in the form of persistent waterbodies (Magoulick and Kobza, 2003; Sheldon et al., 2010). For the fish assemblage, the refuge requirements will differ between species, potentially as a result of predator–prey interactions within isolated waterbodies. As a result, the identification of aquatic refugia needs to consider all waterbodies within a catchment, such as fluvial pools (e.g. Arthington et al., 2005; Bunn et al., 2006; Carini et al., 2006), natural springs (Perez et al., 2005; Kodric-Brown and Brown, 2009) and also any anthropogenic refugia (e.g. small floodplain dams and flowing bore drains, Lutton et al., 2010). In addition, the lateral and longitudinal connectivity of flow patterns (Larned et al., 2010) and temporal water quality variations (Hudson et al., 2003; Costelloe et al., 2005b) can influence the utility of a persistent waterbody as an aquatic refuge.

We contend that an evaluation of the conservation value of aquatic refugia in an ephemeral arid zone catchment requires the following steps:

1. Identify aquatic habitats that persist for periods of > 9–12 months (i.e. between annual wet season streamflow events).
2. Determine the criteria that explain the persistence of the aquatic refugia, e.g. groundwater connectivity, large maximum depths, more frequent inflow.

3. Evaluate the importance of any aquatic refugia in terms of its capacity to link with other refugia during periods of streamflow.

4. Identify temporal and/or spatial water quality variations that may impede the capacity of the aquatic habitat to act as refugia for aquatic fauna.

We illustrate these steps with field data and analysis of the distribution and persistence of aquatic refugia in a large, ephemeral, arid zone river in central Australia.

STUDY AREA

The Neales River is an unregulated ephemeral river system in the arid Lake Eyre Basin of central Australia. The catchment has an area of 34 000 km² before flowing into the endoreic Lake Eyre North (Figure 1), with a median annual rainfall of 140 mm and a mean annual pan evaporation loss of 3577 mm (Australian Bureau of Meteorology). The majority of the rainfall occurs during the summer season (December to March), but significant falls can occur during winter.

The Neales River is ungauged, but streamflow events have been monitored by a number of water-level loggers since 2000 (Costelloe et al., 2005a, 2005b; Costelloe et al., 2007). The catchment contains mapped fluvial waterholes and artesian springs sourced from groundwater discharging from the Great Artesian Basin (GAB). The age of GAB springs could be in the order of >100 000 years (Prescott and Habermehl, 2008), and these refugia are known to provide habitat for endemic fish (Wager and Unmack, 2000; Fensham et al. 2011) and invertebrate species (Perez et al., 2005; Murphy et al., 2009). No endemic fish are reported for the springs in the Neales catchment but endemic invertebrates are reported (Murphy et al., 2009).

Eleven fish species have been recorded in the fluvial waterholes and springs of the Neales catchment (Table I, data from Costelloe et al., 2004; McNeil and Schmarr, 2009). These range from large predator species, such as *Macquaria ambigua* (golden perch), to small saline-tolerant species, such as *Craterocephalus eyresii* (Lake Eyre hardyheads). All species are native to the Lake Eyre Basin except for the exotic species *Gambusia holbrooki* (mosquito fish). The previous studies report generally healthy, abundant and diverse fish populations during wet periods (Costelloe et al., 2004) but much lower diversity following drought periods (McNeil and Schmarr, 2009). The fish species that typically occur in the artesian spring habitats are the small-bodied *Chlamydogobius eremius* (desert gobies) and *G. holbrooki*, although medium-sized fish, such as *Leiopotherapon unicolor* (spangled perch) and *Melanotaenia splendatatei* (desert rainbowfish), can colonize these habitats but only persist for limited periods of time, except in rare deeper pools (Fensham et al., 2011).

METHODS

The evaluation of the conservation value of aquatic refugia requires both field data and analysis of available datasets.

Location of aquatic refugia and characterisation

The locations of the aquatic refugia within the catchment were identified from 1:250 000 scale topographical maps, and additional investigation used a combination of local knowledge, field inspections and satellite images. The GAB artesian springs located in close proximity of the Neales River (i.e. within 1 km of the floodplain) were visited in the field to evaluate their connectivity with river flow.

In addition, a selection of anthropogenic aquatic habitats (e.g. small floodplain dams and free-flowing artesian bores) were also visited and evaluated.

The dimensions of fluvial waterholes were measured using simple 'wet survey' techniques and surveying using a total station. The wet survey data were collected over a series of field trips between 2000 and 2010 and involved measuring between 1 and 13 (mean 6) transects across the waterhole, depending on its length, with a surveying tape to measure length and a weighted tape to measure depth along the transect. Surveys by total station were used to characterize the out-of-water morphology for the remainder of the waterhole survey (i.e. from water level to top of bank and onto the surrounding floodplain). The surveys identified the maximum depth of the waterhole when it first became isolated from other pools in the channel system when flow ceased, termed the 'cease-to-flow depth' (CTFD), and this measure is significantly less than the more common geomorphological descriptor of channels, the bankfull depth. Waterholes surveyed within 2–4 weeks of a flow event were considered to be at the CTFD. Field observations found that the CTFD typically coincided with riparian vegetation zonation (i.e. the base of the lignum (*Muehlenbeckia florulenta*) zone, Costelloe et al., 2004). For waterholes surveyed some months after flow cessation, this vegetation zonation and downstream river bed sill levels were used to estimate the CTFD. The error for the CTFD was considered as ± 0.2 m, incorporating uncertainty in depth measurements, repeat positioning of transects and measuring vegetation zonation.

The occurrence of flow events was evaluated using data from water-level loggers that recorded water-level variations each hour at six locations since 2000 (Dataflow conductance loggers, Odyssey Data Recording Systems) and at a further eight locations (Odyssey temperature-pressure loggers, Odyssey Data Recording Systems) since 2009 (Figure 1). At four of the sites, the loggers were installed in waterholes, and the logger data provided information on waterhole loss rates during no-flow periods.

Conductivity and temperature data were collected from all aquatic refugia at 0.1–0.2 m below the surface of the waterbody using water quality instruments (WP-84, TPS Pty Ltd; U-50, Horiba; YSI 6920).

Characterisation of aquatic refugia persistence, connectivity and water quality

The probability of groundwater discharge contributing to the persistence of fluvial aquatic refugia was investigated using (1) electrical conductivity (EC) data, (2) comparison of observed water-level loss rates (from logger data) to modelled evaporative loss rates and (3) frequency of flow determined from water-level time-series data.

An EC threshold of $>3000 \mu\text{S cm}^{-1}$ in fluvial waterholes (Table III) is used to identify waterholes receiving some saline groundwater discharge. This is based on the assumptions that the conductivity of peak magnitude streamflow was $<300 \mu\text{S cm}^{-1}$ (see minimum EC values measured for waterholes in Table III) and of conservative evapo-concentration behaviour (reasonable for Na-Cl dominated surface water systems of Australia) in a waterhole acting like a sealed basin. The evapo-concentration during isolated drawdown would result in an order of magnitude increase in EC ($3000 \mu\text{S cm}^{-1}$) at 10% of its initial starting volume. The temporal salinity behaviour was evaluated to determine if this could be a limiting factor in the value of that waterbody to act as an aquatic refuge for a wide range of biota.

Water-level time-series data have also been used to determine persistence using two methods. First, the water-level data were used to determine if particular reaches had higher frequencies of flow events that could enhance the refugial value of a waterhole. Second, the waterhole drawdown rates in the

absence of flow were compared with expected open-water evaporation rates. The open-water evaporation rate was calculated using the Penman combination equation and climate data from the closest weather station in the catchment (methods described by Costelloe et al., 2007). For this study, an approximate error bound for the modelled evaporation was estimated by plotting a data envelope of monthly time-series of modelled evaporation and adjusted pan evaporation for the closest three climate stations to the field area; Oodnadatta (Station 017043, 27.56° S, 135.45° E), Moomba (Station 017096, 28.11° S, 140.21° E) and Birdsville (Station 038002, 25.90° S, 139.35° E). The modelled time-series used both wind speed data and wind run data from each climate station, as uncertainty in the wind data was found to be the largest contributor to the variation in modelled open-water evaporation. In addition, the data envelope included adjusted pan evaporation data from Moomba and Birdsville (using pan factors derived by linear regression with Penman modelled evaporation). Approximate error bounds on the modelled evaporation data at Oodnadatta were determined by comparing them to the data envelope formed by all modelled open-water evaporation time-series.

Surveying using differential global positional systems (Leica VIVA GNSS in rover and base mode) and total stations (Sokkia 30R Series) was used to determine water-level thresholds that resulted in connectivity between fluvial waterholes and other refugia (e.g. springs, flowing bores drains and floodplain dams).

RESULTS

Distribution and characteristics of aquatic refugia

A total of 20 significant fluvial waterholes were identified and surveyed, and their distribution is shown in Figure 1, along with the location of all known GAB springs, the inspected small floodplain dams and flowing artesian bores. The hydrological characteristics of these refugia are shown in Table II and salinity characteristics in Table III. The CTFD of the fluvial waterholes ranged from 1.2 to 4.7 m. The deepest waterhole, Algebuckina Waterhole, also had the largest width and length and was typically deeper by 0.7–0.9 m than the next deepest waterbodies (Figure 2). However, measurements of the CTFD of Algebuckina during 2000–2010 showed that aggradation had occurred, and this measure decreased from 4.5 to 3.5 m between 2003 and 2009 and then recovered to 4.7 m following a flood in February 2011 with an approximate recurrence period of one in 10 years. Multiple measurements of CTFD for other waterholes were limited but, where available, did not show any evidence of significant aggradation within the error limits of the surveying techniques. The distribution of the deeper fluvial waterholes is restricted to the middle to lower reaches of the catchment.

Small dams (or water storages) with areas of 2500–5000 m² are occasionally situated on the outer floodplain but do not impede or capture flows along primary channels. Two of these were surveyed (Slate Hole, Eaglehawk Dam, Figure 1) and have depths comparable to many of the natural waterholes (Table II). Their position on outer floodplain channels may result in lower frequencies of inflow and shorter periods of connectivity compared with the natural waterholes, but no field data are available to test this.

The artesian springs occur on both the floodplain and in association with smaller tributaries (Figure 1). The springs range from minor seeps (<10 m² in area) to free-flowing springs that create shallow pools (typically <0.2 m in Figure 2. Cease-to-flow depths for waterholes in the Neales River catchment. The range in values measured for Algebuckina is shown by the error bars. Codes for waterholes are shown in

Table I. Horizontal bars show modelled open-water evaporation losses with error range for 1 year (black solid and dashed lines) and 2 years (grey solid and dashed lines) without inflow.

depth) and outflow tails that create extensive wetlands. The immediate pool area is usually <1600 m², but the flowing tails can create wetlands up to 1000 m long, and both the pool and tail are typically defined by the occurrence of reeds (*Phragmites* and *Typha*) and patches of shallow open water. The EC of the spring pools vary with flow rate but are most commonly in the range of 3000–9000 $\mu\text{S cm}^{-1}$. The largest spring-fed, open-water body occurs at North Freeling pool (<0.3-m maximum depth, 10 000 m² in area) on the outer edge of the floodplain.

Analogous to GAB springs, drains from free-flowing artesian bores can maintain persistent wetlands that can potentially connect with the river system. Most GAB bores are capped, but historically, they were uncapped and free-flowing. Two wetlands currently fed by uncapped bores were identified in the catchment – One Mile and Big Blythe (capped in 2012) bores (Figure 1). Both form large wetland areas (up to 420 000 m²) with considerable macrophyte vegetation (*Phragmites* and *Typha*), but some open pools of limited depth (usually <0.2 m). These wetlands have very similar hydrological characteristics as the springs, i.e. stable but shallow water depths, with wetland areas similar to those of the larger natural springs and occur close to the outer edge of the floodplain (Big Blythe) or on a tributary (One Mile).

Causes of refugia persistence

According to the maximum conductivity criterion, eight of the 20 waterholes received likely saline groundwater discharge (Table III), and all but one (Stewart) occurred in the lower reaches. The effects of unconfined groundwater discharge on the persistence of the saline pools are uncertain, but they were observed reaching very shallow depths (e.g. Peake Crossing <0.2-m depth) indicating that groundwater discharge does not lead to greatly increased waterbody persistence.

The modelled mean annual open-water evaporation rate was 2372 mm with an estimated uncertainty of +14% to -34%. Where available, time-series of water-level data were used to test the assumption of no groundwater connectivity, and this is illustrated with drawdown rate data for the period 2000–2008 from two instrumented waterholes in the Neales catchment (Algebuckina and South Stewart). This analysis (Figure 3) shows that neither waterhole received observable groundwater inflow despite the high conductivity observed in Algebuckina Waterhole (Table III). For instance, Algebuckina Waterhole largely loses water within the uncertainty bounds of the modelled evaporation rate but that some recharge to groundwater (i.e. leakage) could be occurring. South Stewart experiences consistent drawdown rates higher than the uncertainty bounds of the modelled evaporation rates (1.4 m per annum greater than the modelled mean annual evaporation rate), indicating that it is a 'leaky' waterhole and experiences significant loss to the unconfined groundwater. The South Stewart waterhole is within a few hundred metres of Cramps Camp waterhole, the second deepest waterhole in the catchment (Table I), and leakiness within this reach may reduce the refugial value of this cluster of waterholes in the upper reaches of the Neales.

No-flow conditions within a reach were observed to extend for periods greater than 1 year. For instance, Algebuckina Waterhole experienced four periods of no flow between 311 and 427 days in length during the monitoring period. The longest measured period of no flow at any of the monitoring sites in the catchment (without data gaps) was 591 days at Arckaringa Creek. In conjunction with the modelled

evaporation rate data, these no-flow periods indicate that waterholes with CTFD less than the annual evaporation loss (i.e. 2.4 m) are unlikely to play significant roles as long-term refugia.

The logger data (Table IV) indicate that the frequency of flow was similar in the upper and lower reaches of the Neales. Variations in frequency of flow events in the upper and lower reaches of the Peake were more difficult to identify because of data gaps. The lower Peake probably has a similar frequency of flow as the Neales; however, the lower frequency of flows in the upper Peake appears significant but also coincides with a lack of any deep waterholes (i.e. $CTFD > 2.4$ m). Therefore, the persistence of the refugia in this catchment is not significantly enhanced by catchment variations in the frequency of flow events.

Flow and connectivity patterns

The waterbodies with the greatest long-term persistence in the catchment are shallow GAB spring pools, and their conservation importance for aquatic fauna (particularly fish) at the catchment scale depends partly on their connectivity with the river. The connection between springs and the river system can be direct at high river stages for springs occurring on the floodplain of the river (i.e. North Freeling (NO), Figure 1) or indirect for springs that experience periodic connection from tributary flow (e.g. Ockenden Spring (OS), Figure 1) that can connect with river flow, as illustrated in Figure 4. These connectivity patterns indicate that at least some springs could act as refugia for small-bodied fish and be capable of repopulating the river following drought periods.

Salinity variations in refugia

The fluvial refugia of the Neales River showed an extraordinarily large range in conductivities, both spatially and temporally (Table III). Minimum observed conductivities were typically fresh ($< 300 \mu S cm^{-1}$) and were considered to be representative of larger flow events. In the upper reaches, the waterholes remain relatively fresh even at low water levels; however, in the middle to lower reaches, the conductivity of the waterholes ranges from fresh to hypersaline ($> 50\,000 \mu S cm^{-1}$). During periods of no flow, shallow residual pools in this reach were observed with conductivities $> 200\,000 \mu S cm^{-1}$. These lower reaches contain a number of important waterholes, including the deepest waterhole in the catchment, Algebuckina Waterhole. The latter is generally fresh, but after larger flows, its conductivity can be $> 20\,000 \mu S cm^{-1}$ (Table III).

The conductivity of the saline to hypersaline waterholes can vary temporally over several orders of magnitude, and this is illustrated for the Peake Crossing waterhole (Figure 5). Flow events are quite fresh, and the waterhole typically has moderate conductivity ($4000\text{--}9000 \mu S cm^{-1}$) soon after most flow events. However, $EC > 150\,000 \mu S cm^{-1}$ has been observed forming within 195–285 days from the peak of the most recent flow event.

Persistence patterns of fluvial refugia

The modelled evaporation rate and CTFD were used to estimate the persistence time of the surface-fed refugial waterholes (excluding the saline waterholes of Peake, Tardetakarinna, Baltacoodna) and identified the sequence of waterholes drying over the catchment (Figure 6). Allowing for the higher loss rates from the ‘leaky’ waterholes, the evaporation modelling clearly demonstrates that Algebuckina has the longest persistence of the fluvial waterholes of approximately 23–24 months without surface inflow. The uncertainty in this modelling is in the order of 1–2 months. Water loss rates in the saline waterholes

are likely to be less because of some contribution from groundwater discharge, but the refugial value of these waterholes is limited by hypersaline conditions forming within 10 months of a flow event.

DISCUSSION AND CONCLUSIONS

Conservation priorities at the catchment scale

The Neales catchment contains a wide array of aquatic refugia that include waterholes, small floodplain dams, artesian springs and wetlands fed by free-flowing artesian bores. This mix of natural and anthropogenic, surface-fed and groundwater-fed aquatic refugia are not commonly considered when determining critical refugia of an arid zone catchment, where the previous attention has often been on fluvial refugia (e.g. Arthington et al. 2005; Hamilton et al. 2005; Bunn et al. 2006; Sheldon et al. 2010). The mixture of fluvial and groundwater-fed refugia is not that unusual for an arid zone catchment (Box et al. 2008; Keleher and Rader 2008; Fensham et al. 2011) and has important conservation implications, as the different waterbody types provide refuge to different components of the fish assemblage. The most persistent aquatic refugia in the catchment are the artesian springs (Prescott and Habermehl, 2008), but their suitability for the complete range of the fish assemblage is likely to be limited by their limited depth and habitat type (Fensham et al. 2011), being most suitable for aquatic fauna adapted to very shallow water depths (i.e. <0.2 m) and dense macrophytes. Notwithstanding these limitations, the artesian springs showing connectivity with the river system and having larger open pools (e.g. North Freeling) clearly have high conservation value for components of the fish assemblage.

In contrast to the springs, the fluvial waterholes have a maximum persistence of approximately 24 months without inflow and likely have a much lesser long-term persistence but provide a greater range of habitats for a wider variety of fish (Arthington et al. 2005). For waterholes that normally experience at least one flow event per year, a useful definition of a fluvial refuge in an ephemeral river is one that can persist in the event of there being no flow for an entire 'wet' season. Such drought (no-flow) periods would be in the order of 18–24 months, depending on the timing of flow cessation in the previous wet season and the onset of flow in the proceeding wet season. Given the expected evaporation rates in the Neales catchment, refuge waterholes would require a maximum CTFD of >4m to sustain aquatic biota during such drought periods. As shown in Figure 2, in the Neales catchment, only Algebuckina Waterhole has sufficient depth (albeit subject to aggradation) to fulfil this criterion for an aquatic refuge. It clearly has the largest CTFD (despite some temporal variation) and does not show any evidence of receiving groundwater discharge (Figure 3, see also Costelloe et al. 2005b). The conservation importance of other waterholes was lessened by leakiness (e.g. South Stewart) or high salinity (e.g. Baltucoodna) limiting their effective persistence. In addition to its persistence, the size (Table I) and connection potential of Algebuckina Waterhole enhance its conservation importance. Unlike the small dams located on the edges of the floodplain that would only be connected to the river during larger flow events, Algebuckina is located on the primary channel of the Neales River. The significant variation in the CTFD of Algebuckina Waterhole emphasizes the importance of larger floods (i.e. with approximate recurrence intervals of one in 10 years or greater) in re-setting the geomorphic conditions that provide the large waterholes with the capacity to act as refugia. This also provides a note of caution on how even larger floods are likely to change the refugial characteristics of the catchment over the long term.

The catchment positions of any of the refugia do not seem to have any significant bearing on their conservation importance. There was no consistent longitudinal pattern in refugia size that may relate to

stream power, and hence, the drying pattern of the waterholes did not show any consistent spatial distribution pattern. Rather, the patterns of waterhole persistence and water quality were shaped by the local geology. It might be expected that catchments with more consistently decreasing gradients between highland upper reaches and lowland lower reaches may be more influenced by longitudinal gradients in the distribution of aquatic refugia. However, the pattern in the Neales River emphasizes the need to collect data and carefully consider the refugia capacity at the catchment scale.

The effects of salinity variations on refugia are not commonly considered. For instance, Magoulick and Kobza (2003) identified temperature, pH, dissolved oxygen and nutrients as the dominant water quality influences on refugia. However, variability of the salinity of waterbodies during the no-flow phases in arid zone rivers has been recognised as a driver of biotic assemblages, particularly for invertebrates and algae (Costelloe et al. 2005c; Jolly et al. 2008; Sheldon and Fellows 2010). The spatial and temporal variability of saline pools in the Neales catchment represent an extreme example of salinity variations (i.e. ranging from fresh to hypersaline), which exert considerable influence on the capacity of a waterhole to act as an aquatic refuge for fish, as well as for invertebrates and algae (Costelloe et al. 2005c; Shiel et al. 2006) and also may present barriers to the migration of aquatic fauna during low flows. Their salinity trajectory is governed by the degree of mixing between fresh surface water and saline unconfined groundwater, and further evapo-concentration of this mixture. The rapidly rising salinity of the saline pools during the low recession flow and during disconnection will have dramatic implications for the aquatic fauna that can utilise these pools (Jolly et al. 2008). For instance, the saline waterholes only provide medium-term (i.e. <12 months) refuges for salt-tolerant fish. For instance, the highest salinity tolerance reported for any of the fish species of the Lake Eyre Basin was 110 000 mg l⁻¹ for Lake Eyre hardyheads (*Craterocephalus eyresii*, Glover and Sims 1978), and only a few algal species can survive in these hypersaline pools (Costelloe et al. 2005c). As a result, probably all of the recognised saline refuges would not function as long-term refuges, even for salt-tolerant fish species, and this significantly decreases their conservation value.

In terms of the fish fauna of the catchment, the complete drying of fluvial refugia (other than the artesian springs on the floodplain) would see catchment-wide extinction of a number of larger species, and reintroduction would require the migration of fish from larger connecting catchments, through Lake Eyre North. Such an occurrence would probably only occur during significant interannual wet periods, as the migration requires Lake Eyre North to hold sufficient water with only moderate salinity and then flows from the Neales to reach Lake Eyre North while it is inundated (Fensham et al., 2011). Genetic studies of the largest fish in the catchment, *M. ambigua*, have demonstrated close relationships between the Neales and the Diamantina catchment (large catchment to the northwest of the Neales also flowing into Lake Eyre North), implying relatively recent migration between the two catchments (Faulks et al. 2010). So it is probable that complete drying of fluvial refugia and then re-connection with other catchments has occurred in the Holocene.

Conservation implications

This study emphasizes the value in collecting field data to identify aquatic refugia with the highest conservation value based on hydrological criteria. The distribution of refugia can likely be determined from existing datasets (e.g. topographic maps), but depth and water quality data for individual waterbodies, and the occurrence of flow events, require the collection of field data.

In addition to identifying the key aquatic refugia, the field data provide insights into dominant ecohydrological processes, such as the vulnerability of the fluvial refugia to dessication and why the fish assemblage at the catchment scale shows no endemism. In the case of the Neales catchment, only one fluvial waterhole (Algebuckina) would retain water after a period of 24 months without streamflow, and other waterholes connected to local unconfined groundwater are likely to have reached salinity levels toxic to most aquatic species. Thus, despite a lack of endemism or high biodiversity in the fish assemblage of the Neales catchment, this hydrological assessment of aquatic refugia clearly identifies Algebuckina Waterhole as having the highest conservation importance of the fluvial waterholes and would benefit from management plans that protect this site from local scale, deleterious effects, such as water extraction or over-fishing by recreational fishers. The springs fed by the regional GAB groundwater system form the only long-term aquatic refugia in the catchment but their hydrological characteristics are only suitable for a small range of fish species (Fensham et al., 2011). Even so, the springs with the best connection with the river system would also benefit from management plans that protect these habitats from over-grazing and trampling by cattle. The results clearly demonstrate catchment-scale conservation priorities and the need to identify and protect the hydrological and ecological values of the key fluvial waterholes and springs and illustrate their vulnerability to long-term droughts, climate change and water resource use.

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REFERENCES

- Arthington AH, Balcombe SR, Wilson GA, Thoms M, Marshall J. 2005. Spatial and temporal variation in fish-assemblage structure in isolated waterholes during the 2001 dry season of an arid-zone floodplain river, Cooper Creek, Australia. *Marine and Freshwater Research* 56:25–35. DOI: 10.1071/MF04111.
- Balcombe SR, Bunn SE, Arthington AH, Fawcett FJ, McKenzie-Smith FJ, Wright A. 2007. Fish larvae, growth and biomass in an Australian arid zone river: links between floodplains and waterholes. *Freshwater Biology* 52: 2385–2398. DOI:10.1111/j.1365-2427.2007.01855.x.
- Box JB, Duguid A, Read RE, Kimber RG, Knapton A, Davis J, Bowland AE. 2008. Central Australian waterbodies: the importance of permanence in a desert landscape. *Journal of Arid Environments* 72: 1395–1413. DOI: 10.1016/j.jaridenv.2008.02.022.
- Bunn SE, Arthington AH. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* 30: 492-507.
- Bunn SE, Thoms MC, Hamilton SK, Capon SJ. 2006. Flow variability in dryland rivers: boom, bust and the bits in between. *River Research and Applications* 22: 179–186. DOI: 10.1002/rra.904.

402 Carini G, Hughes JM, Bunn SE. 2006. The role of waterholes as 'refugia' in sustaining genetic diversity
 403 and variation of two freshwater species in dryland river systems (Western Queensland,
 404 Australia). *Freshwater Biology* 51: 1434–1446. DOI: 10.1111/J.1365-2427.2006.01585.X.

405 Costelloe JF, Hudson PJ, Pritchard JC, Puckridge JT, Reid JRW. 2004. ARIDFLO scientific report:
 406 environmental flow requirements of arid zone rivers with particular reference to the Lake Eyre
 407 Drainage Basin. Final Report to South Australian Department of Water, Land and Biodiversity
 408 Conservation and Commonwealth Department of Environment and Heritage,
 409 <http://www.lebmf.gov.au/publications/aridflo.html>.

410 Costelloe JF, Grayson RB, McMahon TA. 2005a. Modelling streamflow for use in ecological studies in a
 411 large, arid zone river, central Australia. *Hydrological Processes* 19: 1165–1183. DOI:
 412 10.1002/hyp.5558.

413 Costelloe JF, Grayson RB, McMahon TA, Argent RM. 2005b. Spatial and temporal variability of water
 414 salinity in an ephemeral arid-zone river, central Australia. *Hydrological Processes* 19: 3147–
 415 3166. DOI: 10.1002/hyp.5837.

416 Costelloe JF, Powling J, Reid JRW, Shiel RJ, Hudson P. 2005c. Algal diversity and assemblages in arid zone
 417 rivers of the Lake Eyre Basin, Australia. *River Research and Applications* 21: 337–349. DOI:
 418 10.1002/rra.851.

419 Costelloe JF, Shields A, Grayson RB, McMahon TA. 2007. Determining loss characteristics of arid zone
 420 river waterbodies. *River Research and Applications* 23: 715–731. DOI: 10.1002/rra.991.

421 Faulks LK, Gilligan DM, Beheregaray LB. 2010. Islands of water in a sea of dry land: hydrological regime
 422 predicts genetic diversity and dispersal in a widespread fish from Australia's arid zone, the
 423 golden perch (*Macquaria ambigua*). *Molecular Ecology* 19: 4723–4737. DOI: 10.1111/j.1365-
 424 294X.2010.04848.x. CONSERVATION PRIORITIES OF ARID ZONE AQUATIC REFUGIA 1543
 425 *Ecohydrol.* 7, 1534–1544 (2014)

426 Fensham RJ, Silcock JL, Kerecsy A, Ponder W. 2011. Four desert waters: setting arid zone wetland
 427 conservation priorities through understanding patterns of endemism. *Biological Conservation*
 428 144: 2459–2467. DOI:10.1016/j.biocon.2011.06.024.

429 Glover CJM, Sim TC. 1978. Studies on central Australian fishes: a progress report. *South Australian*
 430 *Naturalist* 52: 35–44.

431 Hamilton SK, Bunn SE, Thoms MC, Marshall JC. 2005. Persistence of aquatic refugia between flow pulses
 432 in a dryland river system (Cooper Creek, Australia). *Limnology and Oceanography* 50(3): 743–
 433 754.

434 Hoagstrom CW, Berry CR. 2006. Island biogeography of native fish faunas among Great Plains drainage
 435 basins: basin scale features influence composition. *American Fisheries Society Symposium* 48:
 436 221–264.

437 Hudson P, Sheldon F, Costelloe JF. 2003. Aquatic macroinvertebrate biodiversity in the western Lake
 438 Eyre Basin: the role of naturally fluctuating salinity. *Records of the South Australian Museum*
 439 *Monograph Series* 7: 135–144.

440 Jolly ID, McEwan KL, Holland KL. 2008. A review of groundwater–surface water interactions in arid/semi-
 441 arid wetlands and the consequences of salinity for wetland ecology. *Ecohydrology* 1: 43–58. DOI
 442 10.1002/eco.6.

443 Junk WJ, Brown M, Campbell IC, Finlayson M, Gopal B, Ramberg L, Warner BG. 2006. The comparative
 444 biodiversity of seven globally important wetlands: a synthesis. *Aquatic Sciences* 68: 400–414.
 445 DOI: 10.1007/s00027-006-0856-z.

446 Keleher MJ, Rader RB. 2008. Dispersal limitations and history explain community composition of
 447 metaphyton in desert springs of the Bonneville Basin, Utah: a multiscale analysis. *Limnology and*
 448 *Oceanography* 53: 1604–1613. DOI: 10.4319/lo.2008.53.4.1604.

449 Kodric-Brown A, Brown JH. 2009. Native fishes, exotic mammals, and the conservation of desert springs.
 450 *Frontiers in Ecology and the Environment* 5: 549–553. DOI: 10.1890/070002.

451 Lancaster J, Belyea LR. 1997. Nested hierarchies and scale-dependence of mechanisms of flow refugium
 452 use. *Journal of the North American Benthological Society* 16: 221–238. DOI: 10.2307/1468253.

453 Larned ST, Datry T, Arscott DB, Tockner K. 2010. Emerging concepts in temporary-river ecology.
 454 *Freshwater Biology* 55: 717–738. DOI: 10.1111/j.1365-2427.2009.02322.x.

455 Lutton S, Sheldon F, Bunn SE. 2010. Morphological characteristics of onfarm water storages and their
 456 similarities to natural water bodies in the Border Rivers Catchment, Australia. *Aquatic*
 457 *Conservation: Marine and Freshwater Ecosystems* 20: 47–57. DOI: 10.1002/aqc.1072.

458 Magoulick DD, Kobza RM. 2003. The role of refugia for fishes during drought: a review and synthesis.
 459 *Freshwater Biology* 48: 1186–1198.

460 Maltchik L, Medeiros ESF. 2006. Conservation importance of semi-arid streams in north-eastern Brazil:
 461 implications of hydrological disturbance and species diversity. *Aquatic Conservation: Marine and*
 462 *Freshwater Ecosystems* 16: 665–677.

463 McNeil DG, Schmarr DW. 2009. Recovery of Lake Eyre Basin fishes following drought: 2008/09 fish
 464 survey report. SARDI Publication No. F2009/000407-1. South Australian Research and
 465 Development Institute (Aquatic Sciences), Adelaide, 61pp.

466 Mittermeier RA, Myers N, Thomsen JB, da Fonseca JAB, Olivieri S. 1998. Biodiversity hotspots and major
 467 tropical wilderness areas: approaches to setting conservation priorities. *Conservation Biology*
 468 12: 516–520.

469 Murphy NP, Adams M, Austin AD. 2009. Independent colonization and extensive cryptic speciation of
 470 freshwater amphipods in the isolated groundwater springs of Australia’s Great Artesian Basin.
 471 *Molecular Ecology* 18: 109–122. DOI: 10.1111/j.1365- 294X.2008.04007.x.

472 Perez KE, Ponder WF, Colgan DJ, Clark SA, Lydeard C. 2005. Molecular phylogeny and biogeography of
 473 spring-associated hydrobiid snails of the Great Artesian Basin, Australia. *Molecular*
 474 *Phylogenetics and Evolution* 34: 545–556. DOI: 10.1016/j.ympev.2004.11.020.

475 Prescott JR, Habermehl MA. 2008. Luminescence dating of spring mound deposits in the southwestern
 476 Great Artesian Basin, northern South Australia. *Australian Journal of Earth Sciences* 55: 167–181.
 477 DOI: 10.1080/08120090701689340.

478 Sheldon F, Fellows CS. 2010. Water quality in two Australian dryland rivers: spatial and temporal
 479 variability and the role of flow. *Marine and Freshwater Research* 61: 864–874. DOI:
 480 10.1071/MF09289.

481 Sheldon F, Bunn SE, Hughes JM, Arthington AH, Balcombe SR, Fellows CS. 2010. Ecological roles and
 482 threats to aquatic refugia in arid landscapes: dryland river waterholes. *Marine and Freshwater*
 483 *Research* 61: 885–895. DOI: 10.1071/MF09239.

484 Shiel RJ, Costelloe JF, Reid JRW, Hudson P, Powling J. 2006. Zooplankton diversity and assemblages in
 485 arid zone rivers of the Lake Eyre Basin, Australia. *Marine and Freshwater Research* 57: 49–60.
 486 DOI: 10.1002/aqc.1072.

487 Wager R, Unmack PJ. 2000. *Fishes of the Lake Eyre Catchment of central Australia*. Department of
 488 Primary Industries and Queensland Fisheries Service, Brisbane.

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493 **Figure 1.** Position of surveyed waterholes (crosses) on a grey-scale digital elevation model of the Neales
 494 River catchment. Areas of high elevation are white, and areas of low elevation are dark grey. The dashed
 495 ovals outline reaches with poor channelisation and no significant residual pools. Waterhole names are
 496 shown in Table I. Also shown are the positions of mound springs (x, unpublished data courtesy of Dr
 497 Wendy Welsh, Australian Bureau of Rural Sciences), named free-flowing bore drains (Big Blythe and One
 498 Mile, squares), small dams (triangles) and water-level monitoring sites (circles, non-waterhole sites
 499 named in text are AR – Arckaringa Creek and OC – Ockenden Creek).

Figure 2. Cease-to-flow depths for waterholes in the Neales River catchment. The range in values measured for Algebuckina is shown by the error bars. Codes for waterholes are shown in Table I. Horizontal bars show modelled open-water evaporation losses with error range for 1 year (black solid and dashed lines) and 2 years (grey solid and dashed lines) without inflow.

Figure 3. Observed loss rates from Algebuckina and South Stewart waterholes versus modelled open-water evaporation rates using the Penman combination equation. Shown are the modelled evaporation rates and upper and lower error bounds.

Figure 4. Top panel – hydrograph at Ockenden Creek (solid line) and Algebuckina Waterhole (dashed line). Horizontal bar shows level at which flow in Ockenden Creek connects with wetland formed by Ockenden Spring. Lower panel – hydrograph at North Freeling spring pool with horizontal bar showing approximate bankfull level. Stage given in Australian Height Datum.

Figure 5. Stage hydrograph and water salinity measurements at the Peake Creek crossing pool.

Figure 6. Plots show general relationship for evaporation loss with time. The upper plot shows the modelled evaporation loss rates. The lower plot shows the mean monthly observed loss rates for the leaky waterholes (SS, ST, CC), which include evaporation and leakage losses. Each curve shows the loss rate starting in a different month (i.e. 'Jan' curve shows loss rate if flow ceases in December and evaporative loss commences in January). The cease-to-flow depths of a selection of the deeper waterbodies are shown (see Table I for codes). Where these lines intersect, the loss rate curves show when that particular waterhole will dry out after flow ceases.

Table I. Fish species sampled in the Neales River catchment (Costelloe et al., 2004).

Family	Species	Size range (mm)	Trophic group
Atherinidae	<i>Craterocephalus eyresii</i>	<70	Microphagic carnivore
Clupeidae	<i>Nematalosa erebi</i>	<360	Detritivore (herbivore)
Gobiidae	<i>Chlamydogobius eremius</i>	<60	Microphagic omnivore
Melanotaeniidae	<i>Melanotaenia splendida tatei</i>	<125	Microphagic omnivore
Percichthyidae	<i>Macquaria ambigua sp.B</i>	<510	Macrophagic carnivore
Plotosidae	<i>Plotosus hyrtl</i>	<280	Macrophagic carnivore
	<i>Porochilus argenteus</i>	<260	Macrophagic carnivore
Terapontidae	<i>Amniataba percoides</i>	<160	Microphagic carnivore
	<i>Leiopotherapon unicolor</i>	<280	Macrophagic carnivore
Poeciliidae	<i>Scortum barcoo</i>	<410	Macrophagic omnivore
	<i>Gambusia holbrooki</i> ^a	<55	Microphagic carnivore

^a Denotes exotic species.

Table II. Waterhole morphology characteristics (nm – not measured).

Waterhole	Type	Maximum cease to flow depth (m)	Bankfull width (m)	Bankfull depth (m)	Length (m)
Afghan (AF)	River channel	1.20	32	2.2	450
Angle Pole (AN)	River channel	2.16	24	2.7	670
Shepherds (SH)	River channel	1.70	22	2.3	730
Hookey (HO)	River channel	2.56	34	3.9	530
Mathieson (MA)	River channel	2.73	59	3.3	630
Stewart (ST)	River channel	3.23	52	3.7	940
South Stewart (SS)	River channel	2.53	23	3.9	470
Cramps Camp (CC)	River channel	3.85	44	4.4	1130
Fish Hole (FH)	Tributary	1.16	47	1.8	380
Hagans Hole (HH)	Tributary	1.20	nm	nm	740
Algebuckina (AL)	River channel	4.70	70	7.9	2700
South Cliff (SC)	River channel	2.50	85	3.3	490
Cliff (CL)	River channel	1.28	24	2.2	700
Tardetakarinna (TA)	River channel	2.20	40	3.6	1600
Warrarawoona (WA)	River channel	2.00	55	4.7	1260
North Freeling (NO)	Spring pool	0.30	32	nm	750
Baltucoodna (BA)	River channel	2.30	37	4.4	960
Peake Crossing (PE)	River channel	1.50	54	4.7	690
Cootanoorina (CO)	River channel	2.10	50	2.5	920
Birribiana (BI)	River channel	1.80	83	2.5	470
Eaglehawk (EA)	Dam	3.13	80	nm	80
Slate Hole (SH)	Dam	3.83	42	nm	70

Locations shown in Figure 1.

530 **Table III.** Waterhole electrical conductivity characteristics from data collected in the period 2000–2011.

Waterhole	<i>n</i>	Mean	Standard deviation	Median	Min	Max
Afghan (AF)	2	207	—	—	171	242
Angle Pole (AN)	2	173	—	—	143	203
Shepherds (SH)	1	227	—	—	—	—
Hookey (HO)	6	317	191	255	189	700
Mathieson (MA)	6	305	169	242	179	631
Stewart (ST)	9	3647	5047	1618	194	15,560
South Stewart (SS)	13	564	382	460	194	1317
Cramps Camp (CC)	4	674	451	548	284	1315
Fish Hole (FH)	6	846	607	574	468	2045
Hagans Hole (HH)	1	760	—	—	—	—
Algebuckina (AL)	15	8661	9377	6420	372	32,748
South Cliff (SC)	9	1112	922	617	494	3096
Cliff (CL)	2	1798	426	3170	—	—
Tardetakarinna (TA)	6	76,669	95,578	41,824	11,891	260,300
Warrarawoona (WA)	3	4852	3568	6477	761	7319
North Freeling (NO)	3	6337	1358	6709	4831	7470
Baltucoodna (BA)	3	45,911	32,853	53,836	9821	74,080
Peake Crossing (PE)	15	123,424	141,152	76,666	4905	520,550
Cootanoorina (CO)	5	242	110	167	155	388
Birribiana (BI)	3	329	134	280	226	481
Eaglehawk (EA)	1	811	—	—	—	—
Slate Hole (SH)	2	166	—	—	160	172

531 *All measurements in $\mu\text{S cm}^{-1}$ corrected to 25 °C.*

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533 **Table IV.** Number of observed flow events and corrected for data availability.

Reach	Data availability (%)		Flow events		Corrected flow events		No flows recorded	
	Feb00 to Oct09	Nov09 to Apr11	Feb00 to Oct09	Nov09 to Apr11	Feb00 to Oct09	Nov09 to Apr11	Feb00 to Oct09	Nov09 to Apr11
Catchment	—	—	24	22	—	—	—	—
Upper Neales	79.8	96.4	14	19	17.5	19.7	5	3
Lower Neales	80.3	95.7	14	20	17.4	20.9	6	1
Upper Peake	73.6	96.4	11	10	14.9	10.4	7	11
Lower Peake	66.2	34.2	15	8	22.7	23.4	2	5

534 *Final two columns show number of events that were not recorded in a reach when flows were occurring in other reaches.*

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