Journal of Environmental Management 393 (2025) 127184

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Conservation planning for environmental water to climate refugia in the manageable Murray–Darling Basin

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ARTICLE INFO

Handling editor: Jason Michael Evans

Keywords:
Catchment management
Complementarity
Freshwater conservation
Conservation prioritisation
Climate change planning
Environmental flows
Water management
Wetlands and lakes

ABSTRACT

One mechanism for improving the resilience of freshwater systems affected by climate change is to use environmental water to support refugial habitats which allow species, ecosystems and functions to persist and recover after severe droughts. We applied systematic conservation planning (SCP) to prioritise wetlands and lakes with the aim of informing the delivery of environmental water for the creation and protection of refugia habitat in the Murray-Darling Basin, Australia. SCP uses a complimentary algorithm to generate planning solutions that protect all target ecological assets for the lowest "cost" of the management constraints considered. Here the ecological assets were 294 wetland dependant taxa including species of fish, frogs, dragonflies, crustacea, molluscs, and plants, 42 different ecosystem types and ecosystem productivity. Managements constraints included resistance to drying, condition and connectivity and the ease of environmental water delivery. Conservation inundation targets were aligned with the approximate annual delivery of environmental water by the Commonwealth Environmental Water Holder. We found that prioritisation of sites for enivronmental water was sensitive to the choice of target ecological assets and less so but to some extent the cost of management. We found environmental water delivery in the Basin is reaching refugial wetlands that support the majority of ecosystem and species diversity. However, certain taxonomic groups, such as invertebrates, are comparatively poorly represented. To effectively manage taxa, more data on ecological and life history traits is needed to better identify the spatial and temporal location of their refugia. This case study demonstrates that the SCP approach offers an objective and repeatable process for informing environmental water allocation and delivery, that could be applied to other basins globally.

1. Introduction

Globally, modification of rivers by human activities is now pervasive (Haddeland et al., 2014). This has resulted in the fragmentation of aquatic habitats, biodiversity and ecosystem functions (e.g. Nilsson et al., 2005; Stoffels et al., 2022). Refugial habitats are an important component of freshwater ecosystem resilience as they allow species to persist and then recolonise following disturbances such as drought (e.g. Magoulick and Kobza, 2003). The high biodiversity value of wetlands and lakes and their ability to buffer against climate disturbance has led to their frequent recognition as climate refugia (Morelli et al., 2020; Selwood and Zimmer, 2020). In this context we refer to the general ecological definition of refugia as the spatial contraction of an individual, population or species range due to adverse conditions (Keppel and Wardell-Johnson, 2012). Therefore, drought refugia are areas of higher resource availability and/or habitat quality than elsewhere in the

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https://doi.org/10.1016/j.jenvman.2025.127184
Received 27 February 2025; Received in revised form 18 August 2025; Accepted 1 September 2025
Available online 6 September 2025

landscape, supporting plants and animals during dry times.

Freshwater ecosystems are dendritic and relatively isolated within landscapes, which limits the ability of freshwater taxa to access cooler areas and leaves them exposed to drying (Woodward et al., 2010). Species that are unable to tolerate or shift their range in response to changes in climate will require in-situ management to ensure their survival (Bennett et al., 2021; Greenwood et al., 2016). Identifying, prioritising, creating, and managing areas that provide refuge to biodiversity from drought may be an effective strategy for conservation managers (Bush et al., 2014; Greenwood et al., 2016; Selwood and Zimmer, 2020). However, in large river basins, locating and prioritising wetland refugia for management remains a major challenge.

The Murray-Darling Basin (hereafter 'the Basin') in Australia, is the world's 5th largest river basin containing more than 30,000 wetlands and lakes (Bino et al., 2015). Many of the Basin's wetlands and lakes are important migratory bird habitats and are recognised under the Ramsar Convention (1971), an intergovernmental treaty for international cooperation and national action for the conservation and sustainable use of wetlands (Bino et al., 2015). The Basin's wetlands and lakes occur across a wide range of physical and climatic environments and support a diversity of plants, animals and ecosystems (Rogers and Ralph, 2010). Anthropogenic climate change has caused substantial warming and has led to an increase in the intensity and duration of dry periods as well as an increase in the intensity of floods across the Basin (Whetton and Chiew, 2021). Impacts of increased frequency and intensity of drought are exacerbated by increasing human water needs including extraction for agriculture, manufacturing and potable use (Prosser et al., 2021). Drought impacts in the Basin have been described in detail in general reviews (e.g. Ayele, 2024; Bond et al., 2008; Capon, 2014; Overton and Doody, 2013) and detailed case studies (e.g. Li et al., 2017; Ning et al., 2013; Thompson et al., 2024) over the last two decades. The Basin's wetlands and lakes provide a natural buffer against the Basin's naturally dry and highly variable climate and management of these refugia is a critical part of protecting the Basin's freshwater and terrestrial biodiversity (MDBA, 2021).

Conservation management of refugia requires that managers understand the unique needs of different target species and ecosystems, and the different approaches to selecting potential refugia for conservation (Ashcroft, 2010; Reside et al., 2014). Systematic conservation planning (SCP) is the most commonly applied prioritisation approach for selecting areas for conservation. Although originally developed for terrestrial protected area selection, the SCP approach could be applied to any spatial prioritisation process (Cattarino et al., 2015). The aim of modern SCP is to represent biodiversity in a reserve network in a complementary and cost-effective way to minimise risk, by representing ecological assets while considering management constraints, which can cover a range of societal, economic, environmental or political costs. The SCP approach has been widely applied to terrestrial and marine systems, however freshwater habitats are under-represented within systematic conservation planning on a global scale (Darwall et al., 2011). This is concerning because freshwater habitats support a disproportionate amount of the world's taxa (~6 %), given they represent only ~0.8 % of the Earth's surface (Dudgeon et al., 2006).

A key lever for sustainable environmental management in the Basin is the use of environmental water (synonymous with 'environmental flows' or 'e-flows' *sensu* Poff and Matthews, 2013; Swirepik et al., 2016). Commonwealth environmental water consists of water allocations within the water market which have been obtained from either direct purchase or water efficiency measures (see Johnson et al., 2021 for a review). Environmental water is allocated based on water plans for individual sub-catchments that target particular ecosystem responses (Sharpe et al., 2021). Use of environmental water is subject to operational constraints including where water can be effectively delivered and avoiding negative outcomes such as the flooding of private land. See Swirepik et al. (2016) for a summary and (Bischoff-Mattson and Lynch, 2017) for a discussion of governance arrangements. Management of environmental water aims to deliver water to maintain instream flows or provide wetland or floodplain inundation (Gawne and Thompson, 2023). These flows have a wide range of ecosystem targets including vegetation condition, providing life history cues and sustaining in-channel productivity (Watts et al., 2020). During drought years, it has become common across multiple catchments in the Basin for environmental water to be targeted for the maintenance of refugial habitats to support the survival of specific or multiple taxonomic groups (Prosser et al., 2021). The systematic prioritisation of environmental water to maintain and protect refugia habitat during drought may be an effective management strategy to support the survival and recovery of water-dependent ecological communities (Linke and Hermoso, 2022).

We tested if systematic conservation planning (SCP) could be applied at the basin scale to identify wetlands and lakes for dependant species, ecosystems and functions that could be supported as refugia to drought conditions. Specifically, we ask (1) if SCP can be applied across the Basin to protect 294 species of wetland dependant taxa, including species of fish, frogs, dragonflies, crustacea, molluscs, and plants, 42 different ecosystem types and ecosystem productivity and (2) how management constraints including feasibility, site condition and connectivity, and resistance to drought affect the distribution of selected priority sites?

2. Methods

2.1. Spatial framework

We used the Australian National Aquatic Ecosystems (ANAE) v3.0 Classification of the Basin (Brooks, 2021) as our underlying spatial framework. The ANAE ecosystem classifications were used because they are based on the best available spatial data for wetlands and lakes from the Australian state and Commonwealth governments mapping including the Australian Hydrological Geospatial Fabric (Geofabric, BOM, 2020). The ANAE classification is broadly applied by relevant management authorities across the Basin including the Murray–Darling Basin Authority (MDBA) and the officers of the Commonwealth Environmental Water Holder (CEWH) to support monitoring, evaluation, and adaptive management of water resources.

Management recommendations need to be restrained by a realistic assessment of achievable outcomes. Accordingly, we tempered our analysis to reflect the constraints on where held environmental water can be delivered. Using attributes in the ANAE classification, planning units were refined into 2 subsets for further analysis: (1) planning units identified to be on the managed floodplain and (2) planning units that have received CEWH environmental water according to available records (Fig. 1).

The managed floodplain was defined as per the Basin-wide Environmental Watering Strategy (MDBA, 2014) and includes areas in the Basin where environmental water could likely be delivered within current operational constraints. Actively watered planning units were identified as those that have received Commonwealth environmental water since these allocations became available (between 2014 and 2020). The unmanaged floodplain is generally reliant on natural large flow events for inundation and as such is beyond the scope of managed environmental watering under the Plan (MDBA, 2014). The unmanaged floodplain was not included within our spatial framework and was excluded from analysis. The aim of using these 2 sets of planning units was to determine i) if the areas where environmental water has been delivered are adequate to represent regional diversity across the managed floodplain; and ii) whether alternative sites could better represent regional diversity and better protect ecosystem processes.

2.2. Ecological assets

2.2.1. Ecosystem diversity

To reflect the fact that many species use multiple habitats, we identified the other ecosystems surrounding each planning unit. We

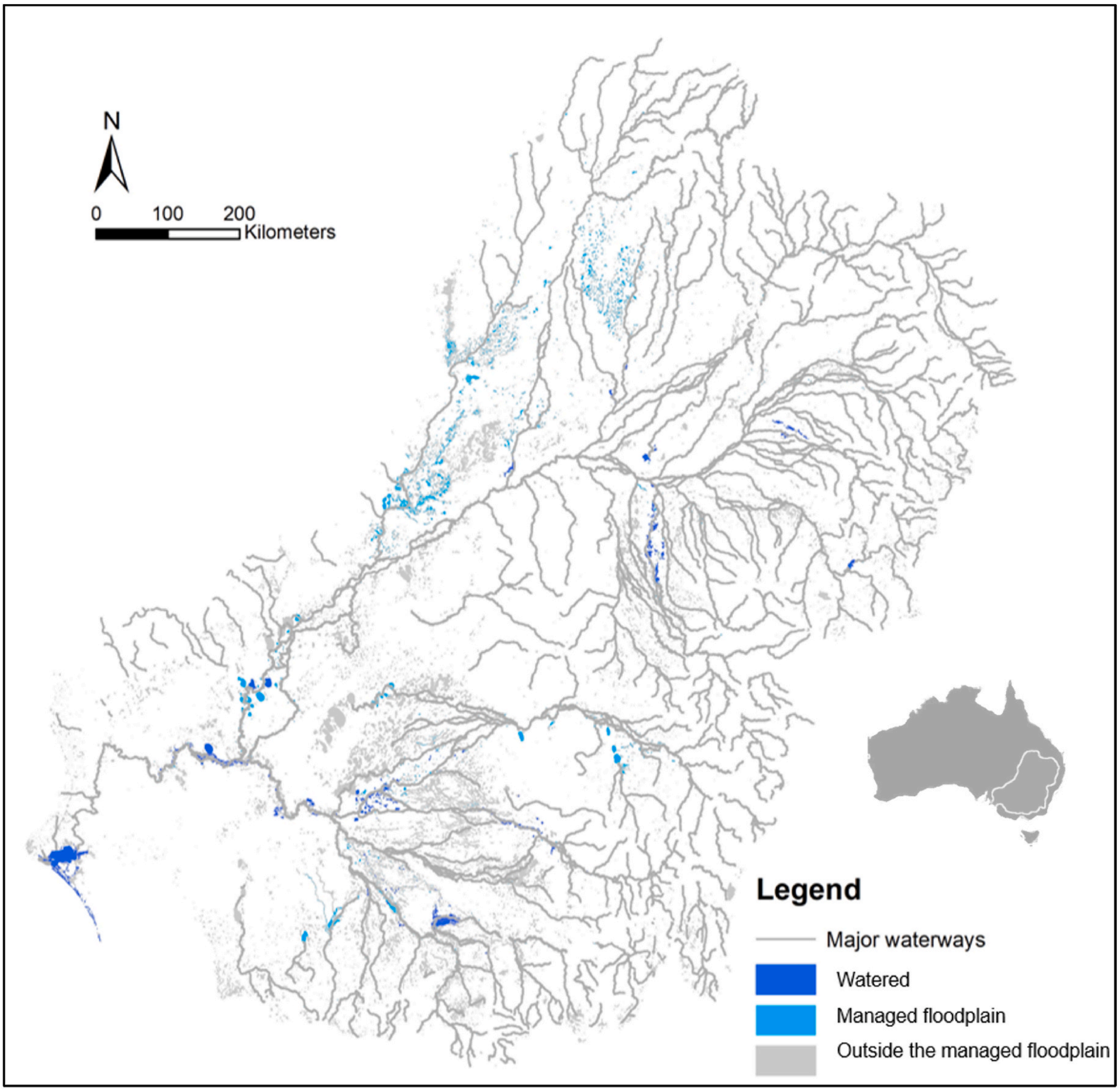

Fig. 1. Depressional wetlands and lakes in the ANAE that are on the managed floodplain (light blue) and have received CEWH delivered environmental water (dark blue). Waterways and wetlands and lakes outside the managed floodplain were not considered in the SPC prioritisation (grey). The map of Australia in grey identifies the location of the Basin outlined in white.

defined the value of a planning unit to each ecosystem in the ANAE classification by identifying each ANAE and its area within a 500 m circle of influence of each planning unit (Supplementary Fig. S1). The ANAE classifies aquatic ecosystems uses attributes relevant to the structure and function of each system (Brooks, 2021). The hierarchical structure of the ANAE is designed to capture spatial patterns at regional and landscape scales and ecological diversity at local scales. The regional and landscapes levels (1 and 2) capture large-scale and mid-scale attributes associated with landform, climate, topography, hydrology and water influence. Level 3 captures local scale attributes such as aquatic ecosystem class (surface water and subterranean), system (e.g. estuarine, lacustrine, riverine, and floodplain) and habitat (e.g. red gum forest). A typology is applied to distil these attributes into distinct aquatic ecosystem classes (e.g. ‘permanent paperbark swamps’ or ‘temporary lakes’). In total the ANAE contains 7948 lakes classified into 8 ecosystem types and 51,830 Palustrine wetlands within 29 different ecosystem types (a list of ecosystem types included in the study can be seen in Table S1).

2.2.2. *Species diversity*

We used a subset of species defined by Rogers and Ralph (2010) as wetland dependant for which distributional data at the scale of the Basin was available, this included 17 species of fish, 73 species of frogs, 87 species of dragonflies, 36 species of crustacea, 33 species of molluscs and 48 species of plants. We used species distribution maps that mapped the probability of occurrence for each taxon at the scale of the Geofabric Level 15 subcatchment (Bush and Hoskins, 2017). The Geofabric maps the Murray–Darling Basin as hierarchically nested catchments, where river basins are sub-divided into successively finer sub-catchments. The lowest level delineates the sub-catchments draining directly to a stream segment (BoM, 2015a). For fish, species distribution models combined state fisheries presence/absence data with spatial data on environmental suitability including climate and catchment physiography. For a full description of model development see (Bond et al., 2014). Habitat suitability models for all other taxa were fitted using a combination of 5 common algorithms; generalised linear models, generalised boosted models, generalised additive models, Maxent and multivariate adaptive regression splines (Buisson et al., 2010; Elith et al., 2006; Thuiller et al., 2009).

We assigned the probability of a species occurring to each planning unit to be the same as that of the surrounding Geofabric Level 15 sub-catchment. When planning units spanned multiple sub-catchments, we assigned the mean probability of occurrence from the surrounding sub-catchments to that planning unit. To calculate the habitat value of each planning unit to a particular taxon, we multiplied the probability of occurrence by the area of each planning unit. This was done so that larger planning units, that contained more habitat, were deemed more valuable that smaller planning units with the same probability of occurrence.

2.2.3. Productivity

The ability of lake and wetland planning units to act as refugia for higher trophic taxa such as fish and birds is dependent on basal energy resources. High ecosystem productivity is positively linked to diversity through multiple mechanisms (Stendera et al., 2012; Waide et al., 1999). To quantify productivity in each planning unit, we estimated carbon sequestration from harmonised global maps of above ground living biomass carbon density for the year 2010 (Spawn et al., 2020, Fig. S2). The harmonised above ground living biomass carbon density map integrates published remotely sensed maps on all major components of living biomass (e.g. woody, herbaceous and crop biomass) from all above ground living plant tissues (stems, bark, branches, twigs) and therefore allows for a holistic accounting of diverse vegetation carbon stocks. For each planning unit, productivity was estimated as the mean carbon sequestration (mg per hectare) for the perimeter of each planning unit multiplied by the area of the planning unit. Larger planning units such as wetlands and lakes may contain deep open water in the middle and, therefore may have zero carbon sequestration despite being highly productive and potentially important habitat and refugia. For this reason, we used the mean carbon sequestration for the perimeter of each planning unit to not devalue these large planning units. In doing this we are assuming aquatic primary productivity is positively correlated with terrestrial productivity in these systems as it is elsewhere (Grasset et al., 2016)

3. Management constraints

3.1. Habitat area

Large planning units contain more habitat. However, larger planning units also require more water to achieve the same standing water level compared to small planning units. Therefore, the area of each planning units was used as a management cost.

3.2. Habitat condition and connectivity

More disturbed areas have a lower conservation value because degraded habitats are less suitable and/or less available to species. Further, highly disturbed sites may have other associated ecological, social or economic costs that need to be considered before conservation actions can successfully achieve their goals. For example, disturbed sites may need considerable restoration before they can support viable species populations and in-turn diverse communities. For our SCP process we therefore quantified the habitat condition of a planning unit.

Measuring the habitat condition of a planning unit requires a multi-scale perspective. The condition of the surrounding landscape in which a wetland or lake is located will likely have the largest effect on condition; however, upstream catchment condition will also play a role due to hydrological connectivity. Simply, flow through dendritic freshwater systems means the negative effects of anthropogenic disturbance, and conversely, the positive effects of more natural areas in upstream catchments can propagate downstream (Hermoso et al., 2011). Further, allocating environmental water could have unintended consequences if it reconnects degraded sites to the network and causes disturbances (e.g. pollution) to propagate downstream (Hermoso et al., 2012).

We accounted for river condition using the River Disturbance Index (RDI, Stein et al., 2014) which has been calculated for all sub-catchments of the Basin in the Australian Hydrological Geofabric (BoM, 2015b). The RDI numerically characterises anthropogenic river disturbance assigning a value ranging between 0 and 1, from pristine to severely disturbed (Fig. S3). The RDI is an estimate of the extent and intensity of anthropogenic disturbances in a river catchment due to land-use and infrastructure such as roads and flow-regime disturbance due to impoundments, flow diversions and levee banks. To account for hydraulic connectivity, the disturbance index was calculated for sub-catchments then weighted by the mean disturbance of all upstream sub-catchments. We assumed that the condition of a lake/wetland planning unit was the same as the condition of the Geofabric Level 15 sub-catchment in which the wetland or lake occurs and assigned to each planning unit the RDI value of the surrounding sub-catchment.

3.3. Resistance to drought

Wetting and drying phases create boom-bust-cycles of resource availability which in turn affect species abundance, recruitment and distributions, and habitat availability, water quality and ecosystem processes (Bunn et al., 2006). Inter-annual flow variability in the Basin is primarily driven by the El Niño Southern Oscillation (ENSO), however, under climate change the intensity of drought periods and severity of floods is increasing (Whetton and Chiew, 2021). The long-term persistence of populations under disturbance is not only determined by their ability to persist (resistance) but also their ability to recover (resilience) (Bennett et al., 2014). Refugia habitats are areas in the landscape where resource availability is consistently high and relatively resistant to bust periods, supporting survival and facilitating resilience by providing a base for recruitment and re-colonisation when conditions improve in the surrounding landscape (Selwood and Zimmer, 2020).

The recent linking of the ANAE classification and the Digital Earth Australia Wetlands Insight Tool (WIT) (GeoSciences Australia, 2024) which contains data on the amount of water, green vegetation, dry vegetation, and bare soil, between the years 1986 until 2022 (Hale et al., 2023) means it is now possible to assess changes in the condition of wetlands and lakes through time at the basin scale. The combination of these two datasets enables the application of SCP to wetlands and lakes, including an assessment of their condition as a product of changes in climate as a critical assessment of their physical refugia qualities (e.g. their resilience as a function of condition in relation to drought).

To identify wetlands and lakes that may maintain their condition during low flow periods we calculated a dryness anomaly using data from the WIT (Dunn et al., 2019). We calculated the dryness anomaly for each planning unit as the median increase in bare soil for the period 2017 until 2019 compared to the median bare soil in the historical record (1986–2022) (Fig. 2). We used the recent dry conditions of 2017–2019 to investigate wetland resistance to drought as it was one of the most extreme Basin-scale multiple-year rainfall deficits (BOM, 2022, 2019). The dryness anomaly was calculated for wetlands and lakes >1 ha, as ANAE polygons smaller than 1 ha are considered too small to be reliably measured using the Landsat data sets that are incorporated into the WIT.

3.4. Analyses

3.4.1. Prioritisation used the MARXAN algorithm

For the prioritisation of planning units that are most likely to be refugia for the conservation of target taxa, ecosystems and ecosystem productivity that should be considered as a priority for environmental water, we used the program MARXAN (Ball et al., 2009). MARXAN uses a simulated annealing algorithm to identify a set of planning units that maximises the representation of ecological asset, while aiming to capture a defined target for each asset and minimise management cost. Here our management target was that all ecological assets should to be

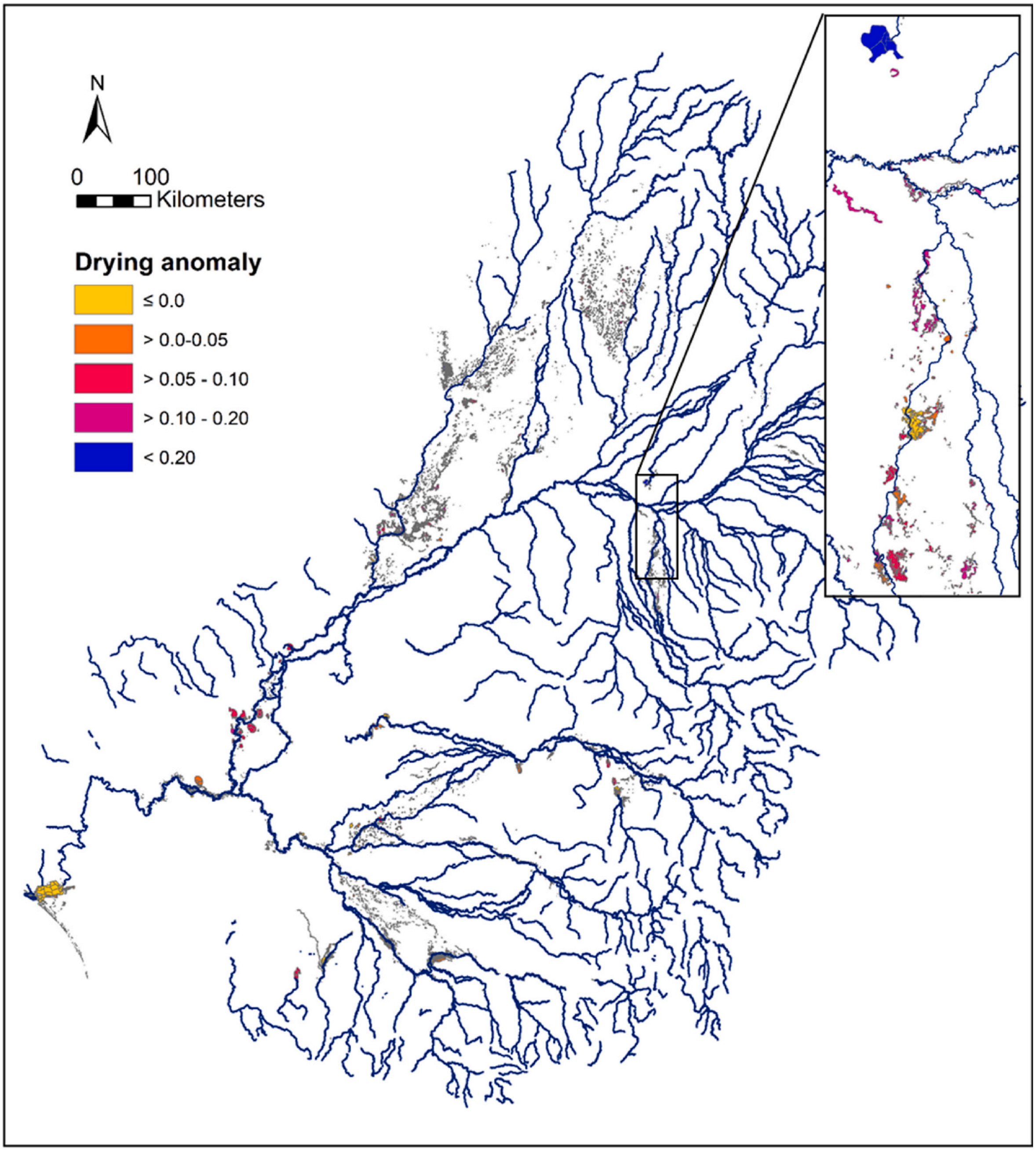

Fig. 2. Dryness anomaly for each planning unit, calculated as the medium increase in bare soil over the period 2017 to 2019 compared to the long-term medium (1986–2022).

represented in all solution. The target area for conservation was based on the average inundated area of wetlands by Commonwealth environmental water per year. In the water years between 2014 and 2019, on average ~150,000 ha (range 117,965 to 171,296 ha) of wetlands and lakes in the Basin received environmental water per year. Therefore, the MARXAN conservation targets were adjusted until a scenario in which inundated planning units collectively totalled approximately 150,000 ha in area, which was that each ecological asset was represent in at least on average 150 ha.

To account for the importance of connectivity, we included a boundary layer in MARXAN to identify adjoining planning units. To force the algorithm to preferentially select adjoining planning units the maximum boundary penalty was applied in all analyses.

3.4.2. Cost

The 'cost' of a planning unit was weighted by three constraints (1) its capacity to act as a refuge (dryness anomaly), (2) catchment condition and connectivity (RDI) and (3) management feasibility (previous environmental water delivery). The rationale for the weighting is outlined in Linke et al. (2012) and it is used so that when planning units are of equal biodiversity value, the algorithm will prioritise planning units that dry out less, in low disturbance catchments, that can be watered, over planning units that are less resistant to drying, are in more disturbed catchments and have no prior history of watering suggesting watering delivery may be difficult. If a planning unit is important to a highly unique asset, the weighting will not affect its selection as the irreplaceability of the planning unit will override the weighting (Linke et al., 2012).

First, the RDI value, and the dryness anomaly of each planning unit was scaled between 0.3 and 1. The 0.3 to 1 range has been shown to allow for effective comparison been planning units, without overriding the prioritisation as shown in Linke et al. (2012). For costs associated with ease of management, planning units that had previously been inundated with environmental water were assigned a weighting value of 0.3, while planning units that had not received environmental water were assigned a value of 1. This was based on the rationale that it must be feasible to deliver environmental water to a planning unit if it had received environmental water in the past. The cost of each planning unit can then be calculated as a weighted average of area multiplied by the scaled RDI, dryness anomaly and/or management feasibility on the managed floodplain.

3.4.3. Scenarios

We aimed to identify key areas in the basin that represent the target ecological assets and if watering action are reaching these key areas. We also sought to determine if management considerations modified the site prioritisations. To achieve this, we compared the distribution of pre-initialised planning units under separate cost scenarios, different levels of feasibility and for individual ecological assets as follows.

1. How different cost constraints influence conservation planning solutions:
   a. **Area scenario**: Area as the only management cost.
   b. **Degradation refugia scenario**: Cost is estimated as the planning unit area weighted by the RDI.
   c. **Climate refugia scenario**: Cost is estimated as the planning unit area weighted by the dryness index.
   d. **Degradation and climate refugia scenario**: Cost is estimated as the average of scenarios a, b and c.
2. We considered how feasibility of environmental water delivery influences conservation planning solutions. Here we only prioritised planning units that had previously received environmental water (watered planning units), as we considered this to be the best indication that a planning unit can receive environmental water. The prioritisation included all ecological assets, and the cost of planning units was calculated as per scenario 1d above.
3. We determined which primary conservation features are driving the prioritisation of the refugia across the managed floodplain when all management cost is considered. To achieve this, we performed analysis on specific ecosystem diversity, ANAE ecosystem classes and productivity measured as carbon sequestration, and on taxonomic groups: crayfish; frogs; molluscs; Odonata; fish and plants.

4. Results

4.1. High priority lakes and wetlands

The prioritisation selected over ~4000 wetlands and lakes for conservation, ~20 % of available planning units in the managed Basin. The selection was highly consistent across all cost scenarios, with ~80 % of planning units consistently selected as either in or out of the best prioritisation networks. Catchment condition and connectivity had the largest effect (~20%) on changes in planning unit selection. Management feasibility (as indicated by previous environmental water delivery) had the smallest effect on changes in prioritisation. There was no apparent geographic effect of the different cost scenarios on the selection of planning units in the best conservation scenarios. Or more specifically, changes in planning unit selection occurred within wetland systems rather than between wetland systems and catchments as the

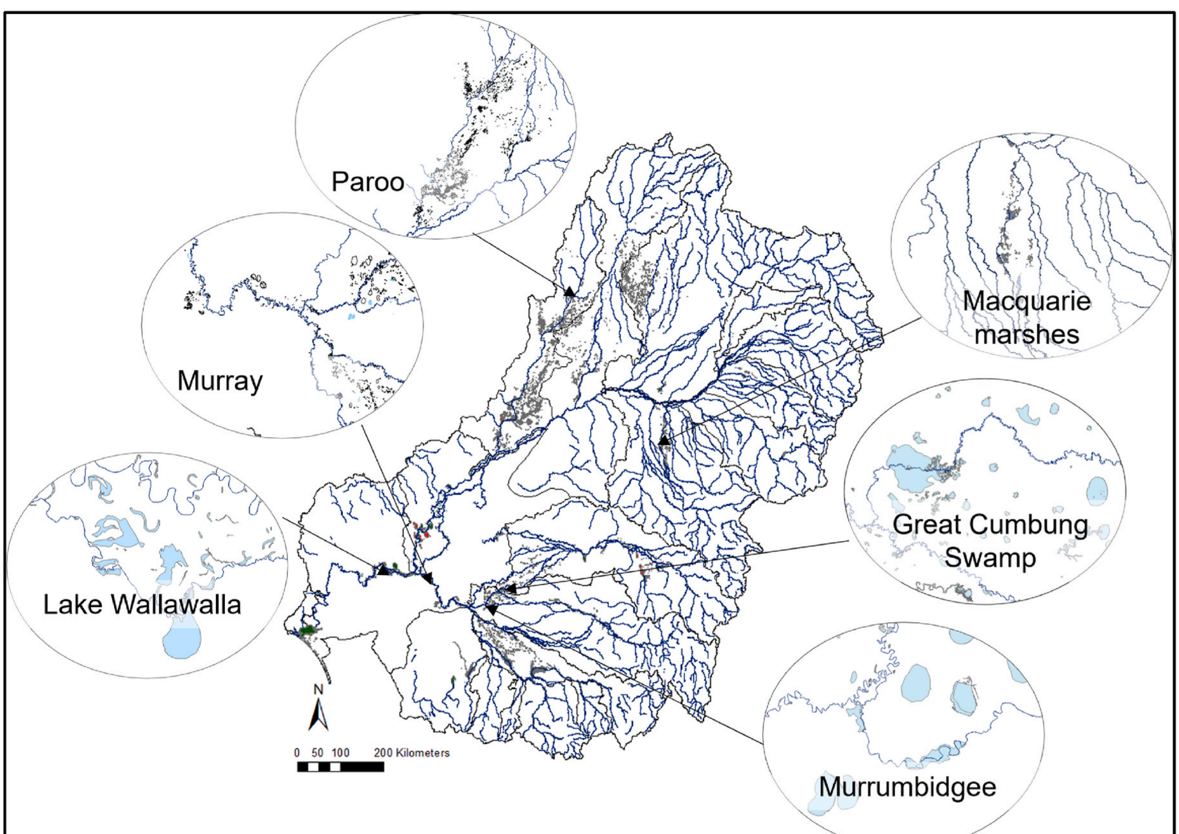

Fig. 3. Key areas prioritised as refugia from disturbance and drying for the conservation of animal, plant and ecosystem diversity and productivity, accounting for management feasibility on the manageable floodplain. Areas identified include Lake Walla Walla and the Great Cumbung Swamp and surrounding wetlands, the Macquarie Marshes, and Wetlands and lakes along the Murrumbidgee, Warrego and Paroo Rivers.

prioritisation consistently picked planning units with high species diversity in distinct regions of the Basin.

Areas identified as high priority refugia, under all cost constraints for all taxa, productivity and ANAE diversity, included (Fig. 3): the wetlands around Lake Alexandrina, the region of Barmah Forest, Lake Wallawalla, Great Cumbung Swamp and surrounding wetlands, Lake Buloke and the Macquarie Marshes, many wetlands and lakes along the Paroo, Murrumbidgee, Warrego, and Murray rivers and some wetlands on the Gwydir, Namoi and Merivale rivers.

Planning units that receive Commonwealth environmental water were well represented in the subset of high conservation value planning units. In the lower Murray, many sites that frequently received Commonwealth environmental water were identified as important for all ecological assets, except ANAE classifications. Planning units in the Murrumbidgee and Warrego Rivers that regularly receive environmental water were also well represented in the SCP (Fig. 4). However, some sites that had not received Commonwealth environmental water in the past that were often selected due to thier importance to ecological assets inlcuded some wetlands and lakes in the Warrego, Lachlan, Border Rivers and Condamine-Culgoa catchments. In particular, multiple wetlands and lakes in the Paroo River catchment were consistently selected in all prioritisation scenarios as important for the representation of ecologcal assets(Fig. 3). However, the Paroo catchment has not recieved Commonwealth environmental water.

4.2. Representation of ecological assets

At the Basin level we considered 294 species with distributions overlapping the depressional wetlands and lakes that comprised the planning units. Within the managed floodplain, 266 species had distributions overlapping with planning units and 219 species had distributions that intersected with planning units that had previously received environmental water (Tables S2–S7). Wetland dependant species with distributions within the Basin, that are not found within the managed floodplain included 9 species of frog, one species of plant, 10 species of crayfish and 8 Odonata. To protect these ecological assets, watering actions in the unregulated floodplain would be needed. Distributions of all species of wetland-dependent fish and molluscs were found within the managed floodplain. All the plant and fish species considered with distributions within the managed floodplain have received Commonwealth environmental water. Species with distributions within the managed floodplain that have not received Commonwealth environmental water included 20 species of frog, 13 species of Odonata, 11 species of crayfish and 3 species of mollusc.

The SCP-prioritisations of planning units were relatively consistent across taxonomic groups and productivity but was less consistent between taxonomic groups and productivity and ANAE classes.

Of the different taxonomic groups considered, fish had the largest effect on the planning units identified as priority areas as refugia for biodiversity conservation.

The mean rate of carbon sequestration was higher in watered planning units (i.e., those that have received environmental water) than the mean rate of carbon sequestration across the managed floodplain (i.e., watered and unwatered planning units on the managed floodplain), 302.37 Mg/ha ±320.46 SD compared to 165.16 Mg/ha ±255.16 SD respectively.

5. Discussion

The ecological health of the Murray-Darling Basin, like many large agriculturally developed regions globally, is considered to be poor (Davies et al., 2010). The use of environmental flows to help rehabilitate these ecosystems is key strategy in place in the Basin, however the adequacy of existing allocations and their use is much debated (e.g. Chen et al., 2020; Colloff et al., 2024; Colloff and Pittock, 2022). The challenge in any river system where there is an effort to balance ecological values with human water needs is how to carry out defensible prioritisation of water use within a finite envelope of water availability (Sheldon et al., 2024).

Prioritisation of areas for the use of environmental water is a multiphase planning process whereby water plans are submitted by state governments to the Murray-Darling Basin Authority, which approves

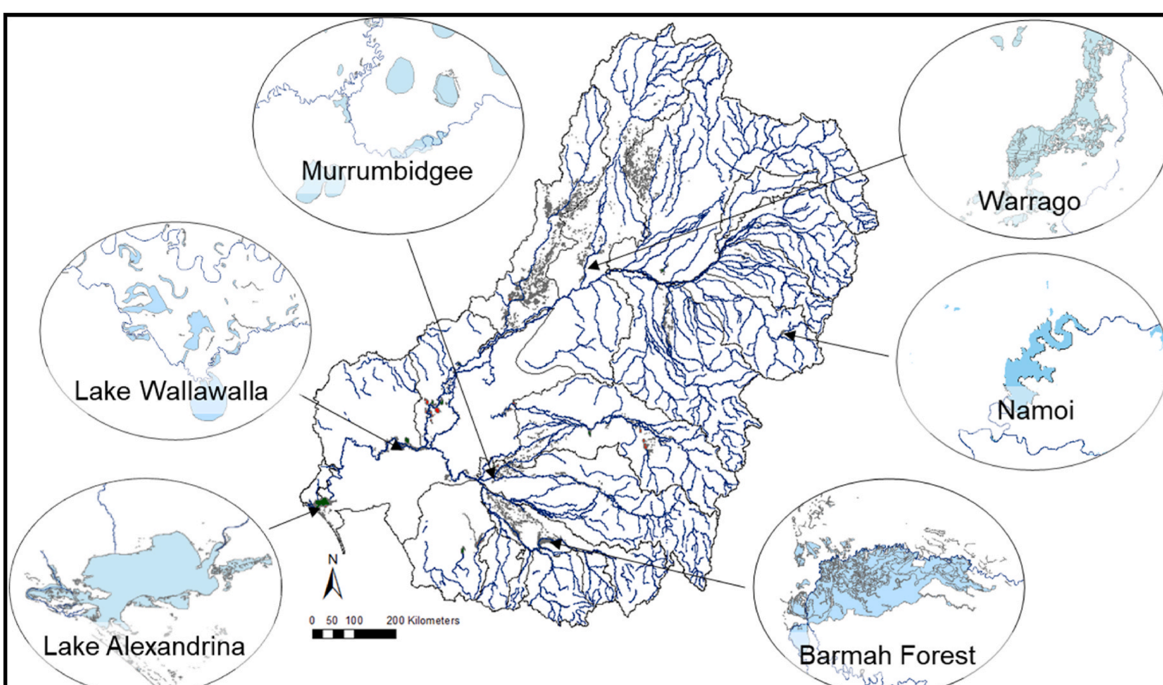

Fig. 4. Key areas prioritised as refugia from disturbance and drying for the conservation of animal, plant and ecosystem diversity and productivity that have received CEWH water. Areas include Lake Alexandrina, Barmah Forest, Lake Wallawalla and surrounding wetlands, Wetlands and lakes along the Murrumbidgee, Warrego and Namoi Rivers and some on the Gwydir.

plans based on the environmental returns which are likely to derive from the environmental water allocation (e.g. Johnson et al., 2021; Sharpe et al., 2021; Wallis and Ison, 2011). However, there are many additional limitations including the relatively small amount of environmental water available, physical constraints on delivery through the river system, and the avoidance of negative outcomes such as the flooding of private land. In some cases, high priority areas for watering may be relatively distant from the main channel and require much larger amounts of water or development of infrastructure to deliver water to them, relative to lower priority areas which are close to channels. Watering of some very high priority sites across the Basin requires complex logistic arrangements (such as pumping, permission to flood private land, development of alternate access points) and favourable climatic context (such as existing high flows) in combination with allocations of environmental water. The complex interplay of environmental water needs, environmental water availability, current and predicted climatic conditions and delivery constraints is navigated by state-based water planners in coordination with basin-scale planning by Commonwealth agencies (Bischoff-Mattson and Lynch, 2017). These values are broadly consistent with those identified as targets when carrying our targeted water reform in the Basin (MDBA, 2014).

Here we demonstrate that it is possible to apply a systematic conservation planning (SCP) method to environmental water allocation for the protection of refugial wetlands and lakes in the Murray–Darling Basin for a set of important ecological values. There have been notable previous applications of SCP for selecting protected areas within the Basin (e.g., Bino et al., 2015; Linke et al., 2015). Bino et al. (2015) used long-term aerial surveys of water birds and applied the SCP methods to identify important wetlands acting as waterbird refugia in wet and drought periods. While, Linke et al. (2015) piloted the use of SCP for prioritising sub-catchments at the Basin-scale and wetlands within the Murrumbidgee catchment for the conservation of a wide range of taxa.

The challenge we have applied SCP to is of international significance. For example, in the Amazon, many fish species which are important to the local economy rely on refugial wetlands (Goulding et al., 2019) and allocating environmental flows for their protection has been highlighted as a possible key management strategy (Couto et al., 2024). Furthermore, the management constraints considered here are globally relevant. For example, in Northern Europe, social and economic pressures often outweigh ecological considerations in environmental water allocations, frequently sidelining critical ecosystem needs (Pahl-Wostl et al., 2013). This underscores the importance of using replicable, and objective decision-making tools to ensure balanced and transparent water management.

5.1. Drivers of the prioritisation networks

We found the prioritisation of planning units was influenced more by their importance to different conservation features than their cost as refugia (i.e., size, condition, connectance, water permanence and feasibility of management). The prioritisation was relatively consistent between taxonomic groups, suggesting that distinct assemblages between geographic regions was driving much of the spatial variation in the prioritisation. Where there was changes in the distribution of selected planning units, the major taxonomic driver was fish species. This is consistent with previous prioritisations within the Basin. For example, Linke et al. (2015) conducted SCP using sub-catchments and found the largest influences to planning unit selection were from fish and plants.

Overall, ANAE type had the largest effect on the prioritisation, causing the largest difference in site selection when considered alone. Even the prioritisation of planning units for ecosystem diversity (ANAE) and targeted plant species was not well correlated despite vegetation types, which are often defined by the dominant plant species or assemblage contributing to the ANAE classifications. The lack of a correlation between the ANAE and plant species prioritisations is likely due to the additional complexity included in the ANAE and the broad dominant vegetation categories used (e.g. Aquatic grass/sedge/forb, Black box, Bogs and fens, River red gum, etc). We recommend that ecologically and culturally important plant species such as *Marsilea drummondii* should be separately targeted if they are the aim of conservation watering actions.

The different cost scenarios did not appear to strongly affect where selected planning units were located across the Basin. This is because the distribution of the target taxa and ecosystems was much more important to the SCP process than management feasibility and water permanence. It is possible that the limited difference between cost constraints is because they are all acting similarly on the site prioritisations. For example, previous delivery of environmental water was used as an indication of 'feasibility' and this may affect water permanence as environmental water is often used to top-up permanent waterbodies to prevent drying.

5.2. Environmental water

Over three quarters of the ecosystem types within the managed floodplain had received environmental water or are proximal to a planning unit that have. Consequently, the distinct ecosystems and communities within these ANAE types may benefit from environmental water. Our result suggests a large proportion of the ecosystem diversity in the Basin is already serviced by environmental water (Table S1). There are some temporary ANAE ecosystem types on the managed floodplain that have not previously received environmental water and that could benefit from delivery and should be considered in future environmental water planning.

We found planning units that have previously received Commonwealth environmental water had higher productivity in terms of carbon sequestration on average than planning units that had not received Commonwealth environmental water. This suggests management is supporting higher productivity sites, i.e., more productive sites are more likely to receive environmental water. Thus, environmental water may be supporting greater species abundances, more diversity and higher trophic levels as these are known to be driven by high productivity (Waide et al., 1999).

Current actions to deliver environmental water are supporting the majority of the wetland species on the managed floodplain that we quantified. This is particularly true for the fish and molluscs as all were found in wetlands and lakes that have received environmental water. Many of the wetlands and lakes identified in this study as priority areas for biodiversity conservation, have also been identifed in previous studies. For example, using wetland birds as thier target taxa Bino et al., (2015) identified many of the same wetlands and lakes as the present study. However, watering actions targeted towards the protection of fish and birds may not protect other taxa and by comparison, frogs and many invertebrates including molluscs, crayfish, and Odonata on the managed floodplain are not currently in areas that have received Commonwealth environmental water. Despite, many of the Basin's invertebrate species including mussels and crayfish being considered keystone species and important components of a healthy riverine and terrestrial food-weds (Balzer et al., 2024; Noble et al., 2018; Sheldon et al., 2020).

Ecological and life-history trait information are often limited for many invertebrate taxa, and this is especially true for freshwater species (Bennett et al., 2018; Marsh et al., 2022). Further the environmental water needs of many of the Basin's invertebrate taxa are poorly understood (Marsh et al., 2022). For example, little is known about the ecology and life-cycle of the small range endemic crayfish species *Engaeus orientalis* which is found within the managed floodplain but has not been recorded in any area receiving environmental water. Further study is needed to determine if the species that are not currently receiving environmental water would benefit from its delivery now or in the future. Many species of wetland frogs, crayfish and Odonata are underrepresented on the managed floodplain and additional management levers other than environmental water may need to be considered for their conservation.

Our analysis is restricted to the areas where environmental water can currently be delivered, i.e. the managed floodplain. As part of the Murray-Darling Basin Plan (2012) reforms, the Murray-Darling Basin Authority developed the Constraints Management Strategy that aims at expanding the area environmental water can be delivered to in seven regions of the basin (Freak and Miller, 2024; Hart, 2016; Swirepik et al., 2016). In the future, the analytical approach taken in this study can be expanded to include new areas and habitats that environmental water can reach. The SCP approach could also be used to prioritise which habitats and potential refugia would benefit most from future environmental water deliveries and relaxation of constraints.

5.3. Limitations and knowledge gaps

Here we used individual wetlands and lakes mapped by the ANAE as our spatial planning units. Although there are many benefits to using the ANAE as a spatial framework, it also presents several challenges. Firstly, little is known about inundation patterns in individual wetlands (Linke et al., 2015), although recent improvements in mapping e.g. the wetland insights tool (WIT) are promising advancements in this area. Environmental water will likely flow between wetland complexes and further work is needed to understand and incorporate these patterns into conservation planning. This is especially necessary when accounting for re-use of environmental water in return-flows. Secondly, the resolution and spatial extent of this study may need to be tailored for implementation in management decisions. For example, management decisions may be made at the catchment scale or sub-basin scale, when only a proportion of the Basin is under water stress. In this instance, SCP could be conducted at multiple scales to ensure basin and catchment level diversity is captured by watering plans. Finally, the difference scales of the data sources may also affect the outcomes of prioritisation. For example, there are often multiple ANAE wetlands within each of the Level 15 Geofabric sub-catchments and ANAE wetlands and lakes can also span multiple sub-catchments. In future work, RDI and species distribution models at the sub-catchment scale could be refined to the wetland scale by enhancing the existing data via expert knowledge or switching to the latest version of the Geofabric which is at much higher resolution.

Future work should incorporate species trait data into the analysis (Gallagher et al., 2021). What habitats will act as refugia for a given organism will depend on their traits, especially those that relate to life history. For example, species with long generation times will take longer to respond to environmental water and may need multiple watering events or longer wettings to complete their life cycle. Furthermore, species with distinct life stages may require multiple refugia habitats to complete their life cycle (Wilbur, 1980). For example, some frog species are only water dependant for half the year and some for the other half (Rogers and Ralph, 2010). In these cases, the most logical way to easily adapt the prioritisation process would be to conduct seasonal SCP optimizations.

One of the major objectives of the Ramsar Convention on Wetlands (1971) is that the ecological character of wetlands of international importance be preserved. Many of the wetlands and lakes in the Basin are experiencing altered wetting regimes due to flow regulation. Therefore, the dryness anomaly used here may not reflect the natural drying patterns of the wetlands and lakes and therefore may not represent the original characteristics of the wetlands. In future analysis, refugia could be identified using climate-tracking and microclimate approaches. Climate tracking is used to project where current climate conditions will be spatially redistributed under future climate scenarios to identify in situ (where climate remain stable) and ex situ refugia (where suitable climatic conditions will be in the future) (Ashcroft, 2010). This approach can incorporate information on microclimate to identify environments where the local climate is decoupled from the regional climate due to topography (Ashcroft et al., 2012) or groundwater inputs (Davis et al., 2013). Further, climate change will increase both the frequency and intensity of drought, which may threaten the long-term health and functioning of wetlands and lakes (Hirabayashi et al., 2008). Once data availability improves future research should include plausible climate and hydrological change scenarios on flow and drought frequency. These advancements outlined above would allow us to project where important refugia will be in future climates.

5.4. Conclusion and recommendations

Here we piloted the use of readily available systematic conservation planning (SCP) tools and spatial data and applied it to the prioritisation of environmental water, with the aim of protecting refugia habitats for biodiversity and ecosystem function. The approach provides an objective and repeatable process that can be applied broadly at multiple scales to support annual environmental watering priorities. While there were many similarities in the areas included in the prioritisations under different cost scenarios and for different ecological assets, there were also marked differences between some ecological assets showing the importance of setting clear goals and objectives for the prioritisation. Further, more data on ecological traits is needed to identify spatiotemporal changes in refugia for target species. Given the treats facing the Murray-Darling Basin are affecting river basins all over the world, we believe SCP can provide an objective and repeatable approach for the delivery of environmental water or other protective management of refugial habitats.

CRediT authorship contribution statement

Joanne M. Bennett: Writing – original draft, Visualization, Validation, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Simon Linke:** Writing – review & editing, Methodology. **Shane Brooks:** Writing – review & editing, Formal analysis, Data curation. **Alex Bush:** Writing – review & editing, Methodology, Formal analysis. **James Hitchcock:** Writing – review & editing. **Carmel Pollino:** Project administration, Funding acquisition. **Ross M. Thompson:** Writing – review & editing, Project administration, Funding acquisition.

Data statement

The date and meta-data are on-line via the Australian governments data portal

https://data.gov.au/data/dataset/flow-mer-research-marxan-refuge-habitat

https://data.gov.au/data/dataset/9ce97171-1456-43dc-ab67-438155dee18e

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Joanne Bennett reports financial support was provided by University of Canberra. Joanne Bennett reports financial support was provided by Charles Sturt University. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

JMB, RT, SB, CP and JH were funded in part through the Basin Scale Flow-MER 1 program (2019–2025) jointly led by CSIRO and the University of Canberra and funded by the Australian Commonwealth Government through the Commonwealth Environmental Water Holder. The views represented in this paper do not represent those of the Australian Government. We would like to acknowledge the program-level support from Emily Barbour. Nikki Thurgate and Susan Cuddy (CSIRO), and the previous work carried out under the Long Term Intervention Monitoring Program led by the Murray Darling Freshwater Research Centre and La Trobe University (Ben Gawne and Nick Bond). RT is the Dame Carolyn Burns Research Professor in Freshwater Science at the University of Otago. JMB is currently supported by the Australian Research Council Australian Discovery Early Career Award (project number DE220100144) funded by the Australian Government.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.127184.

Data availability

Data and metadata are avaliable online. A link has been provided in the manuscript.

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