Ecological assets of the Victoria catchment to inform water resource assessments

A technical report from the CSIRO Victoria River Water Resource Assessment for the National Water Grid

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The Assessment was guided by two committees:

i. The Assessment's Governance Committee: CRC for Northern Australia/James Cook University; CSIRO; National Water Grid (Department of Climate Change, Energy, the Environment and Water); Northern Land Council; NT Department of Environment, Parks and Water Security; NT Department of Industry, Tourism and Trade; Office of Northern Australia; Queensland Department of Agriculture and Fisheries; Queensland Department of Regional Development, Manufacturing and Water

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Responsibility for the Assessment's content lies with CSIRO. The Assessment's committees did not have an opportunity to review the Assessment results or outputs prior to their release.

Asset content in this report was reviewed by Dr Adam Liedloff (CSIRO, Australia) and Tom Vanderbyl (Principal Badu Advisory Pty Ltd, Australia).

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Acknowledgement of Country

CSIRO acknowledges the Traditional Owners of the lands, seas and waters, of the area that we live and work on across Australia. We acknowledge their continuing connection to their culture and pay our respects to their elders past and present.

Photo

The Victoria River near the Victoria River Roadhouse. Source: CSIRO

Director’s foreword

Sustainable development and regional economic prosperity are priorities for the Australian and Northern Territory (NT) governments. However, more comprehensive information on land and water resources across northern Australia is required to complement local information held by Indigenous Peoples and other landholders.

Knowledge of the scale, nature, location and distribution of likely environmental, social, cultural and economic opportunities and the risks of any proposed developments is critical to sustainable development. Especially where resource use is contested, this knowledge informs the consultation and planning that underpin the resource security required to unlock investment, while at the same time protecting the environment and cultural values.

In 2021, the Australian Government commissioned CSIRO to complete the Victoria River Water Resource Assessment. In response, CSIRO accessed expertise and collaborations from across Australia to generate data and provide insight to support consideration of the use of land and water resources in the Victoria catchment. The Assessment focuses mainly on the potential for agricultural development, and the opportunities and constraints that development could experience. It also considers climate change impacts and a range of future development pathways without being prescriptive of what they might be. The detailed information provided on land and water resources, their potential uses and the consequences of those uses are carefully designed to be relevant to a wide range of regional-scale planning considerations by Indigenous Peoples, landholders, citizens, investors, local government, and the Australian and NT governments. By fostering shared understanding of the opportunities and the risks among this wide array of stakeholders and decision makers, better informed conversations about future options will be possible.

Importantly, the Assessment does not recommend one development over another, nor assume any particular development pathway, nor even assume that water resource development will occur. It provides a range of possibilities and the information required to interpret them (including risks that may attend any opportunities), consistent with regional values and aspirations.

All data and reports produced by the Assessment will be publicly available.

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Project Director

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Shortened forms

SHORT FORM	FULL FORM
AHGF	Australian Hydrological Geospatial Fabric
ALA	Atlas of Living Australia
CAMBA	China–Australia Migratory Bird Agreement
CI	confidence interval
CL	carapace length
DIWA	Directory of Important Wetlands in Australia
EPBC Act	*Environment Protection and Biodiversity Conservation Act 1999*
GDE	groundwater-dependent ecosystem
IBA	Important Bird and Biodiversity Area (BirdLife International)
IUCN	International Union for Conservation of Nature
JAMBA	Japan–Australia Migratory Bird Agreement
MDB	Murray–Darling Basin
MICE	Models of Intermediate Complexity for Ecosystem assessment
MODIS	Moderate Resolution Imaging Spectroradiometer
NAWRA	Northern Australia Water Resource Assessment
NPF	Northern Prawn Fishery
NVIS	National Vegetation Information System
OUT	operational taxonomic unit
PCR	polymerase chain reaction
ROKAMBA	Republic of Korea–Australia Migratory Bird Agreement

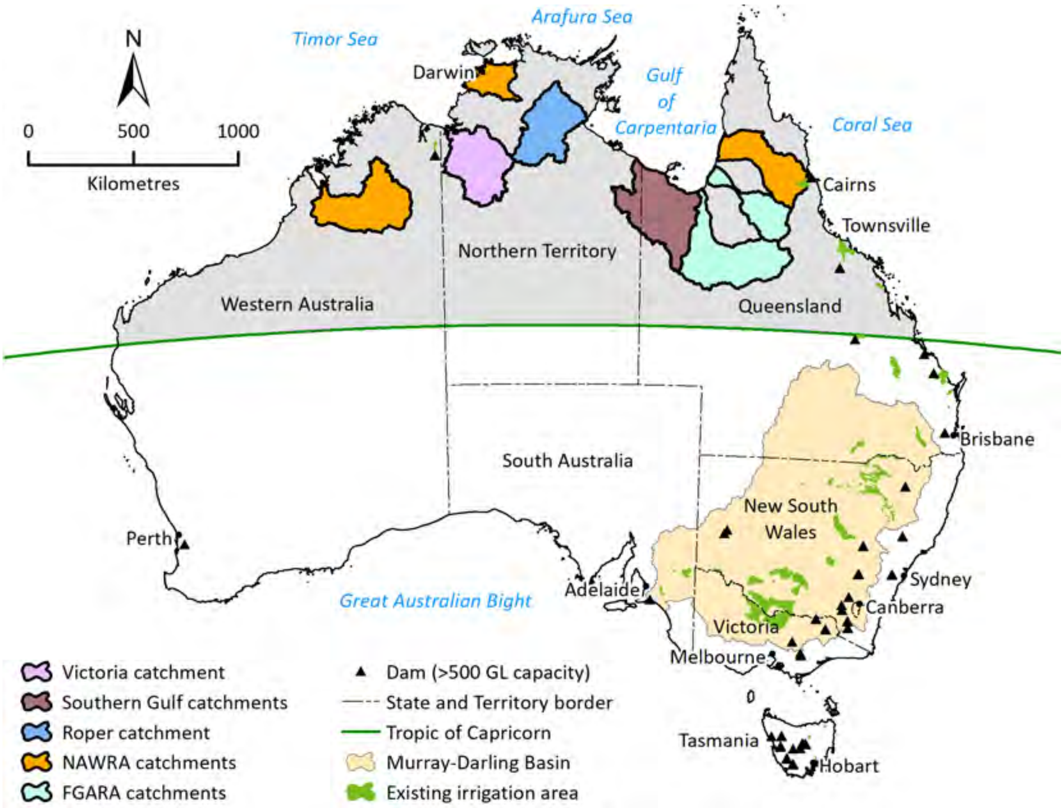
Units

UNIT	DESCRIPTION
cm	centimetre
ha	hectare
kg	kilogram
km	kilometre (1000 metres)
m	metre
mm	millimetre
mS	milliSiemens
ppt	parts per thousand

Preface

Sustainable development and regional economic prosperity are priorities for the Australian and NT governments and science can play its role. Acknowledging the need for continued research, the NT Government (2023) announced a Territory Water Plan priority action to accelerate the existing water science program 'to support best practice water resource management and sustainable development.'

Governments are actively seeking to diversify regional economies, considering a range of factors. For very remote areas like the Victoria catchment (Preface Figure 1-1), the land, water and other environmental resources or assets will be key in determining how sustainable regional development might occur. Primary questions in any consideration of sustainable regional development relate to the nature and the scale of opportunities, and their risks.



Preface Figure 1-1 Map of Australia showing Assessment area (Victoria catchment and other recent CSIRO Assessments

FGARA = Flinders and Gilbert Agricultural Resource Assessment; NAWRA = Northern Australia Water Resource Assessment.

How people perceive those risks is critical, especially in the context of areas such as the Victoria catchment, where approximately 75% of the population is Indigenous (compared to 3.2% for Australia as a whole) and where many Indigenous Peoples still live on the same lands they have inhabited for tens of thousands of years. About 31% of the Victoria catchment is owned by Indigenous Peoples as inalienable freehold.

Access to reliable information about resources enables informed discussion and good decision making. Such information includes the amount and type of a resource or asset, where it is found (including in relation to complementary resources), what commercial uses it might have, how the resource changes within a year and across years, the underlying socio-economic context and the possible impacts of development.

Most of northern Australia's land and water resources have not been mapped in sufficient detail to provide the level of information required for reliable resource allocation, to mitigate investment or environmental risks, or to build policy settings that can support good judgments. The Victoria River Water Resource Assessment aims to partly address this gap by providing data to better inform decisions on private investment and government expenditure, to account for intersections between existing and potential resource users, and to ensure that net development benefits are maximised.

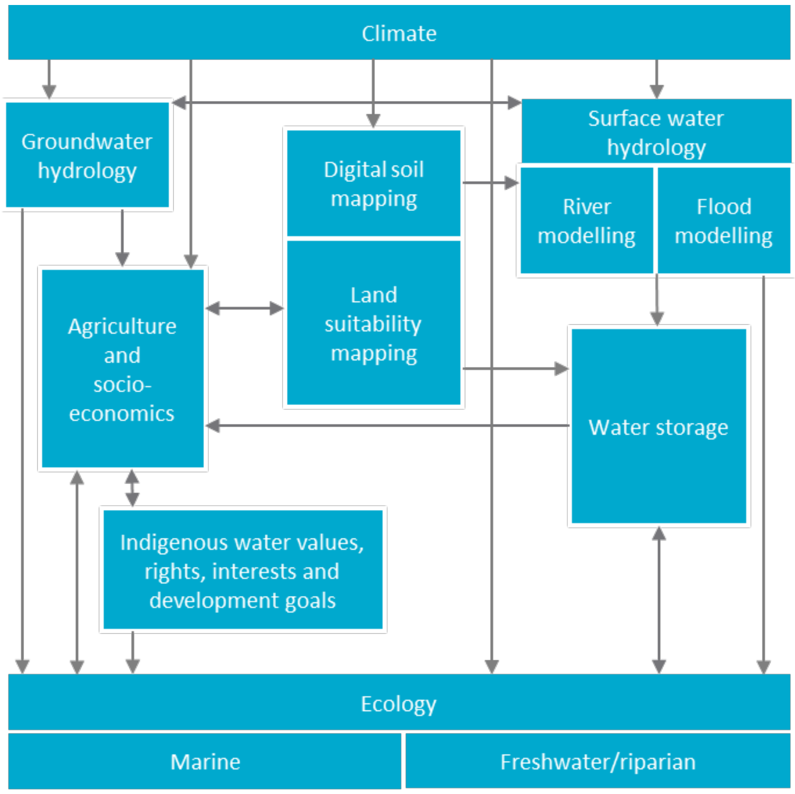
The Assessment differs somewhat from many resource assessments in that it considers a wide range of resources or assets, rather than being a single mapping exercise of, say, soils. It provides a lot of contextual information about the socio-economic profile of the catchment, and the economic possibilities and environmental impacts of development. Further, it considers many of the different resource and asset types in an integrated way, rather than separately. The Assessment has agricultural developments as its primary focus, but it also considers opportunities for and intersections between other types of water-dependent development.

The Assessment was designed to inform consideration of development, not to enable any particular development to occur. The outcome of no change in land use or water resource development is also valid. As such, the Assessment informs – but does not seek to replace – existing planning, regulatory or approval processes. Importantly, the Assessment does not assume a given policy or regulatory environment. Policy and regulations can change, so this flexibility enables the results to be applied to the widest range of uses for the longest possible time frame.

It was not the intention of – and nor was it possible for – the Assessment to generate new information on all topics related to water and irrigation development in northern Australia. Topics not directly examined in the Assessment are discussed with reference to and in the context of the existing literature.

CSIRO has strong organisational commitments to reconciliation with Australia's Indigenous Peoples and to conducting ethical research with the free, prior and informed consent of human participants. The Assessment consulted with Indigenous representative organisations and Traditional Owner groups from the catchment to aid their understanding and potential engagement with its fieldwork requirements. The Assessment conducted significant fieldwork in the catchment, including with Traditional Owners through the activity focused on Indigenous values, rights, interests and development goals. CSIRO created new scientific knowledge about the catchment through direct fieldwork, by synthesising new material from existing information, and by remotely sensed data and numerical modelling.

Functionally, the Assessment adopted an activities-based approach (reflected in the content and structure of the outputs and products), comprising activity groups, each contributing its part to create a cohesive picture of regional development opportunities, costs and benefits, but also risks. Preface Figure 1-2 illustrates the high-level links between the activities and the general flow of information in the Assessment.



Preface Figure 1-2 Schematic of the high-level linkages between the eight activity groups and the general flow of information in the Assessment

Assessment reporting structure

Development opportunities and their impacts are frequently highly interdependent and, consequently, so is the research undertaken through this Assessment. While each report may be read as a stand-alone document, the suite of reports for each Assessment most reliably informs discussion and decisions concerning regional development when read as a whole.

The Assessment has produced a series of cascading reports and information products:

- Technical reports present scientific work with sufficient detail for technical and scientific experts to reproduce the work. Each of the activities (Preface Figure 1-2) has one or more corresponding technical reports.
- A catchment report, which synthesises key material from the technical reports, providing well-informed (but not necessarily scientifically trained) users with the information required to inform decisions about the opportunities, costs and benefits, but also risks associated with irrigated agriculture and other development options.
- A summary report provides a shorter summary and narrative for a general public audience in plain English.
- A summary fact sheet provides key findings for a general public audience in the shortest possible format.

The Assessment has also developed online information products to enable users to better access information that is not readily available in print format. All of these reports, information tools and data products are available online at https://www.csiro.au/victoriariver. The webpages give users access to a communications suite including fact sheets, multimedia content, FAQs, reports and links to related sites, particularly about other research in northern Australia.

Executive summary

This activity (the ecology activity) seeks to determine the relative risks between different water resource development scenarios in the Victoria catchment using a set of prioritised water-dependent assets. Environmental assets are selected from freshwater, marine and terrestrial habitats.

The key questions that this activity seeks to address in the catchment include:

- What is the main environmental context of the catchment that could influence water resource development?
- What are the key environmental drivers and stressors that are currently occurring or likely to occur in the catchment (including key supporting and threatening processes such as invasive species, water quality and habitat changes)?
- What are the known linkages between flow and ecology?
- What are the key ecological trade-offs between different water resource developments considering impacts from potential changes in flow on species and habitats?

This report provides a synthesis of the prioritised ecology assets occurring in the Assessment catchment including developing asset knowledge bases, conceptual relationships, and evidence narratives, including the flow–ecology relationships, and considering their context and application in the Assessment catchment.

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Part I Ecology of the Victoria catchment

Development of water resources can lead to a range of impacts on the environment including changes in flow regimes, land use impacts and changes to connectivity caused by building instream structures. The rivers, floodplains and coastal regions of northern Australia are highly diverse and have significant conservation, cultural and economic values. To understand the potential risks to the natural environment associated with water resource development, this investigation (the ecology activity) takes an ecological asset approach. This involves identifying and prioritising assets; developing asset knowledge bases, conceptual relationships and evidence narratives, including flow–ecology relationships; and considering the context and distribution of the assets in the catchment of the Victoria River. The investigation to understand impacts draws upon the knowledge base of these assets and aspects of ecosystem function to model outcomes of water resource development and climate change scenarios. This technical report presents this knowledge base for the prioritised freshwater-dependent ecological assets across freshwater, marine and terrestrial habitats in the Victoria catchment.

1 Introduction

1.1 Ecology and water resource development

Development of water resources to support agriculture or aquaculture including water harvest from river flows, instream engineered dams or groundwater development can lead to a range of impacts on the environment, including changes in flow regimes, land use related habitat loss and changes to connectivity caused by building instream structures. The flow regimes of rivers are a primary driver of riverine, wetland, floodplain and near-shore coastal ecology (Bunn and Arthington, 2002; Junk et al., 1989; Poff and Zimmerman, 2010). Water resource development alters flow regimes and leads to potentially significant changes in important flow attributes, such as the magnitude, timing, duration and rate of change of flow events upon which flora and fauna of the ecosystem depend. Also, changes in the frequency and duration of wet or dry spells and any modification of water quality (including changes to temperature regimes or sediment discharges) can affect species and their habitat. Additional impacts can result from instream barriers, land use change and a range of other threatening processes, either directly, indirectly or in synergy with development. The result is potential ecological changes and consequences for the biota, habitats and ecosystem processes of a catchment (Poff et al., 1997).

Freshwater and estuarine systems in northern Australia contain a high level of diversity, with many unique and significant species and habitats. The catchments of northern Australia support at least 170 fish species, 150 waterbird species, 30 aquatic and semi-aquatic reptile species, 60 amphibian species and 100 macroinvertebrate families (van Dam et al., 2008). The freshwater and estuary habitats of northern Australia are critical in supporting productive fisheries, which increase production as freshwater inflow to estuaries increases (Aquatic Ecosystems Task Group, 2012). Catchment flows support high-value commercial and recreational marine fisheries, such as the Northern Prawn Fishery, as well as barramundi (*Lates calcarifer*), mud crab (*Scylla serrata*) and a suite of other species important to commercial, recreational and/or Indigenous fisheries. Species and habitats of conservation significance also depend on discharge of water and nutrients and include migratory waterbirds, sea turtles and a variety of sharks and rays, including sawfish (*Pristis* spp.) and river sharks (*Glyphis* spp.).

This activity (the ecology activity) seeks to determine the relative risks of alternative water resource development scenarios in the Victoria catchment. The key questions that this activity seeks to address include:

- What is the main environmental context of the catchment that could influence water resource development?
- What are the key environmental drivers and stressors that are currently occurring or likely to occur in the catchment (including key supporting and threatening processes, such as invasive species, water quality and habitat changes)?
- What are the known linkages between flow and ecology?
- What are the key ecological trade-offs between different water resource developments, considering impacts from potential changes in flow on species and habitats?

To understand the potential risks to the natural environment associated with water resource development, the ecology activity combines an ecological asset approach with other approaches that explore system processes such as lateral and longitudinal connectivity, inundation, habitat provision and end-of-system discharge. The ecology work builds upon and adapts the methods used in the ecology synthesis and assessment components of the Northern Australia Water Resource Assessment (NAWRA) (Pollino et al., 2018a; Pollino et al., 2018b). This involves undertaking a review and prioritising assets; developing asset knowledge bases, conceptual relationships and evidence narratives, including flow–ecology relationships; and considering the context and distribution of the assets in the catchment of the Victoria River.

To understand the possible changes and impacts resulting from different development and climate change scenarios the ecology activity uses a range of modelling methods for the prioritised ecology assets (methods and results provided in the Asset Analysis reporting). Ecology assets include species, groups of species, habitats and their processes that are freshwater dependent and significant within the Victoria catchment and for which there is sufficient understanding. The use of modelling methods for specific assets depends on the relationships between flow and ecological outcomes and if these relationships are sufficiently known and can be suitably supported by the knowledge base of the asset. Quantitative ecology modelling uses hydrology scenarios developed by the surface water hydrology activity as primary inputs. The ecological modelling operates at catchment scales and compares outcomes as relative differences between the scenarios and a baseline to identify where change occurs and by how much. The analysis enables the identification of assets that may be most sensitive to the type of changes in the different scenarios, and the scenarios that lead to the greatest ecological change. An overview of the ecology approach is provided in Section 1.3.

1.2 Water resource development and ecological changes

The importance of the natural flow regime for supporting environmental function has become increasingly well understood, as has the importance of rivers operating as systems, including the connection of floodplains via inundation, the distribution of refuges, and discharges into coastal regions. Globally, water resource development has a range of known impacts on ecological systems. The influence of each of these depends upon a range of factors, including catchment properties (e.g. physical, geographic and climate characteristics), the kind of development (e.g. dams, water harvesting, groundwater development), the source location or distribution of the development within the catchment, the magnitude and pattern of change, how any changes may be managed or mitigated, and the habitats and species that will be affected and their distribution.

Impacts associated with water resource development include the following, which are described below:

- flow regime change (Section 1.2.1)
- altered longitudinal and lateral connectivity (Section 1.2.2)
- habitat modification and loss (Section 1.2.3)
- increased invasive and non-native species (Section 1.2.4)
- synergistic and co-occurring processes both local and global (Section 1.2.5).

1.2.1 Flow regime change

Water resource development including water harvesting and creating instream structures for water retention can influence the timing, quality and quantity of water that is provided by catchment runoff into the river system. The natural flow regime (including the magnitude, duration, timing, frequency and pattern of flow events) is important in supporting a broad range of environmental processes upon which species and habitat condition depend (Lear et al., 2019; Poff et al., 1997). Flow conditions provide the physical habitat in streams and rivers which determines biotic use and composition and to which life-history strategies are adapted, and enables movement and migration between habitats and exchange of nutrients and materials (Bunn and Arthington, 2002; Jardine et al., 2015). In a river system, the natural periods of both low and high flow (including no-flow events) are important to support the natural function of habitats, their ecological processes and the shaping of biotic communities (King et al., 2015). Through the attenuation of flows, water resource development can lead to impacts significant distances downstream of the development, including into coastal and near-shore marine habitats (Broadley et al., 2020; Pollino et al., 2018a).

1.2.2 Altered longitudinal and lateral connectivity

River flow facilitates the exchange of biota, materials, nutrients and carbon along the river and into the coastal areas (longitudinal connectivity), as well as between the river and the floodplain (lateral connectivity) (Pettit et al., 2017; Warfe et al., 2011). Physical barriers such as weirs and dams, or a reduction in the magnitude of flows (and the duration or frequency) can affect longitudinal and lateral connectivity, changing the rate or timing of exchanges (Crook et al., 2015). These impacts can include changes in species' migration and movement patterns as well as altered erosion processes and discharges of nutrients into rivers and coastal waters (Brodie and Mitchell, 2005). Seasonal patterns and rates of connection and disconnection caused by flood pulses are important for providing seasonal habitat, enabling movement of biota into new habitats and their return from refuge habitats following larger river flows (Crook et al., 2020).

1.2.3 Habitat modification and loss

Water resource development can cause direct loss of habitat. For example, an artificially created lake inundates the habitat behind an impoundment resulting in the loss of the existing terrestrial and stream habitat. Agricultural development converts existing habitat to more intensive agriculture. Infrastructure including roads can fragment terrestrial habitat, while streams and canals can artificially connect aquatic habitats that had been historically separated.

1.2.4 Increased invasive and non-native species

Water resource development often homogenises flow and flow related habitats, for example, through changed patterns resulting from capture and release of flows or creation of impoundments for storage and regulation. It is recognised that invasive species are often at an advantage in such modified habitats (Bunn and Arthington, 2002). Modified landscapes, such as lakes or homogenised perennial streams that were previously ephemeral, can be a pathway for introduction and support the incidental, accidental or deliberate establishment of non-native species, including pest plants and fish (Bunn and Arthington, 2002; Close et al., 2012; Ebner et al., 2020). Increased human activity can increase the risk of invasive species being introduced.

1.2.5 Synergistic and co-occurring processes both local and global

Along with water resource development comes a range of other pressures and threats, including increases in fishing, vehicles, habitat fragmentation, pesticides, fertilisers and other chemicals, erosion, degradation due to increased stock pressure, changed fire regimes, climate change and other human disturbances, both direct and indirect. Some of these pressures are the result of changes in land use associated with or accompanying, water resource development, others may occur locally, regionally or globally and act synergistically with water resource development and agricultural development to increase the risk to species and their habitats (Craig et al., 2017; Pettit et al., 2012).

1.3 Ecology asset-based approach to modelling and assessment

The goal of the ecology activity is to understand the potential impacts of water resource development on ecological systems. This is achieved by modelling a set of ecology assets, including species, habitats and catchment ecosystem functions. Ecology assets in northern Australia depend upon freshwater flows to support their persistence or function. Assets are spread across freshwater, marine and terrestrial habitats (including terrestrial habitats dependent on groundwater or flood flow and inundation). The ecological assets have different distributions within the catchment, have different flow associations and needs, and are likely to have different trajectories of change when exposed to a potential range of threatening processes. The ecology activity uses these assets in a range of models to infer what impacts may occur and where within the catchment as a result of different water resource development and climate change scenarios.

The ecology activity is built upon four main components of work, which are described below:

- identifying and prioritising assets (Section 1.3.1)
- reviewing, conceptual modelling and developing evidence narratives (Section 1.3.2)
- mapping the location and distribution of the assets (Section 1.3.3)
- understanding flow–ecology relationships and quantitative modelling (Section 1.3.4).

1.3.1 Identifying and prioritising assets

For the purpose of the ecology activity, assets can be considered either partially or fully freshwater dependent, including terrestrial or marine assets dependent upon freshwater flows (or services provided by freshwater flows). To identify assets for the ecology analysis, species, species groups and habitats have been reviewed and prioritised for the Victoria catchment. Assets can include:

- species − individual species (such as barramundi in Section 3.1.1)
- taxonomic groups − groups of species that are closely related (such as grunters in Section 3.1.3)
- functional groups − groups of often unrelated species that may occupy similar niches, use similar habitat, have other attributes or requirements, and that are likely to respond to change in a similar way (such as colonial and semi-colonial nesting wading waterbirds in Section 3.2.1)

- habitats − important habitats include geographical areas identified as sharing similar characteristics (such as position on the floodplain or channel, water retention or shedding properties) or other structural features that may make it important for the catchment ecology and support biota within and around the catchment. Habitats include floodplain wetlands (Section 3.4.1) and mangroves (Section 3.4.4). Habitats are important for supporting species or communities and may include, but are not limited to, identified or listed locations such as national parks or Directory of Important Wetlands in Australia (DIWA) sites.

Freshwater-dependent assets were considered for the ecology assessment if they meet any of the following:

- a species or community that is listed as Threatened, Vulnerable or Endangered (EPBC or State/Territory listing)
- a habitat, species or community that is formally recognised in conservation agreements
- habitat that provides vital, near-natural, rare or unique habitat for water-dependent flora and fauna
- supporting important or notable biodiversity of water-dependent flora and fauna
- providing recreational, commercial or cultural value.

From the full range of potential assets identified in the Victoria catchment, the process for selecting priority assets considered if they are:

- distinctive – so able to create an association between flow and outcomes of change for assets, and to consider a broad range of water requirements across the variety of different assets
- representative – so able to consider flow requirements for other biota and ecological processes that are not explicitly modelled
- describable – having sufficient available peer-reviewed evidence to identify and describe relationships with flow
- significant – considering ecological, conservation, cultural and recreational importance, and relevance to the Assessment catchment.

The prioritised ecological assets described in this report are listed in Table 1-1. These assets are used in the ecology analysis to assess the relative risks associated with different water resource development scenarios considering the type, magnitude and location of change.

Table 1-1 Freshwater, marine and terrestrial ecological assets with freshwater flow dependences

ASSET GROUP	ASSET	SYSTEMS	PAGE
Fish, sharks and rays	Barramundi (*Lates calcarifer*)	Freshwater and marine	21
	Catfish (order Siluriformes)	Freshwater	28
	Grunters (family Terapontidae)	Freshwater	34
	Mullet (family Mugilidae)	Freshwater and marine	40
	River sharks (*Glyphis* spp.)	Freshwater and marine	45
	Sawfishes (*Pristis* and *Anoxypristis* spp.)	Freshwater and marine	50
	Threadfin (*Polydactylus macrochir*)	Marine	56
Waterbirds	Colonial and semi-colonial nesting wading waterbirds	Freshwater	62
	Cryptic wading waterbirds	Freshwater	71
	Shorebirds	Freshwater and marine	77
	Swimming, grazing and diving waterbirds	Freshwater	89
Turtles, prawns and other species	Banana prawns (*Penaeus indicus*)	Marine	101
	Freshwater turtles (family Chelidae)	Freshwater	108
	Mud crabs (*Scylla serrata*)	Marine	116
Flow-dependent habitats	Floodplain wetlands	Freshwater	122
	Groundwater-dependent ecosystems	Freshwater and terrestrial	127
	Inchannel waterholes	Freshwater	143
	Mangroves	Marine	148
	Saltpans and salt flats	Marine	153
	Surface-water-dependent vegetation	Freshwater and terrestrial	157

1.3.2 Reviewing, conceptual modelling and developing evidence narratives

Literature and data were reviewed for each asset. Information collated includes catchment-specific data and knowledge, important water and flow requirements, habitat use and distribution, and information required to support an understanding of the asset's response to potential environmental change, considering aspects such as life history, flow triggers, movement, refuge, foraging and productivity. A conceptual model for each asset was developed to represent and summarise this ecological understanding and an evidence narrative used to support and communicate the flow–ecology relationships and pathways to change for each asset based upon available data and literature. The asset conceptual models provide a simplified visual summary and outline an evidence-based hypothesis for the potential impacts under water resource development and climate change scenarios.

The conceptual models are box and arrow models with a standard structure that represents links between the threats, drivers, effects and outcomes.

These key terms are defined as follows:

- Threat is an action or activity that has the capacity to adversely affect an ecological asset and its value (Hart et al., 2005).

- Driver (ecological driver) is the mechanism, process or change by which a threat affects an asset.
- Effect is the direct change in, or response of, the asset that has occurred as a consequence of the driver.
- Outcome is the overall observable or measurable impact on the asset or its function within the catchment (tangible or otherwise).

The conceptual models explore relationships between key potential threats (including water resource development, land use and climate change) and the effects and outcomes of these threating processes from the perspective of each asset, including loss of biodiversity or habitat quality.

1.3.3 Mapping the location and distribution of the assets

The location or distribution of assets has been mapped across the Victoria catchment to understand the important locations and/or occurrences of the assets across the catchment and the near-shore coastal region. Water resource development scenarios consider a range of development options and pathways within each catchment, with the flow downstream of each development being affected. Assets located downstream of these scenario water resource development sites will be exposed to changes in flow with different scenarios resulting in different changes. The level of flow change and impact will depend on the asset's location relative to the development. Assets that have habitat that spans barriers may also be affected by changes in connectivity due to the placement of new instream barriers, or changes in flow occurring over existing barriers.

A range of data sources, including Atlas of Living Australia (www.ala.org.au), government department databases and fisheries catch records, were used to develop maps and spatial relationships across the different parts of the Assessment catchment. For species with suitable data, species distribution models have been created to extrapolate potential suitable habitat within the catchment based upon environmental relationships established across northern Australia, thereby drawing upon a larger number of data points. The species distribution modelling uses the observed locations of species across northern Australia to create correlations with environmental covariates that are then used to predict suitable habitat associations within the Victoria catchment (the predictor variables are shown in Appendix A).

1.3.4 Understanding flow–ecology relationships and modelling

The analysis approach undertaken in the ecology activity uses a combination of modelling methods with the choice of method(s) used for each asset depending upon the ability to support modelling given the strength of the knowledge base, the asset data and the types of relationships important for each asset. Modelling requires understanding relationships between flow, ecological responses and the potential outcomes of changes. Different models are used to represent and incorporate varying processes (including changes in flow, inundation and connectivity) that range in importance for different assets. The primary inputs to analysis are daily hydrology data generated with river system models (flow discharge and quantitative properties of the hydrograph) or hydrodynamic models (depth, velocity, inundation) that can be used to quantify the relative differences between scenarios and a baseline over the same modelled period (Hughes et al., 2024). A summary of some of the ecology modelling used in the ecology assessment is provided below, and further details are provided in the companion report an asset analysis (Stratford et al., 2024).

Flow requirements

A common base to the analysis is the flow requirements method which is used for all assets. This analysis identifies the specific components of the hydrograph that are important to each asset (quantified using hydrometrics: statistical properties of the long-term flow regime). The specific set of flow metrics identified for each of the assets are important for ecological function such as supporting life-history requirements, movement or provision of important habitat (for example, high flows may be required to support migration upstream or onto the floodplain for fish species thereby enhancing habitat quality or availability resulting to improved condition and higher abundances). The flow requirements assessment calculates the change in these asset-specific hydrometrics occurring between the model scenarios and the range of conditions exhibited in the baseline to enable relative comparison between locations, assets and scenarios. Depending upon the scenario, and whether the development is a single point source or distributed, changes in flow could accumulate or diminish as flows attenuate through the catchment. Results indicate sensitivities to the types of change in the hydrograph arising because assets have different flow associations and needs and occupy different locations within the catchment and because different water resource development scenarios manifest different changes in hydrology.

Connectivity assessment

The connectivity assessment uses hydrodynamic modelling to develop a daily time series of inundation extents or depths for a range of scenarios. For these scenarios, across a sample of flood events, the pattern and extent of inundation can quantify the connectivity of assets (e.g. floodplain wetlands) to the main river channel via connection across the floodplain or via flood runners (latitudinal connectivity) or across instream barriers that may limit movement along the river channel during periods of low flow (longitudinal connectivity). Differences in the extent and/or duration of connections to these assets is quantified between the scenarios.

Hydrodynamic habitat suitability

Hydrodynamic habitat suitability modelling uses depth, velocity and inundation extent outputs from hydrodynamic models to map and quantify the occurrence of flow habitat that would be suitable or preferred for species across different flood events. The method uses species' (or species groups') specific habitat preferences informed by literature and/or data to provide mechanistic links between hydraulic variables from hydrodynamics modelling. The form of these relationships can be used for a range of biota, such as fish and waterbirds, for which depth and velocity are important determinants of habitat preferences or associations. For example, habitat preference can be informed by the results of tracking studies that position species' use of inchannel and floodplain habitat and relate this to the experienced hydraulic properties of the location. Changes in the hydrodynamics between flood events and between scenarios may reduce or enhance the availability of suitable or preferred habitat within the catchment with an effect on the availability of resources for the assets.

Models of Intermediate Complexity for Ecosystem assessment

Models of Intermediate Complexity for Ecosystem assessment (MICE) are methods for simultaneously assessing the status of both fisheries and other non-targeted species, including those of high conservation concern, and evaluating the trade-offs among management plans aimed at addressing conflicting objectives. They are dynamic, spatially resolved models of intermediate complexity that draw on quantitative and statistical methods of stock assessment approaches and extend this to a representation of stressors and outcomes in an ecosystem.

2 Ecology of the Victoria catchment

2.1 Ecology of the Victoria catchment

2.1.1 Victoria catchment and its environmental values

The comparatively intact landscapes of the Victoria catchment are important for the ecosystem services they provide, including recreational activities, tourism, fisheries (Indigenous, recreational and commercial), military training and agricultural production (notably cattle grazing on native pastures). The catchment also holds important ecological and environmental values. The Victoria River is a large perennial (i.e. maintaining flow all year) river originating near Judbarra / Gregory National Park. At over 500 km in length, it is one of the longest perennial rivers in the Northern Territory (NT). The catchment area of 82,400 km^2 makes it one of the largest ocean-flowing catchments in the NT with flows that enter the south-eastern edge of the Joseph Bonaparte Gulf. The catchment and the surrounding marine environment contain a rich diversity of important ecological assets, including species, ecological communities, habitats, and ecological processes and functions (see the conceptualised summary in Figure 2-1). The ecology of the Victoria catchment is maintained by the river's flow regime, shaped by the region's wet-dry climate and the catchment's complex geomorphology and topography, and driven by patterns of seasonal rainfall, evapotranspiration and groundwater discharge.

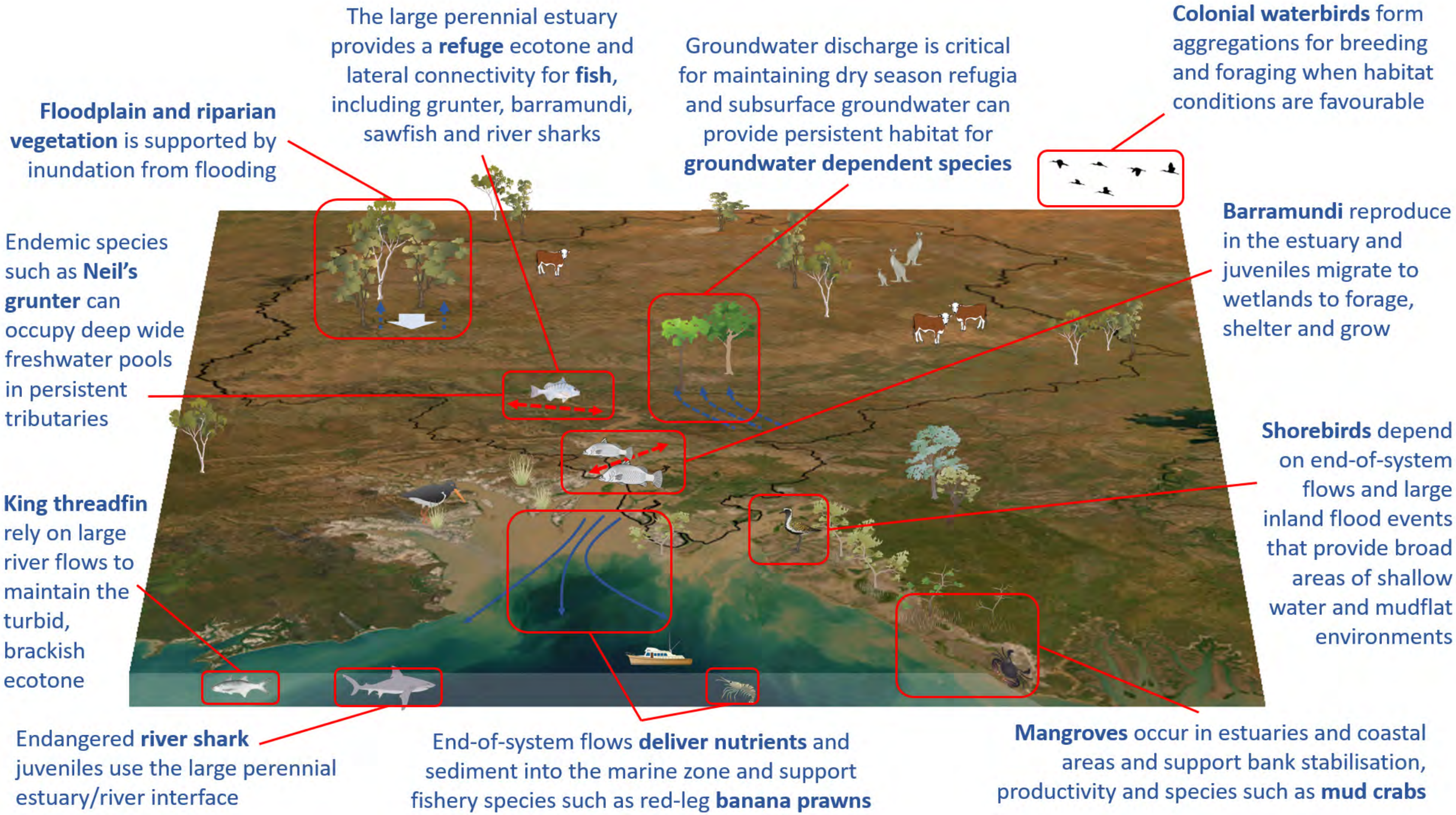

Figure 2-1 Conceptual diagram of selected ecological assets of the Victoria catchment. Ecological assets include species of significance, species groups and important habitats

See Table 1-1 for a complete list of the freshwater, marine and terrestrial ecological assets considered in the Victoria catchment.

Biota icons for the Victoria catchment adapted from the Integration and Application Network (2023)

Much of the natural environment of the Victoria catchment consists of rolling plains, mesas, escarpments and plateaux with savanna woodlands and various grasslands including spinifex (Kirby and Faulks, 2004). The wet-dry tropical climate results in highly seasonal river flow with 90% of rainfall occurring between November and March (Kirby and Faulks, 2004). As typical for the region, the dynamic occurring between wet and dry seasons provides both challenges and opportunities for biota (Warfe et al., 2011). During the dry season, river flows are reduced with many of the streams in the catchment receding to isolated pools. However, some of the larger tributaries in the catchment are perennial, including sections of Wickham River (upstream of Humbert River junction) and the Angalarri River (Midgley, 1981). In parts of the Victoria catchment, the persistence of water during the dry season is supported by discharge from groundwater-fed springs that persist during most dry seasons (Bureau of Meteorology, 2017); these habitats support aquatic life and fringing vegetation. In the dry season, the streams and waterholes that persist provide critical refuge habitat for many species both aquatic and terrestrial.

During the wet season, many low-lying parts of the catchment flood, inundating floodplains, connecting wetlands to the river channel and driving a productivity boom. While the extent of floodplain wetlands is comparatively moderate compared to many other tropical catchments, flooding can be more evident in the lower parts of the catchment, including the floodplains, wetlands and intertidal flats of the estuary and around the junction of the Victoria with both the West Baines and Angalarri rivers due to catchment topography . Annual flooding delivers extensive sediment-rich discharges into the marine waters of the southern Joseph Bonaparte Gulf with sediment plumes that can extend large distances into the gulf.

2.1.2 Protected, listed and significant areas of the Victoria catchment

The protected areas located in the Victoria catchment include one gazetted national park (Judbarra/Gregory), a proposed extension to an existing national park (Keep River), two marine national parks, two Indigenous Protected Areas and two Directory of Important Wetlands in Australia (DIWA) sites (Figure 2-2). Judbarra / Gregory National Park is the second largest national park in the NT covering approximately 1,300,000 ha (Australian Government, 2022b) and is popular for tourism, showcasing gorges, escarpment country and sandstone formations, boab trees and fishing. Once fully gazetted, the Keep River National Park including the proposed extension from the neighbouring Keep River catchment into the Victoria catchment will cover a total area of approximately 272,000 ha with the goal to have the additional 215,000 ha gazetted by 2026 (Australian Government, 2022b; Department of Environment Parks and Water Security, 2023). The Wardaman Indigenous Protected Area extends across the northern Victoria catchment and beyond and covers a total area of approximately 225,000 ha (Australian Government, 2022b), while the Northern Tanami Indigenous Protected Area abuts the southern boundary of the Victoria catchment with only a minimal portion within the Victoria catchment. The Joseph Bonaparte Gulf Marine Park is a Commonwealth marine park of 15 to 100 m depth and approximately 860,000 ha (Australian Government, 2022a). This marine park straddles the offshore portion of the Victoria catchment marine region and has tides up to seven metres and is home to the Australian snubfin dolphin (*Orcaella heinsohni*) (Department of Agriculture Water and the Environment, 2021; Parks Australia, 2023). The eastern edge of the North Kimberly Marine Park (WA) is adjacent to the

Joseph Boneparte Gulf Marine Park and follows the Western Australia coastline to the WA/NT border.

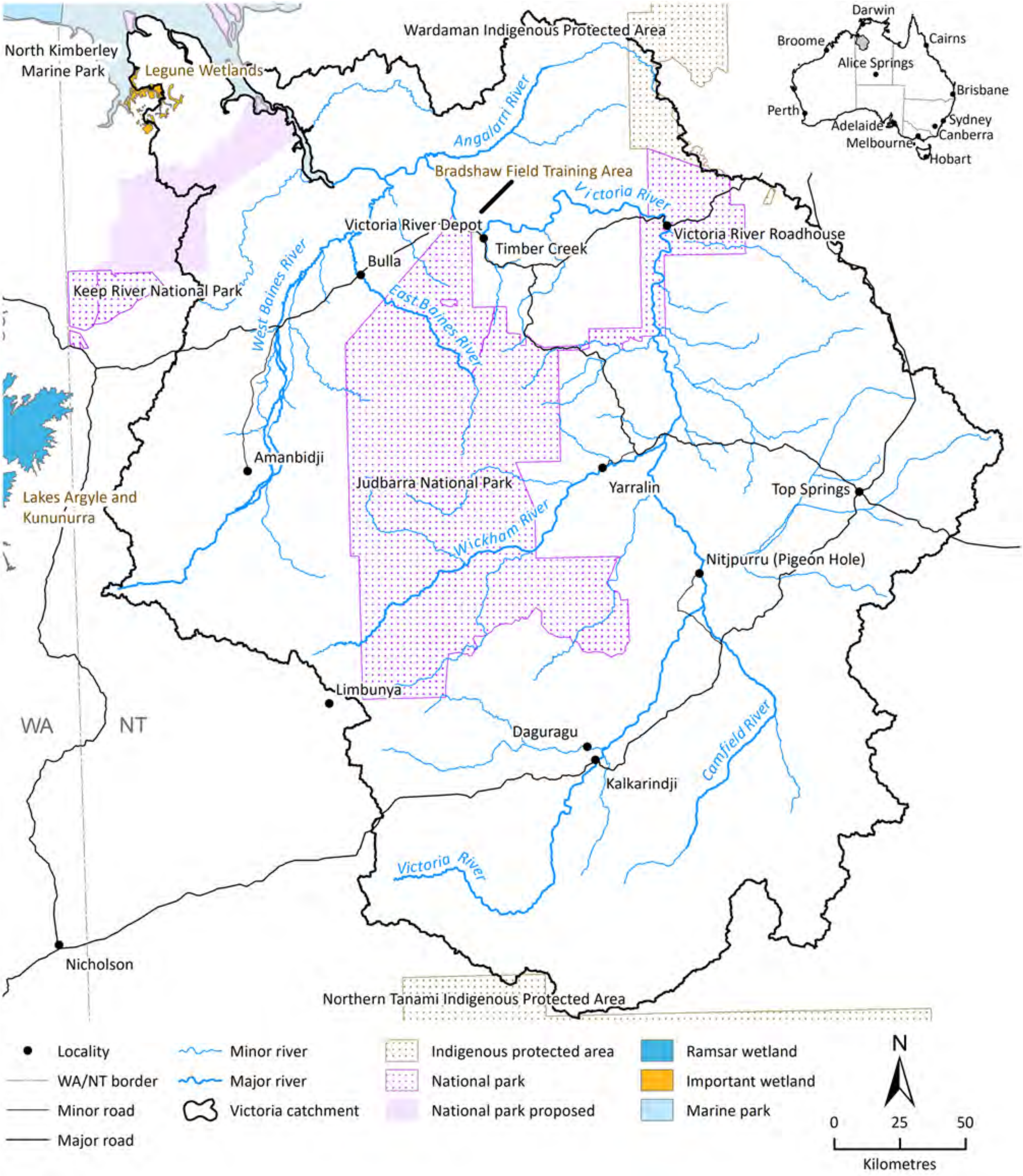

Figure 2-2 Location of protected areas and important wetlands within the Victoria catchment assessment area

Protected areas include management areas mainly for conservation through management intervention as defined by the International Union for Conservation of Nature (IUCN).

Data sources: Department of Agriculture, Water and the Environment (2020a; 2020b); Department of the Environment and Energy (2010), Department of Climate Change, Energy, the Environment and Water (2024)

The two DIWA sites are the Bradshaw Field Training Area and the Legune Wetlands (Figure 2-2). The Bradshaw Field Training Area is a military training area between the Victoria River and the Angalarri River in the northern part of the catchment. The Legune Wetlands span the border of the Victoria River and Keep River catchments adjacent to the upper estuary and salt flats of the Keep River. These two DIWA wetlands highlight the diversity of aquatic habitats that can be found within the Victoria catchment. The Victoria catchment contains no Ramsar sites but the neighbouring Ord catchment contains two: the Lakes Argyle and Kununurra Wetlands and the Ord River Floodplain.

The Bradshaw Field Training Area DIWA site lies north of the Victoria River near Timber Creek. It is bound by the Fitzmaurice River to the north and the Victoria River to the south. The site includes two wetland complexes coving a total of approximately 871,000 ha within the Victoria Bonaparte biogeographic region (Department of Agriculture Water and the Environment, 2023a). Large areas of the wetlands are inundated each wet season by floods from both the Fitzmaurice and Victoria rivers, with flooding enhanced during coincidence with high tides. Some areas of the site retain permanent water during the dry season (Department of Agriculture Water and the Environment, 2023a). The Bradshaw field training wetland site has very high wilderness value and includes areas of Monsoon Vine Forest; it forms an important component of the conservation network within the Victoria catchment (Department of Agriculture Water and the Environment, 2023a; Lands Planning and Environment, 1998). The Bradshaw field training Australian Heritage area has a species richness of mammals, reptiles and frogs that is significant on the national level and is considered a stronghold for species that have recorded declines in other locations including the Gouldian finch (*Erythrura gouldiae*), the northern quoll (*Dasuurus hallucatus*) and the pale field rat (*Rattus tunneyi*). Over 850 flora species and 375 fauna species comprising 22 frog, 77 reptile, 212 bird, 50 mammal and 26 fish species are known to occur here (Department of Climate Change Energy the Environment and Water, undated).

The Legune Wetlands straddles the Keep and Victoria river catchments with inflows from surface water from local creeks and in wet years from major floods in the Keep River providing some additional inflows (Department of Agriculture Water and the Environment, 2023b). The wetlands include areas identified as an Important Bird and Biodiversity Area (IBA) by Birdlife International with surveys recording more than 15,000 individuals from over 45 species including magpie goose (*Anseranas semipalmata*), brolga (*Antigone rubindicus*) and red-capped plover (*Charadrius ruficapillus*) (BirdLife International, 2023; Department of Agriculture Water and the Environment, 2023b). Habitats of importance include seasonal marshes and swamps, freshwater mangroves, mudflats and salt flats and provides important dry season habitat for waterbirds (BirdLife International, 2023; Department of Agriculture Water and the Environment, 2023b).

2.1.3 Important habitat types and values of the Victoria catchment

The freshwater sections of the Victoria catchment include diverse habitats such as perennial and intermittent rivers, anabranches, wetlands, floodplains and groundwater-dependent ecosystems (GDEs). The diversity and complexity of habitats, and the connections between habitats within a catchment, are vital for providing the range of habitats needed to support both aquatic and terrestrial biota (Schofield et al., 2018).

In the wet season, flooding connects rivers to floodplains. This water exchange means that floodplain habitats support higher levels of primary and secondary productivity than surrounding areas with less frequent inundation (Pettit et al., 2011). Infiltration of water into the soil during the wet season and along persistent streams often enables riparian habitats to form an important interface between the aquatic and terrestrial environment. While riparian habitats often occupy a relatively small proportion of the catchment, they frequently have a higher abundance and species richness than surrounding habitats (Pettit et al., 2011; Xiang et al., 2016). Riparian habitats that fringe the rivers and streams of the Victoria catchment have been rated as having moderate to high cover and structural diversity for riparian vegetation with some impacts at some locations (Kirby and Faulks, 2004). These riparian habitats include widespread *Eucalyptus camaldulensis* overstorey with *Lophostemon grandiflorus*, *Terminalia platyphylla*, *Pandanus aquaticus* and *Ficus* spp. *Acacia holosericea* and *Eriachne festucacea* occur as dominant understorey species across many parts of the catchment (Kirby and Faulks, 2004). Further away from the creeks and rivers, vegetation in the Victoria catchment becomes sparser.

In the dry season, biodiversity is supported within perennial rivers, wetlands and the inchannel waterholes that persist in the landscape. In ephemeral rivers, the waterholes that remain become increasingly important as the dry season progresses; they provide important refuge habitat for species and enable recolonisation into surrounding habitats upon the return of larger flows (Hermoso et al., 2013). Waterholes provide habitat for water-dependent species including fish, sawfish and freshwater turtles, as well as providing a source of water for other species more broadly within the landscape (McJannet et al., 2014; Waltham et al., 2013).

GDEs occur across many parts of the Victoria catchment and come in different forms, including aquatic, terrestrial and subterranean habitats. Aquatic GDEs contain springs and river sections that hold water throughout most dry seasons due to groundwater discharge. Aquatic GDEs are important for supporting aquatic life and fringing vegetation and in the wet-dry tropics often provide critical refuge during periods of the late dry season (James et al., 2013). Vegetation occurring adjacent to the waterways in the Victoria catchment relies on water from a range of sources (surface water, soil water, groundwater) which are seasonally dynamic and highly spatially variable across riparian and floodplain habitats. Perennial floodplain vegetation that uses groundwater when it is within reach of the root network, particularly during the dry season or drought, but the origin of the groundwater used has only been infrequently investigated (e.g. Canham et al. (2021)). In some locations vegetation may be sustained by water available in unsaturated soils and so never use groundwater. In other locations vegetation may use groundwater sourced from local alluvial recharge processes; alternatively, regional groundwater may be critical for maintaining vegetation condition. The latter situation applies to habitats of monsoon vine forest located within the Bradshaw Field Training Area DIWA site (Lands Planning and Environment, 1998). Subterranean aquatic ecosystems in the Victoria catchment include known sinkholes associated with the Montejinni Limestone that are mapped along the south-eastern edge of the Victoria catchment. These sinkholes may contain groundwater and support aquatic ecosystems throughout the dry season, but their connection to groundwater is currently unknown. Some subterranean species are distributed across a broad spatial range, while others have highly restricted ranges, which makes them more vulnerable to local changes where they occur (Oberprieler et al., 2021).

Marine and estuarine habitats in northern Australia are highly productive and have high cultural value and include some of the most important, extensive and intact habitats of their type in Australia, many of which are recognised as being of national significance. The mouth and estuary of the Victoria River is up to 25 km wide and includes extensive mudflats and mangrove stands (Kirby and Faulks, 2004). Although mangroves and mudflats are prominent along coastal margins, the mangrove communities along the estuary are recognised as being low in species richness with about ten species recorded. The dominant mangrove species in the catchment is *Avicennia marina*, which is largely confined to the estuary (Kirby and Faulks, 2004). The Legune IBA extends along the south-west shores of the inner Joseph Bonaparte Gulf, from the mouth of the Keep River in the west to the mouth of the Victoria in the east and then north beyond the Victoria catchment. The Legune IBA can support over 15,000 waterbirds across mudflats, salt flats, seasonally inundated wetlands (BirdLife International, 2023). Marine habitats in northern Australia are vital for supporting important fisheries, including banana prawn, mud crab and barramundi, as well as for biodiversity more generally, including waterbirds, marine mammals and turtles. In addition, the natural waterways of the sparsely populated catchments support globally significant stronghold populations of endangered and endemic species (e.g. sharks and rays) that use a combination of both marine and freshwater habitats.

2.1.4 Significant species and ecological communities of the Victoria catchment

There are a number of aquatic and terrestrial species in the Victoria catchment currently listed as Critically Endangered, Endangered and Vulnerable under the EPBC Act and by the Northern Territory Government's wildlife classification system, which is based on the International Union for Conservation of Nature (IUCN) Red List categories and criteria (Figure 2-3). The Commonwealth's Protected Matters Search Tool (PMST; Department of Agriculture Water and the Environment (2021)) lists 45 Threatened species for the Victoria catchment, four of which are listed as Critically Endangered (Nabarlek (*Petrogale concinna concinna*), Rosewood keeled snail (*Ordtrachia septentrionalis*), curlew sandpiper (*Calidris ferruginea*) and the eastern curlew (*Numenius madagascariensis*)). Also listed are 49 migratory species.

The aquatic habitats of the Victoria catchment support some of northern Australia's most archetypical and important wildlife species, including sawfish, marine turtles, Australian snubfin dolphins and river sharks (Department of Agriculture Water and the Environment, 2021) that occur in the estuaries of the Victoria River and the coastal waters of the Joseph Boneparte Gulf. Recent surveys demonstrate the river to be a globally significant stronghold for three endangered species: freshwater sawfish (*Pristis pristis*; listed as Vulnerable under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and Critically Endangered on the International Union for Conservation of Nature (IUCN) Red List of Threatened Species); speartooth shark (*Glyphis glyphis*; Critically Endangered, EPBC Act and IUCN); and northern river shark (*Glyphis garricki*; Endangered, EPBC Act and IUCN). While the spear tooth shark is not among the four species that are listed as Critically Endangered in the Commonwealths PMST, recent surveys have identified the species in the estuarine habitats of the Victoria River Dr Richard Pillans, CSIRO Environment, Brisbane, 2022, pers. comm.). Saltwater crocodiles (*Crocodylus porosus*) frequent the Victoria River and have been recorded considerable distances into freshwater reaches (Atlas of Living Australia, 2023).

Across the catchment are many lesser-known plants and animals that are also of great importance. Owing to healthy floodplain ecosystems and free-flowing rivers (Grill et al., 2019; Pettit et al., 2017), very few freshwater fishes in the study area are threatened with extinction. Many of these fish species do not enter the marine environment and remain within the riverine and wetland habitats of the catchment. Neil’s grunter (*Scortum neili*) is endemic to the Victoria catchment and is listed as Endangered (IUCN). Neil’s grunter is restricted to sections of the East Baines and Angalarri rivers where it inhabits narrow, deep sections of the river that have slow flowing freshwater that is shaded by overhanging trees (Gomon and Bray, 2017). Species including the eastern curlew (Critically Endangered), the curlew sandpiper (Critically Endangered) and the red knot (*Calidris canutus*; Endangered) use habitats including the Legune Wetlands IBA as an important stopover habitat (EPBC Act) (Australian Government, 1999; Department of Agriculture Water and the Environment, 2023b). The night parrot (*Pezoporus occidentalis*; Endangered) has been recorded in sandstone/spinifex country in the region of the nearby Keep River National Park (Department of Agriculture Water and the Environment, 2021; McKean, 1985) as well as occurrences of the purple-crowned fairy-wren (*Malurus coronatus*; Endangered) and the Gouldian finch (*Erythrura gouldiae*; Endangered) occurring within the catchment (Atlas of Living Australia, 2023; Department of Agriculture, Water and the Environment, 2021).

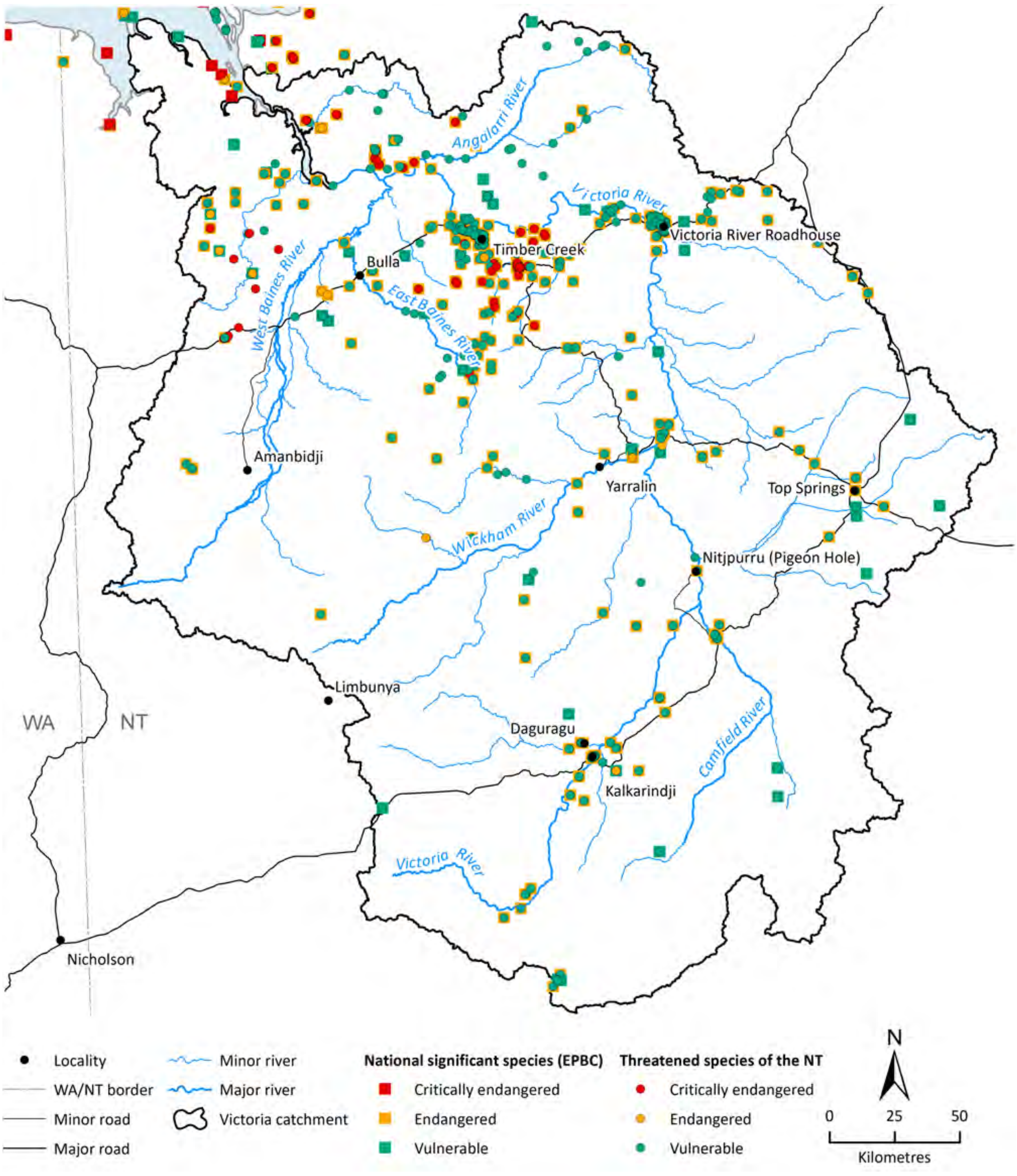

Figure 2-3 Distribution of threatened fauna species listed under the EPBC Act (Cth) and by the Northern Territory Government in the Victoria catchment

Data source: Department of Environment Parks and Water Security (2019a)

2.1.5 Current condition and potential threats in the Victoria catchment

There is a range of economic enterprises, infrastructure and other human impacts in the Victoria catchment. The nature and extent to which human activities have modified the habitats and affected species of the Victoria catchment varies, but most sites have some level of impact (Kirby and Faulks, 2004). Previous assessments have rated the riverine habitat in the Victoria catchment as being high or very high overall quality and largely intact with high wilderness value and predominantly unaffected by clearing or development at the time of assessment, although threatening processes operate. These include grazing, roads, river crossings and impacts from pest species, including both feral animals and weeds (Department of Agriculture Water and the Environment, 2023a; Kirby and Faulks, 2004).

The study area includes the localities and towns of Timber Creek, Yarralin, Nitjpurru (Pigeon Hole) and Top Springs, which provide Indigenous homelands, support a vital tourism industry and act as regional hubs for many of the stations across the catchment. While a moderate proportion of the catchment is under conservation reserves, the catchment does face environmental threats. Fishing in northern Australia is highly valuable, and the waters of the Victoria catchment and the nearby marine zone contribute to important recreational, commercial and Indigenous catches including barramundi, redleg banana prawns (*Penaeus indicus*) and a variety of other species.

Northern Australia more broadly encompasses some of the last relatively undisturbed tropical riverine landscapes in the world with low levels of flow regulation and low development intensity (Pettit et al., 2017; Vörösmarty et al., 2010). Riparian vegetation characteristics of the Victoria catchment are considered not to be affected by extensive clearing or development, although impact that occurs is often associated with stock and pest species accessing watering points (Kirby and Faulks, 2004).

One of the most significant environmental threats to remote regions across northern Australia is that of introduced plants and animals. In the Victoria catchment, pig (*Sus scrofa*), water buffalo (*Bubalus bubalis*), camel (*Camelus dromedarius*), donkey (*Equus asinus*), cat (*Felis catus*) and cane toad (*Rhinella marina*) are among the invasive animals (Department of Agriculture Water and the Environment, 2021). Weed species of interest in and around the Victoria catchment list 20 species of national significance. Invasive plants of concern include gamba grass (*Andropogon gayanus*), para grass (*Brachiaria mutica*), giant sensitive plant (*Mimosa pigra*) and prickly acacia (*Vachellia nilotica*) (Department of Agriculture Water and the Environment, 2021). Some of these, including sensitive tree and para grass, have significantly affected undeveloped rivers more broadly in northern Australia (Davies et al., 2008). Surveys within the Bradshaw Field Training Area indicated the presence of six feral species, namely, cats, horses, donkeys, pigs, wild cattle and buffalo (Lands Planning and Environment, 1998). eDNA analysis of water samples taken in this study detected cane toads, wild pig, cattle and dingo at several sites (Appendix B).

3 Ecological assets from the Victoria catchment and marine region

Northern Australia's rivers, floodplains and coastal regions contain high diversity, including at least 170 fish species, 150 waterbird species, 30 aquatic and semi-aquatic reptiles, 60 amphibian species and 100 macroinvertebrate families (van Dam et al., 2008). The ecologies of the freshwater and freshwater-dependent terrestrial and marine systems are supported by, and adapted to, the highly seasonal flow regimes of the wet-dry tropics. Water resource development and climate change threaten to affect these habitats and species. This section provides a synthesis of the prioritised assets relevant to the Victoria catchment for the purpose of understanding the ecology outcomes of flow regime change. Table 1-1 lists the assets used in the ecology activity.

3.1 Fish, sharks and rays

3.1.1 Barramundi (*Lates calcarifer*)

Description and background to ecology

Barramundi are a large (>1 m standard length) opportunistic-predatory fish (order Perciformes) that occurs throughout northern Australia. The species is catadromous (i.e. it migrates down rivers to spawn in the sea) and occurs in 'catchment to coast' habitats throughout the west Indo-Pacific region, including estuaries, rivers, lagoons and wetlands across northern Australia (Crook et al., 2016; Pender and Griffin, 1996; Roberts et al., 2023; Russell and Garrett, 1983; 1985). The fish is long lived (living up to about 32 years) and fast growing, and individuals begin life as a male but change to female as they age (protandrous hermaphrodite): they occupy freshwater habitats as males in the first years of life and saltwater habitats as older females. The species is of ecological importance, capable of modifying the estuarine and riverine fish and crustacean communities (Blaber et al., 1989; Brewer et al., 1995; Milton et al., 2005; Russell and Garrett, 1985).

Barramundi is arguably the most important fish species for commercial, recreational and Indigenous subsistence fisheries throughout Australia's wet-dry tropics. It makes up a substantial component of the total commercial fish catch in northern Australia (Savage and Hobsbawn, 2015). In 2013–14, barramundi comprised 28% of the $31 million wild-caught fishery production in the NT. Commercial catch per unit effort in the NT has increased from about 7 kg per 100 m of net per day in the early 1980s to over 30 kg per 100 m of net per day in the 2010s (Northern Territory Government, 2018). Commercial and recreational catches make up the largest proportions of all catches in the NT, though the Indigenous catch is significant in some years.

Barramundi is a fish of cultural significance for the Indigenous community as well as being an important food source (Jackson et al., 2012). The movements of barramundi between habitats are indicators of the change in season for Indigenous communities across tropical Australia (Green et al., 2010). The movements relate to the barramundi's habitat requirements during its life cycle, which rely on seasonal variation in river flows to access habitats. In the NT, the quantity of Indigenous catch of barramundi in the study area is less certain than other fisheries.

Barramundi life history renders the species critically dependent on river flows (Plagányi et al., 2023; Tanimoto et al., 2012). Large females (older fish) and smaller males (younger fish) reside in estuarine and littoral coastal habitats. Mating and spawning occur in the lower estuary during the late dry season to early wet season, and new recruits move into supra-littoral and freshwater habitats. Coastal salt flat, floodplain and palustrine (i.e. non-tidal wetland) habitats depend on overbank flows for maintenance and connectivity (Crook et al., 2016; Russell and Garrett, 1983; 1985).

Young fish, as males, may move large distances upstream and reside in palustrine billabongs for 3 to 4 years before maturing and migrating downstream. This ontogenetic migration (i.e. migration between habitats at different life stages) requires palustrine–riverine and riverine–estuarine connectivity; hence migration depends on catchment flows. Barramundi transform to females at about 6 years old when they mix with younger males within river estuaries and breed.

Over the last decade, studies using otolith microchemistry and fish-tag telemetry have provided greater understanding of barramundi use of freshwater, estuarine and marine habitats (Crook et al., 2016; Roberts et al., 2019) than initial life-history studies in the 1980s (Pender and Griffin, 1996). Crook et al. (2016) proposed three primary life-history strategies employed by barramundi: (i) some male adults return to the estuary to spawn after spending several years in freshwater habitats, (ii) some individuals delay downstream spawning migrations for 6 to 10 years until they have undergone the transition to females in freshwater habitats, and (iii) some barramundi remain in estuarine waters and complete their life cycle without entering freshwater systems (Crook et al., 2016; Roberts et al., 2019; Robins et al., 2021). The variation in migration strategy has been shown to be triggered by variation in the strength of the annual monsoon and resultant flow regime (Crook et al., 2016; Roberts et al., 2023), making the species particularly vulnerable to water resource development (Robins et al., 2021).

Moreover, the effects of different levels of river flow are now better understood. During high-flow years (a strong wet season), barramundi tend to remain within the estuary (Roberts et al., 2023); the estuary becomes a brackish habitat, and terrestrial and palustrine productive inputs probably render the estuary prime habitat (Burford and Faggotter, 2021). During low-flow years (a drier wet season), barramundi are twice as likely to immigrate to riverine and palustrine habitats, probably seeking better foraging conditions among freshwater habitats (~80% move to riverine habitats in dry years) (Crook et al., 2022; Roberts et al., 2023). In years with strong monsoon rainfall, about 60% of barramundi remain within the estuarine habitats that are strongly linked and receiving inputs from the catchment (Roberts et al., 2023). Overall, approximately 62% of barramundi in tropical Australian rivers were catadromous, that is, migrating to freshwater habitats and returning to saline waters to spawn (Roberts et al., 2019). In Gulf of Carpentaria rivers, quantitative modelling of the relationship between flow and catch for barramundi has confirmed the dependence of barramundi on flow regime, predicting declines in barramundi biomass directly related to water extraction or impoundment that modifies the seasonal historical flows (Plagányi et al., 2023).

Barramundi in the Victoria catchment

Barramundi are abundant in the relatively pristine habitats of the estuarine and freshwater reaches of the Victoria River. However, there are few data on recreational or commercial catch or the presence or absence of barramundi in the Victoria catchment (Figure 3-1). Historically, the remoteness of the Victoria River resulted in the system's fish fauna being poorly studied (Larson et al., 2013). In addition, the high tide range experienced in the estuarine sections of the river renders it difficult to navigate safely and limits access for commercial fishers. Over recent decades, recreational fishers report attractive barramundi catches from the Victoria River and its tributaries, suggesting the fish is common throughout the catchment, as would be expected based on scientific knowledge of the species' distribution in other tropical rivers (videos found on the YouTube platform show active barramundi fishing in the catchment).

A study of recreational fishing trends in the NT (2009–10) determined that 23% of barramundi catch taken by recreational fishers came from the 'west coast' region of the NT (West et al., 2012). However, the 'west coast' regions included the Daly River (a river with high levels of fishing) and extended south-west to the NT and WA border, including the Victoria River.

Nowadays, a major highway crosses the Victoria catchment, allowing access to the river. Access using four-wheel-drive vehicles and the desire of recreational fishers for a 'wilderness experience' have supported the establishment of accommodation, boat ramps and other river access that encourage fishing and other activities. For example, the Parks and Wildlife Commission of the Northern Territory produces a fact sheet for the Judbarra / Gregory National Park that covers part of the Victoria catchment. The fact sheet highlights fishing as a key activity within the park and barramundi as a target catch (see Parks and Wildlife Commission of the Northern Territory (n.d.)).

The Victoria River currently experiences low levels of commercial fishing for barramundi; however, barramundi are common in the river estuary and commercial interest in fishing the river is increasing (Thor Saunders (Northern Territory Fisheries Research), 2022, pers. comm.). A large tidal range and strong currents within the estuary are deterrents to successful commercial fishing (Thor Saunders (Northern Territory Fisheries Research), 2022, pers. comm).

Water samples for eDNA analyses collected for this study detected barramundi in the Baines River, a tributary of the Victoria River, and at Nitjpurru (Pigeon Hole) on the upper Victoria River about 400 km from the estuary. In addition, during 2018 and 2019, Dr Richard Pillans sampled various locations in the Victoria catchment for sawfish (see Figure 3-13 for sample site locations). Though not recorded as they were not target catch, Dr Pillans suggested that 3 to 5 barramundi were caught per site in gillnets set to catch sawfish, about 50 barramundi per sampling trip (Richard Pillans (CSIRO Environment), 2023, pers. comm.).

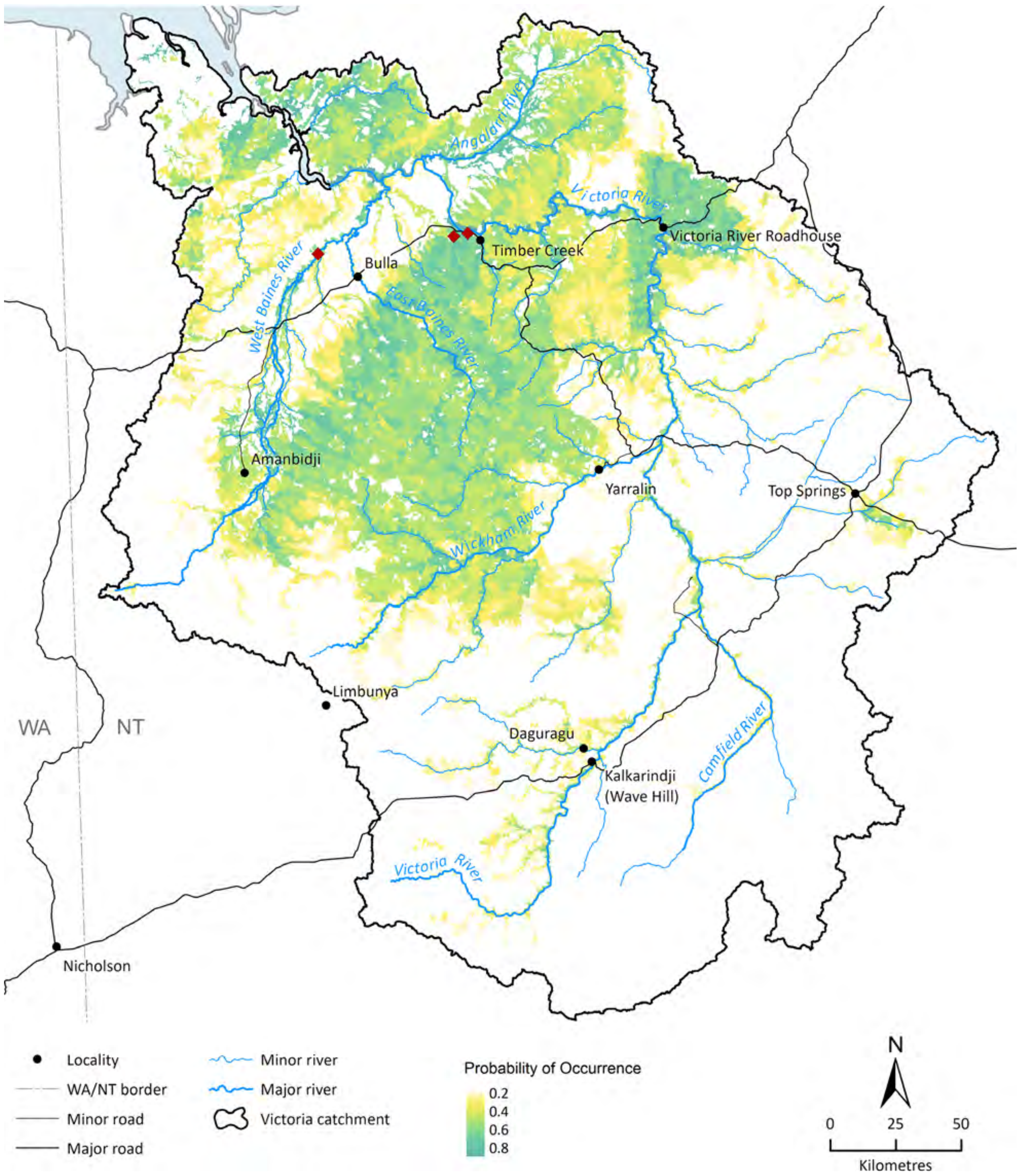

Figure 3-1 Observed locations of barramundi (*Lates calcarifer*) and their modelled probability of occurrence in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for barramundi

The barramundi life-history strategy is critically dependent on the natural flow regime in the wet-dry tropics and would be affected by interruptions to the natural flows of northern Australian rivers (Crook et al., 2022; Roberts et al., 2023). Spawning occurs in the lower estuary and young male fish move into floodplain and freshwater habitats when suitable flows provide access (Crook et al., 2016; Roberts et al., 2019; Russell and Garrett, 1985).

It had been thought that individual male barramundi moved upstream to freshwater riverine and palustrine habitats at about 3 to 4 years old before maturing and migrating downstream to the estuary at about 6 years old; there they would transform into females and mix with younger males and breed. Recent studies using new technologies have proposed that subsets of young-of-year individuals adopt a range of estuarine and riverine strategies (Roberts et al., 2019). Different migration strategies across freshwater and marine zones are triggered by variation in the wet-season flow regime and connectivity (Crook et al., 2016), making the species particularly vulnerable to water resource development (Doupé et al., 2005).

The recruitment of barramundi to nursery habitats is moderated by floodwater access to supra-littoral, lagoon and riverine habitats (Russell and Garrett, 1983). Both longitudinal and floodplain connectivity require significant flood heights that let fish travel upstream or out of the river channel in search of habitats that increase their survival and growth during their juvenile stage. Peak spring tides also may facilitate access to supra-littoral habitats, supplemented by small early-season floods (Russell and Garrett, 1983); however, individuals also recruit to the population after spending larval and juvenile stages completely in estuarine water (Milton et al., 2008). Individuals around 1 year old move out of the supra-littoral habitats − they may move upstream into freshwater reaches (Russell and Garrett, 1985) or return to the estuary (Blaber et al., 2008; Milton et al., 2008) where they may reside for several years.

Adolescents and adults remain in perennial freshwater habitats for periods of months to years until flood-moderated connectivity lets them return to the river before emigrating downstream to the estuary and near-shore zones as adults to spawn (Blaber et al., 2008). Consequently, the annual wet season, and subsequent increase in flows, is a major determinant of their access to juvenile habitat and connectivity back to the coastal zone. There is a correlation between seasonal flood flow and juvenile recruitment strength and subsequent adult stocks, possibly lagged by 1 to 5 years (Halliday et al., 2012; Leahy and Robins, 2021; Robins et al., 2005). Historically, most northern rivers are unregulated with no large dams as barriers to migration, which supports life-history diversity in response to variability in monsoon-driven river flows (Roberts et al., 2023). Large instream dams that sever upstream−downstream connectivity may have greater effects on barramundi populations than reduced flows by blocking access to riverine and palustrine habitats within a large proportion of a catchment (Doupé et al., 2005). The ecological functions that support barramundi, and their associated flow requirements, are summarised in Table 3-1.

Table 3-1 Ecological functions supporting barramundi and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of riverine waterhole persistence as freshwater habitat during the dry season	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintenance of natural ephemerality vs conversion to perennial systems	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintenance of water quality in riverine waterholes as freshwater habitat during the dry season	Baseflows into waterholes	Low flows
Maintenance of palustrine waterhole persistence as habitat during the dry season and reconnection to support migration during the wet season	Floodplain inundation extent and duration during the wet season	High flows
Riverine waterhole flush and reconnection, and establishing brackish estuarine conditions in the late dry season	Late dry-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue barramundi immigration to riverine and palustrine habitats, their growth and survival in these habitats, and emigration to marine habitats	Moderate flood flows. Higher barramundi growth and survival during wet-season high-flood flows	Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that cue barramundi emigration from freshwater to marine habitats, and palustrine inundation and extensive, ephemeral barramundi habitat during the wet season	Large flood flows to inundate the floodplain and maintain palustrine wetlands and productivity hotspots during the wet season. High-level barramundi growth and superior survival	High flows, their frequency and seasonal reoccurrence

Pathways to change for barramundi

In the wet-dry tropics of northern Australia, the wet season stimulates primary productivity and connectivity within and between isolated palustrine and riverine habitats that are stressed by the end of the dry season (Ndehedehe et al., 2020a; Ndehedehe et al., 2021). Barramundi juvenile recruitment to freshwater habitats and fish growth rates are enhanced by large wet-season flows during the ‘peak flows’ wet-season months of January to March (Crook et al., 2022; Leahy and Robins, 2021) and also by flows higher than baseflows that precede and follow the wet-season peak-flow months. Higher flows during an early start to the wet season (October to December) or late-season rainfall (April to June) also promote superior growth than do low-level flows over the same months (Leahy and Robins, 2021). The research demonstrates that both the level and the seasonality (timing) of flood flows affects barramundi growth.

High river flows expand the extent of palustrine and estuarine-margin habitats, increase connectivity, deliver nutrients from terrestrial landscapes, create hot spots of high primary productivity and food webs, increase prey productivity and availability, and increase migration within the river catchment (Burford et al., 2016; Burford and Faggotter, 2021; Leahy and Robins, 2021; Ndehedehe et al., 2021). These factors promote the successful recruitment of juvenile barramundi to freshwater habitats and the growth and survival of those that inhabit both freshwater and estuarine habitats within the river catchments. In years of poor wet seasons and low rainfall that result in naturally low-level flows, or flows that are reduced by anthropogenic activity such as water extraction, the range of facultative habitat access and ecosystem processes available to barramundi is reduced; hence, growth and survival are reduced (Crook et al., 2020; Leahy and Robins, 2021; Roberts et al., 2023; Robins et al., 2006; Robins et al., 2005).

The impact of water resource development such as the construction of dams or water harvest at several levels of extraction on coastal barramundi populations has been modelled (Plagányi et al., 2023). An array of water harvest and impoundment scenarios on the Mitchell, Gilbert and Flinders rivers in the eastern and southern Gulf of Carpentaria reduced both the biomass and commercial catch of barramundi by 4% to 61% depending on the water resource development scenario. The risk to the barramundi population was assessed as minor for one of four water resource development scenarios; the remaining three were assessed as negligible. However, risk to the commercial harvest of barramundi was assessed as moderate for two scenarios, minor for one and negligible for one (Plagányi et al., 2023).

Recent research on monsoon-driven habitat use by barramundi has shown that, during drier years with lower river flows, a large proportion of the juvenile barramundi immigrate upstream from estuarine spawning habitat to freshwater habitats (Roberts et al., 2023). The flow-catch modelling undertaken for Gulf of Carpentaria rivers showed that maintaining low-level flows was important to support the barramundi population (Plagányi et al., 2023). Low-level flows maintained by pump-initiation thresholds that protected river flows from extraction until relatively high river flow levels were reached had less impact on barramundi biomass than the same allocation of water pumped at low-level river flows. The results of Roberts et al. (2023) assist to interpret the benefit of water-extraction-pump thresholds for barramundi: during years of limited monsoon-driven flood flows, a higher proportion of juvenile barramundi move upstream than during high flood flow years; the individuals immigrating upstream use those unregulated lower-level flows to access riverine and palustrine habitats, to the benefit of the population in subsequent years. During years of high-level flood flows driven by a strong monsoon, most barramundi remain in the estuary, benefiting from riverine inputs transported to the estuary and not necessarily migrating upstream.

Plagányi et al. (2023) showed that both constructing dams and harvesting river flows via pumped water extraction affect aspects of the barramundi life history that limit the resilience of its population. Anthropogenic reduction in the volume and duration of high-level flows and induced variability in the seasonality and volume of low-level flows affect habitat connectivity, migration, predation, growth and survival of barramundi (Leahy and Robins, 2021; Plagányi et al., 2023; Roberts et al., 2023). The ecological outcomes of threatening processes on barramundi in northern Australia catchments, and their implications for changes to growth, mortality, refuge and habitat, and community structure, are presented in Figure 3-2.

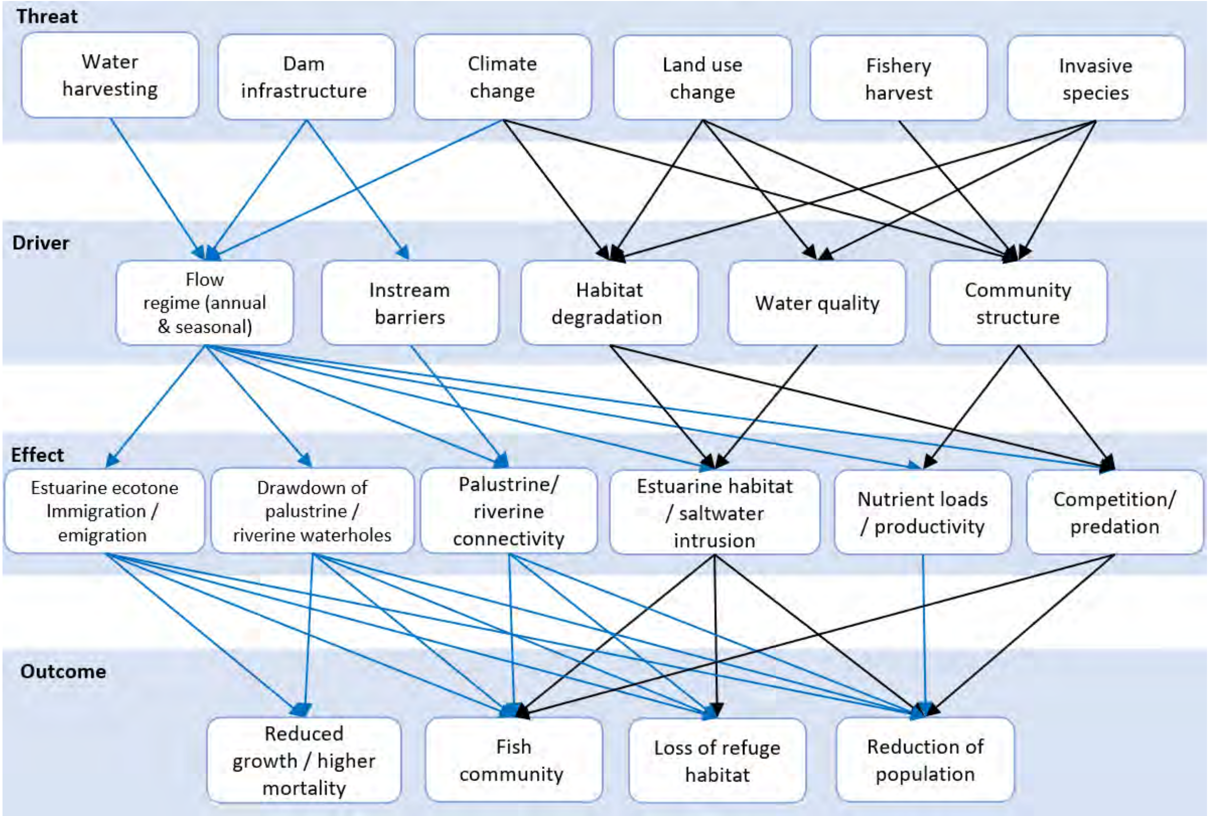

Figure 3-2 Conceptual model showing the relationship between threats, drivers, effects and outcomes for barramundi in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.2 Catfish (order Siluriformes)

Description and background to ecology

Catfish are a highly diverse group that inhabit both inland and coastal waters globally. In northern Australia, catfish are found both in freshwater and marine habitats. The group includes freshwater species, marine species, and some that move between the river and the estuary (Pusey et al., 2020). Catfish in northern Australia belong to two families: Plotosidae (19 species in total) and Ariidae (17 species). Plotosidae are found in the Eastern Pacific and Indian Ocean whereas Ariidae are a global family. Both are found in both freshwater and marine habitats.

Most catfish are bottom-feeding omnivores that feed on algae, submerged macrophytes, invertebrates and smaller fish. Species within the Ariidae are slow growing and generally large bodied. The family is notable for its reproductive traits: it has the largest eggs of any teleost group (>10 mm) and males exhibit strong parental care behaviour, incubating the eggs and developing the young in the mouth for up to 5 weeks (Pusey et al., 2004). Because of the tendency to feed opportunistically, ariid catfish can be very competitive, consuming a variety and large volumes of food. Thus, they can make up a lot of biomass in a catchment (Crook et al., 2020).

The key plotosid species are reasonably tolerant to high temperatures and low dissolved oxygen levels, but fish kills at very low dissolved oxygen levels have been reported (Bishop, 1980). The key threat to the two main *Neosilurus* species is potential flow barriers. Plotosidae need high flows to trigger spawning migration, and they require a barrier-free passage to spawning grounds in the headwater streams. While not as culturally and commercially important as barramundi or sooty grunters (*Hephaestus fuliginosus*), the fork-tailed catfish (*Neoarius graeffei*) has considerable importance as a subsistence fish for Indigenous communities (Finn and Jackson, 2011; Jackson et al., 2011).

Catfish in the Victoria catchment and marine region

Ten species of catfish have been recorded in the Victoria catchment, including six species mapped from the Atlas of Living Australia data below (Figure 3-3). An additional four species are recorded in the ALA but not mapped: *Anodontiglanis dahli and Porochilus rendahli* are found in freshwater systems and *Nemapteryx armiger* and *Netuma proxima* are marine. Larger-bodied ariid catfish like *Neoarius graeffei, Neoarius midgleyi* and *Sciades leptaspis* are mainly found in the main stems of the Victoria River and the larger tributaries like the Wickham River. The usually smaller-bodied *Neosilurus* species are mainly found in smaller tributaries. The modelled distribution of *N. graeffei* is shown in Figure 3-4.

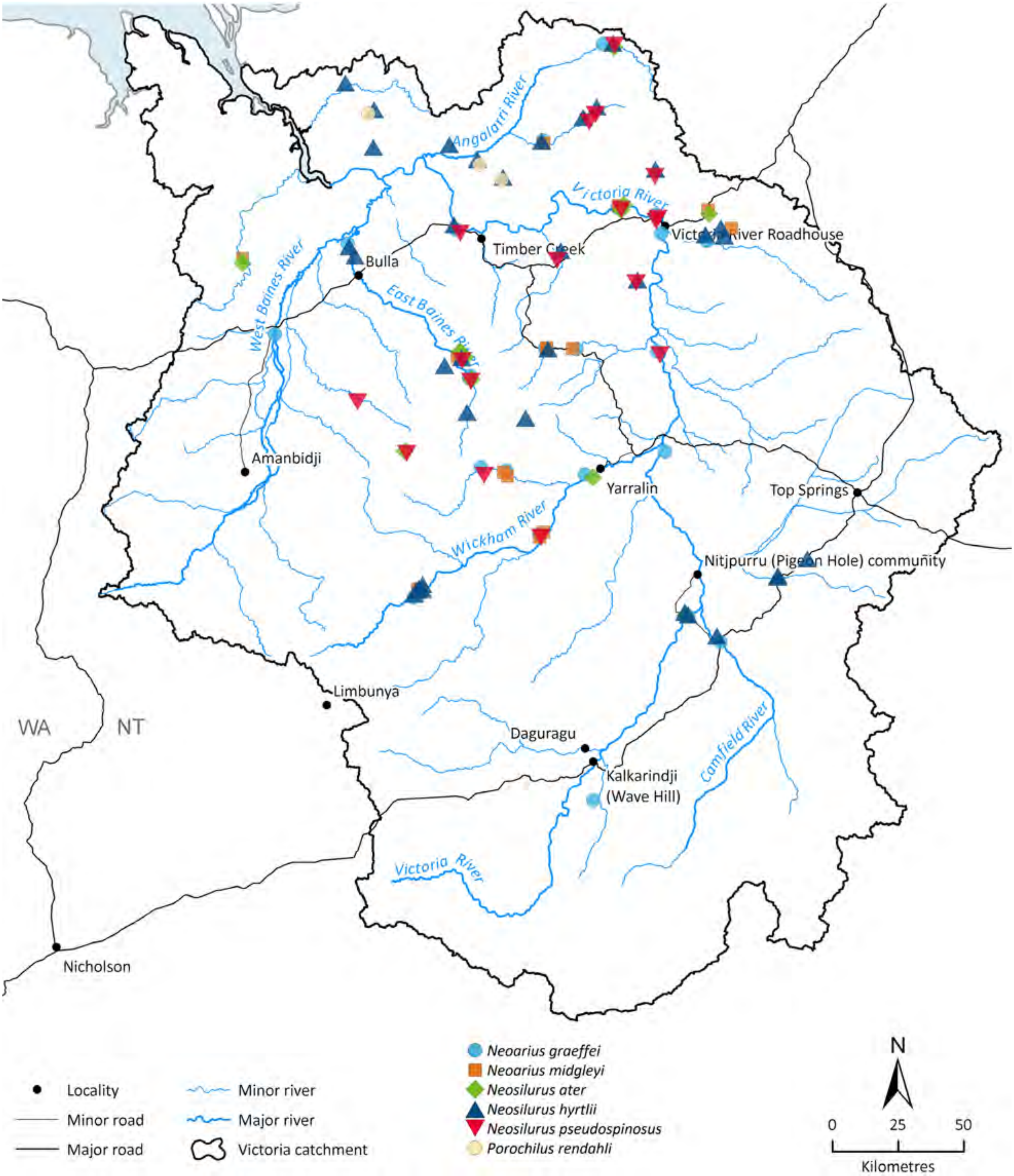

Figure 3-3 Observed locations of catfish in the Victoria catchment

Data source: Atlas of Living Australia (2023)

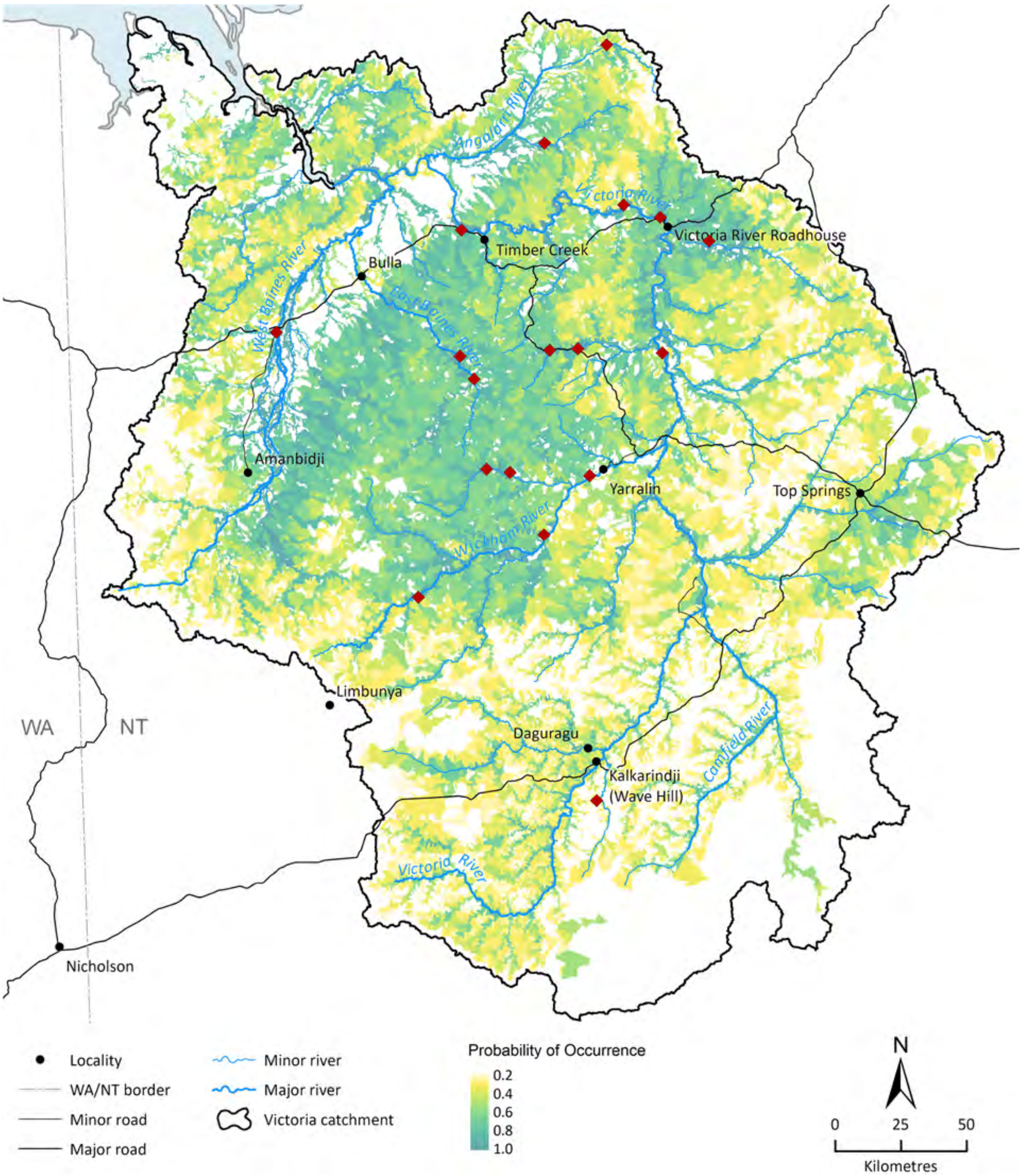

Figure 3-4 Modelled probability of occurrence of fork-tailed catfish (*Neoarius graeffei*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for catfish

Flow–ecology relationships for catfish depend upon life histories, and therefore differ between the two families Plotosidae and Ariidae. The smaller-bodied Plotosidae can be found in many types of hydraulic habitat, including dune lakes (Arthington, 1984), but not the very large estuarine reaches (Pusey et al., 2004). Habitat use can change seasonally – a study in the Alligator River found *Neosilurus hyrtlii* in sandy creeks only during the late dry season. In the late wet and early dry seasons, *N. hyrtlii* was recorded from lowland lagoons, floodplain lagoons and perennial streams of the escarpment. Distribution seems to depend on context and habitat, as another study in a Queensland catchment found upstream migration in the wet season (Orr and Milward, 1984).

Neosilurus ater prefers faster-flowing habitats in the main channel (Allen, 1982; Bishop et al., 1990) but has also been found to migrate upstream from its adult habitat in the lowland rivers to tributaries to spawn (Orr and Milward, 1984). Based on these observations, Pusey et al. (2004) conclude that 'the development of water infrastructure that inhibits upstream movement, or which captures high-flow events and therefore removes the probable stimulus for spawning migrations, is highly likely to negatively impact on this species'.

Ariidae is a family of fairly resilient catfish that, unlike Plotosidae, often prefer larger river channels or estuaries (Bishop et al., 2001). This is especially the case for *Neoarius graeffei* and *Sciades leptaspis*, which can tolerate slow-flowing or stagnant water; however, barriers can hinder dispersal for smaller size classes, even if barrier mitigation is provided (Stuart and Berghuis, 1999). Also, while not directly flow related, cold water from stratified impoundments can hinder spawning cues for *N. graeffei* (Kailola and Pierce, 1988).

Neoarius midgleyi requires connection to the offchannel floodplain as habitat during the dry season. Kailola and Pierce (1988) also report *N. midgleyi* in a variety of habitats, including fast-flowing rivers, billabongs, creeks, deep pools and desiccating waterholes. This species is found less in the main channel and estuary. The ecological functions that support catfish, and their associated flow requirements, are summarised in Table 3-2.

Table 3-2 Ecological functions supporting catfish and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintain riverine waterhole persistence as freshwater habitat during the dry season	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintain natural ephemerality vs conversion to perennial systems	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintain water quality in riverine waterholes as freshwater habitat during the dry season	Baseflows into waterholes	Low flows
Maintain palustrine waterhole persistence as habitat during the dry season and reconnection to support migration during the wet season	Floodplain inundation extent and duration during the wet season	High flows
Riverine waterhole flush and reconnection, and establishing brackish estuarine conditions in the late dry season	Late dry-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue spawning migration to upstream and palustrine habitats, their growth and survival in these habitats	Moderate flood flows. Higher fish growth and survival during wet-season high-flood flows	Flow timing and magnitude. Seasonality of flows

Pathways to change for catfish

This section discusses the possible ecological outcomes of threatening processes on catfish in northern Australia and their implications for changes in habitat shifts, community structure and population sizes presented in a conceptual model. Four of the key threats in the conceptual model are related to flow modification: water harvesting, dam infrastructure, river regulation and climate change. Overall, the ariid catfish species present in some northern catchments are fairly pollution tolerant, yet they all depend to some degree on a natural flow regime (Pusey, 2004). All species depend on connections to the floodplain, often for the purpose of recruitment. River regulation and extraction can reduce overbank flows, reducing connection frequency and therefore recruitment opportunities. As agricultural growing seasons often overlap with fish spawning seasons, water is likely extracted at these important times. Furthermore, environmental flows can be released at the wrong time (Linke et al., 2011), again leading to a possible reduction in recruitment and thus population size.

Apart from being barriers to movement, dams can contribute to cold water pollution as released, stratified water can be significantly colder. While there are no data on catfish in tropical streams, Pusey (2004) hypothesises that in upland areas winter thermal tolerances of *Neoarius graeffei* are close to the thermal limit, indicating potential vulnerability to cold water releases from a dam.

Some Plotosidae species prefer flowing water in the main channel. This could be affected by overextraction or even structural changes like dams, which can alter cease-to-flow periods (Allen, 1982; Bishop et al., 1990). As described above, the combination of impact on movement and missing spawning migration triggers is highly likely to affect population sizes of Plotosidae, especially *Neosilurus ater* (Pusey, 2004). However, this could differ under varying circumstances; for example, some catfish in Queensland have been found to migrate upstream in the wet season (Orr and Milward, 1984).

The ecological outcomes of threatening processes on catfish in northern Australia, with their implications for changes to habitat, community structure and population size, are illustrated in Figure 3-5.

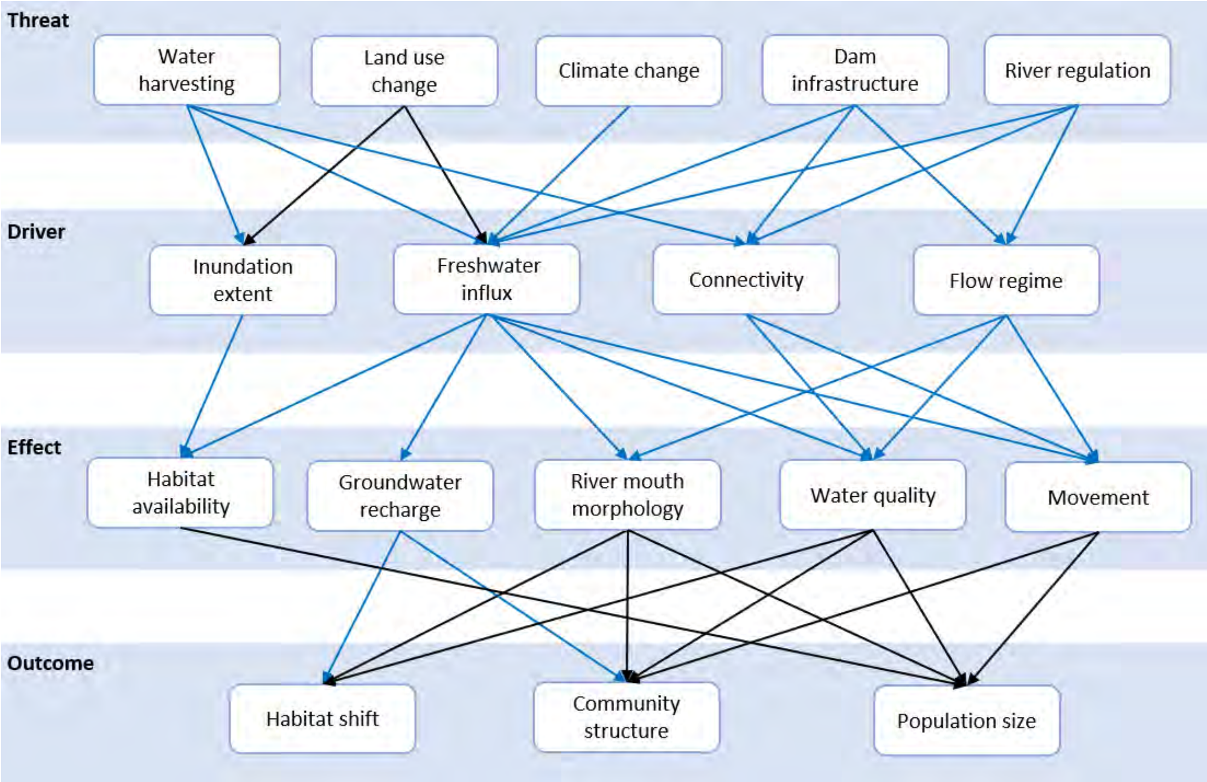

Figure 3-5 Conceptual model showing the relationship between threats, drivers, effects and outcomes for catfish in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.3 Grunters (family Terapontidae)

Description and background to ecology

In northern Australia there are a 37 species of grunter from 11 genera, with the most species-rich genera being *Hephaestus, Scortum, Syncomistes* and *Terapon*. Grunters inhabit riverine, estuarine and marine waters. Many grunter species spend their entire lives in fresh water, while other species inhabit marine or estuarine waters, only sometimes venturing into fresh water (Pusey et al., 2004).

The Terapontidae are a perciform (perch like) family of fishes of medium diversity, restricted to the Indo-Pacific region. They are characterised by a single long-based dorsal fin, which has a notch marking the boundary between the spiny and soft-rayed portions. Terapontidae are soniferous (i.e. they can both vocalise and hear well) so may be sensitive to noise (Smott et al., 2018).

One of the most widespread species across northern Australia is the sooty grunter (*Hephaestus fuliginosus*). Sooty grunters are omnivorous and eat a diverse diet, including terrestrial insects and vegetation, fish, aquatic insect larvae, macrocrustacea (shrimps and prawns) and aquatic vegetation. Sooty grunters switch diet from being insectivorous while juvenile to being top-level predators as adults, often feeding on smaller fish as well as juvenile grunters. Juvenile grunters are often associated with flowing water, suggesting that water harvesting that reduces or ceases flow could pose a threat. Tree root masses and undercut banks are also important microhabitat, especially for adult fish (Pusey et al., 2004).

Grunters prefer medium to high oxygen levels as well as medium to low salinity (Hogan and Nicholson, 1987). Grunters will move out of the dry-season refugial habitats and into ephemeral wet-season habitats for spawning (Bishop et al., 1990), with juveniles known to swim up to 7 km.

The sooty grunter is an important recreational species, and in some of their range environmental flow is managed to maintain suitable habitat conditions (Chan et al., 2012). Because grunters are omnivorous and able to integrate many sources of food, as well as having a high total biomass, they are an important link in the overall food chain. They link lower trophic levels with top-level predators, such as long tom (*Strongylura krefftii*) and crocodiles. Grunters are also important species for Indigenous Peoples in northern Australia, both culturally (Finn and Jackson, 2011; Jackson et al., 2011) and as a food source (Naughton et al., 1986).

Grunters in the Victoria catchment and marine region

The Victoria catchment has a slightly different composition of grunters compared to catchments that drain into the Gulf of Carpentaria. Apart from the widespread spangled grunter (*Leiopotherapon unicolor*) and barred grunter (*Amniataba percoides*), the western sooty grunter (*Hephaestus jenkinsi*) replaces the eastern species *H. fuliginosus.* Less abundant species include the sharpnose grunter (*Syncomistes butleri*)*,* Drysdale grunter (*Syncomistes rastellus*) and Neil's grunter (*Scortum neili*).

Of these grunters, the western sooty grunter is the key species for recreational and cultural purposes (Chan et al., 2012). Grunters are likely widespread in the Victoria River, with headwaters being spawning and nursery grounds for larger species as well as habitat for adults of the smaller species (e.g. spangled grunter). Waterholes on the main stem provide habitat for adult grunters.

Neil's grunter is endemic to the Victoria catchment and is listed as Endangered on the IUCN Red List of Threatened Species. Adults occur in small, well-shaded, slow-flowing streams with mixed sand, silt and rock bottoms, and also in deeper rocky pools in gorges. Preferred water conditions are typically fresh and clear, between 21 and 28 °C, with a neutral or slightly basic pH. Occurrences of grunter species in the Victoria catchment can be seen in Figure 3-6. Modelled distribution of *Leiopotherapon unicolor* is shown in Figure 3-7.

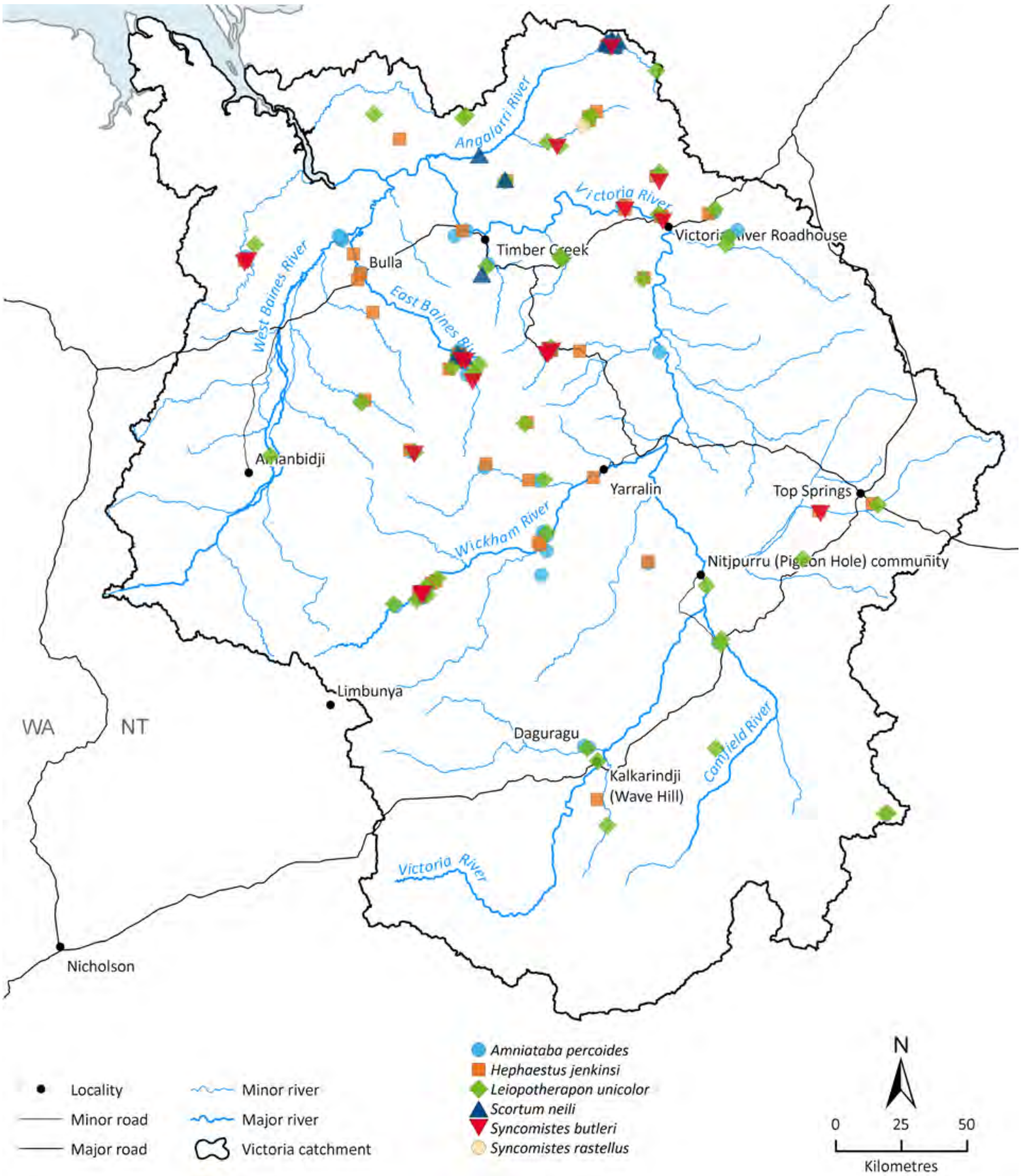

Figure 3-6 Observed locations of grunters in the Victoria catchment

Data source: Atlas of Living Australia (2023)

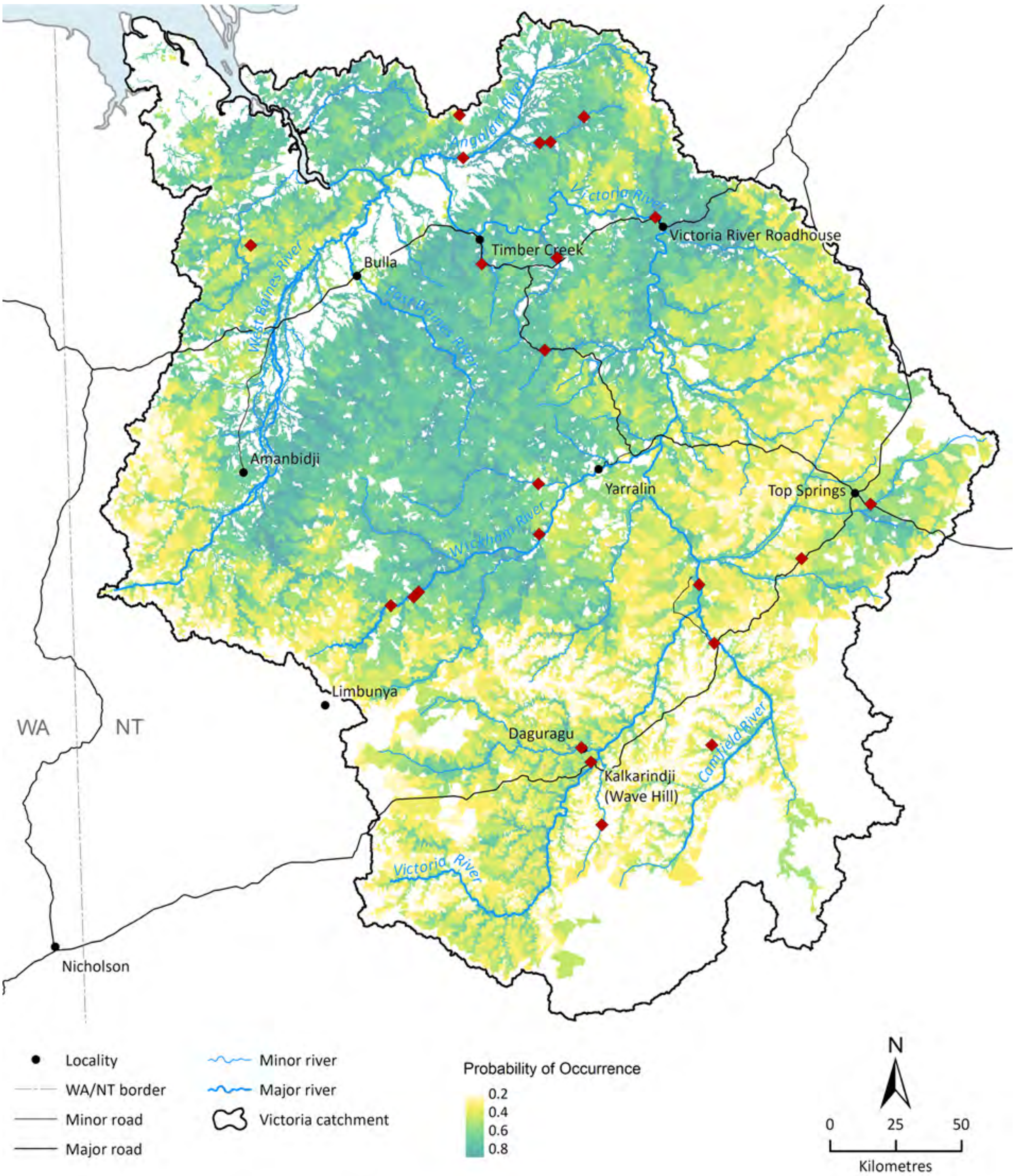

Figure 3-7 Modelled probability of occurrence of spangled grunter (*Leiopotherapon unicolor*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for grunters

Terapontidae have varying flow requirements. Not many studies have investigated the western sooty grunter *H. jenkinsi*; however, the very close relative *H. fuliginosus* is likely to have very similar preferences (Allen et al., 2002). It cannot be assumed that knowledge of the flow requirements of a charismatic taxon such as barramundi can be a surrogate for understanding the flow requirements of all life stages of sooty grunters.

The most important northern Australian species for recreational and cultural reasons, *H. fuliginosus*, is found in a variety of habitats, between headwater streams and the river mouths of the larger northern Australian streams; adults are rheophilic (i.e. have a preference for flowing water) (McDowall, 1996). In some tropical streams in the Cape York Peninsula, passage to spawning habitat has been reported as a requirement (Herbert et al., 1995).

There is scientific consensus that altered flow regimes are of concern to *H. fuliginosus* populations. It is the most rheophilic grunter species and is highly adapted to flowing water conditions (Pusey et al., 2004). Impoundments can inundate upstream riffles and fast-flowing sections that provide critical spawning areas. In general, regulation can both dry out critical habitat and connections and drown shallow refuge habitats. This can reduce riffles and runs, and reducing the diversity of flow environments within reaches is likely to reduce spawning success (Harris and Gehrke, 1994). While too much water can reduce population fitness, the loss of medium-magnitude flushing flows in the wet season would affect spawning sites (Hogan, 1994). Medium-size flow events also stimulate secondary production, and their loss could lead to a lack of food for grunter populations. Movement in both directions must be possible in order to accommodate the needs of both adult and juvenile grunters.

Leiopotherapon unicolor and other smaller-bodied grunters have additional requirements to hydraulic habitat. *Leiopotherapon unicolor* needs a relatively high spawning water temperature of 20 to 26 °C (Allen et al., 2002). This can be compromised by hypolimnetic releases from impoundments (i.e. releases of the cold bottom layer of water). Movement is also key for *L. unicolor*'s life cycle. As a smaller species than the two *Hephaestus* species, *L. unicolor* often prefers to use floodplain wetlands as nursery habitat, making intermittent flooding important for recruitment success (Merrick and Schmida, 1984).

Amniataba percoides is a highly adaptable species, but data on its life-history are scarce. Pusey et al. (2004) postulated the fish also needs a balanced flow regime that is close to the natural regime on the basis of the following observations:

- *Amniataba percoides* moves during its life cycle, hence impoundments are a threatening process (Bishop et al., 1995).
- *Amniataba percoides* shows a preference for flowing water, but high flows are likely to reduce population sizes.
- While *Amniataba percoides* is not dependent on floodplain spawning like some other fish taxa, floodplain connections increase population fitness.
- *Amniataba percoides* needs high spawning temperatures, and desynchronisation of flow and thermal regimes by impoundments can reduce their fitness.

The ecological functions that support grunters, and their associated flow requirements, are summarised in Table 3-3.

Table 3-3 Ecological functions supporting grunters and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintain riverine waterhole persistence as freshwater habitat during the dry season	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintain natural ephemerality vs conversion to perennial systems	Dry-season duration, groundwater discharge and intensity of evaporation	Low flows
Maintain water quality in riverine waterholes as freshwater habitat during the dry season	Baseflows into waterholes	Low flows
Maintain palustrine waterhole persistence as habitat during the dry season and reconnection to support migration during the wet season	Floodplain inundation extent and duration during the wet season	High flows
Riverine waterhole flush and reconnection, as well as establishing brackish estuarine conditions in the late dry season	Late dry-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue spawning migration to upstream and palustrine habitats, their growth and survival in these habitats	Moderate flood flows. Higher fish growth and survival during wet-season high-flood flows	Flow timing and magnitude. Seasonality of flows

Pathways to change for grunters

The possible ecological outcomes of threatening processes on grunters in northern Australia are discussed in this section and presented in the conceptual model (Figure 3-8). Four of the key threats in the conceptual model are related to flow modification: water harvesting, dam infrastructure, river regulation and climate change. For *A. percoides*, changes in flow regimes that lead to faster-flowing environments can lead to decreased population viability – for example, a dam structure that first holds back water then releases it at higher velocity (Pusey et al., 2004). The key mechanisms for this are desynchronisation of thermal regimes and juvenile mortality caused by out-of-season high flows.

The impact of regulation on *L. unicolor* has been documented by Gehrke (1997), who found that abundance was greatly reduced in regulated reaches. This is partly attributable to barriers to mobility, but also to a change in sediment composition, which leads to habitat alteration. Similarly, *H. fuliginosus* relies on flowing water, especially for spawning runs (Hogan, 1994). Barriers can interrupt these runs, leading to lower population viability. The loss of flushing flows can also lead to sediment build up in key pool habitats – this effect is exacerbated by land use change.

Climate change will interact with these threats in two ways: (i) it will enhance high flows that reduce populations, and (ii) droughts will interact with flow regimes, including increased impacts of water extraction. The ecological outcomes of threatening processes on grunters in northern Australia, and their implications for changes to habitat, community structure and population size, are synthesised in the conceptual model shown in Figure 3-8.

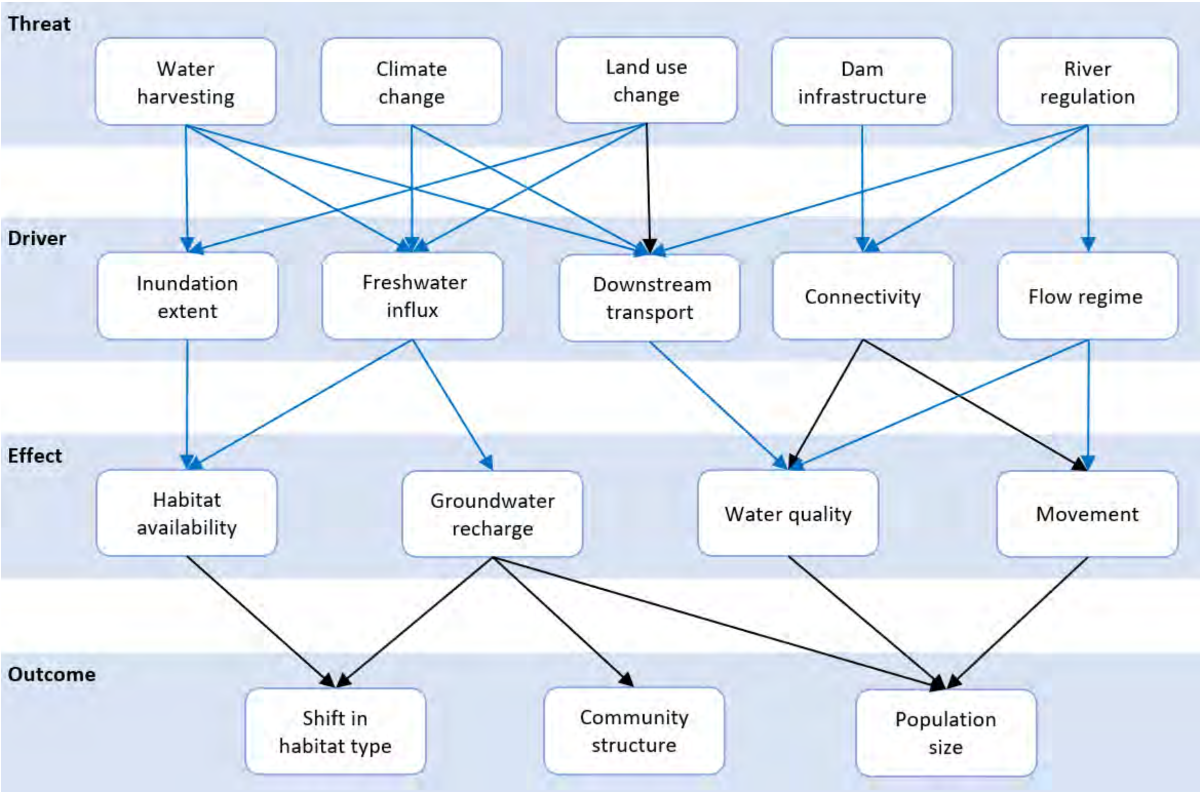

Figure 3-8 Conceptual model showing the relationship between threats, drivers, effects and outcomes for grunters in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.4 Mullet (family Mugilidae)

Description and background to ecology

Mullet (a guild including the genera *Liza*, *Mugil* and *Moolgarda*) are fish that use marine habitats as adults to spawn and freshwater habitats as juveniles (i.e. catadromous). They have life histories that entail ‘catchment to coast’ habitats (i.e. freshwater, estuarine and marine habitats) (Marin et al., 2003; Whitfield et al., 2012). Mullet are distributed in tropical and temperate coastal waters worldwide. About 20 tropical mullet species occur in northern Australian waters from Townsville on the east coast to Broome in the west (Blaber et al., 2010). Diamond-scale mullet (*Liza vaigiensis*), largescale mullet (*Liza macrolepis*), greenback mullet (*Liza subviridis*), sea mullet (*Mugil cephalus*), roundhead mullet (*Moolgarda cunnesius*), bluespot mullet (*Moolgarda seheli*) and bluetail mullet (*Moolgarda buchanani*) are common species in the Australian tropics and range across the Indo-Pacific (Larson et al., 2013; Whitfield et al., 2012).

These catadromous species are an abundant component of the fish community, being both forager and prey in the coastal ecosystem. Larvae are planktivorous, and juveniles feed on benthic invertebrates as well as prey in the water column. Adult mullet feed on organic detritus, benthic microalgae, filamentous algae, meiofauna (i.e. tiny sediment-dwelling invertebrates) and small invertebrates (Górski et al., 2015; Soyinka, 2008; Whitfield et al., 2012). Being themselves preyed upon by larger species, mullet transfer energy from low to high trophic levels in the estuarine fish community; mullet are thus ecological link species (Górski et al., 2015). Their position as detritivores in the food chain and their fast growth rates and high fecundity make them a species group with high harvest potential.

Mullet tend to grow fastest during the summer or tropical wet season, suggesting the influence of a seasonal increase in productivity of coastal waters (Grant and Spain, 1975; Whitfield et al., 2012). By about 4 years of age, they leave nursery habitats for lower estuaries and the ocean. In general, mullets in Australia aggregate and spawn in marine waters in the lower reaches of estuaries or adjacent coastal waters in autumn to mid-winter before moving into coastal open-water habitats (De Silva, 1980; Grant and Spain, 1975; Kailola et al., 1993; Robins et al., 2005).

Short-lived, fast-growing and productive, mullet are important as a commercial, recreational and Indigenous fish resource. Mullet are one of the most important species groups taken in NT recreational catches and the third most prominent species in (non-Indigenous) recreational catches in the east coast (Gulf of Carpentaria area) of the NT (West et al., 2012). Most of the NT recreational mullet catches (92.4%) are targeted (West et al., 2012) rather than bycatch. Mullet are of cultural significance for Indigenous communities throughout Australia and among the most numerous species in their catch (Henry and Lyle, 2003). In NT fisheries, they are a target for Aboriginal coastal fishing licences (Boyer, 2018; Wilton et al., 2018) and a target or bycatch in several fisheries (Northern Territory Government, 2022).

Mullet in the Victoria catchment and marine region

Records of the presence of mullet in the Victoria catchment and marine region are poor. Water samples for eDNA analyses were collected during this study. Mullet DNA (*Planiliza ordensis*, river diamond mullet) was detected in the lower Victoria River in the vicinity of the Victoria River Roadhouse and in the East Baines River, a tributary of the Victoria River. In the mid-to-upper Victoria River, mullet DNA was collected at Dashwood Crossing in the vicinity of Victoria River Downs Station and at Nitjpurru (Pigeon Hole) on the upper Victoria about 400 km from the estuary. The Atlas of Living Australia records *Planiliza ordensis* (river diamond mullet), *Moolgarda buchanani* (bluetail mullet) and *Moolgarda seheli* (bluespot mullet) as present in the Victoria River. Occurrences of mullet species in the Victoria catchment can be seen in Figure 3-9.

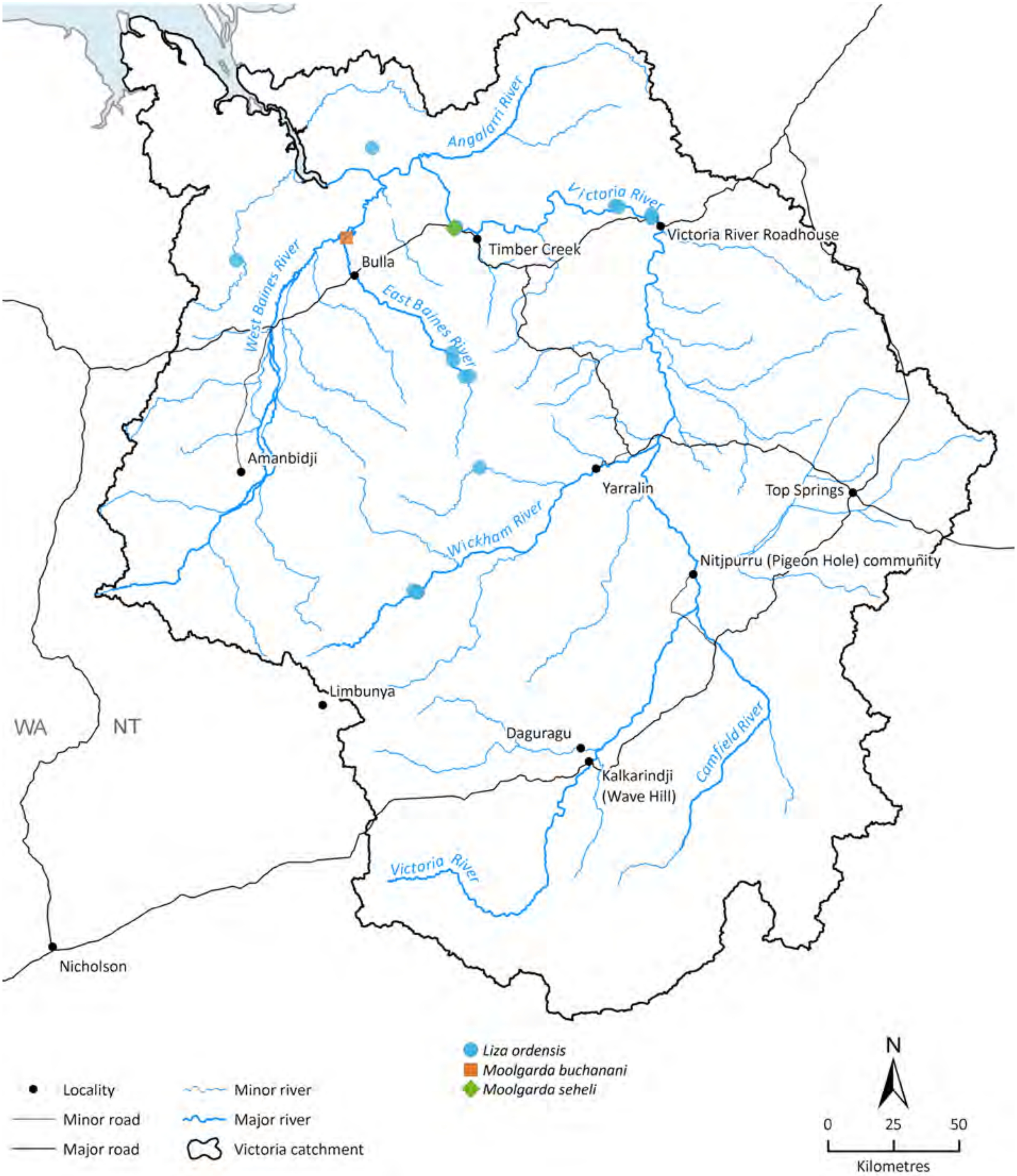

Figure 3-9 Observed locations of mullet in the Victoria catchment

Mullet juveniles use the mangrove and mudbank habitats within the estuary and adult fish are caught within the estuary and in shallow subtidal habitats in the littoral zone. Mullet may use brackish and freshwater habitats during their juvenile phase.

Data source: Atlas of Living Australia (2023)

Flow–ecology relationships for mullet

Mullet spawn in coastal marine areas where the larvae inhabit marine-salinity waters. As they grow, juvenile mullet migrate into estuaries and upstream to freshwater habitats (including palustrine wetlands) (Blaber et al., 1995; Gillson et al., 2009; Rolls et al., 2014). The frequency and duration of high-flood events supports the inundation and availability of river floodplain and estuarine supra-littoral habitats used extensively by juvenile mullet during the wet season (O'Mara et al., 2021). Flooded palustrine habitats are hot spots for primary productivity (Burford et al., 2016; Ndehedehe et al., 2020a; Ndehedehe et al., 2020b) and refugia during the subsequent dry season (O'Mara et al., 2021). Reduced river flow volume and disrupted seasonality of flows affect mullet negatively by reducing the extent and connectivity of estuarine and freshwater habitats, affecting growth and survival via lower seasonal food accessibility and non-optimal environmental conditions (Jardine et al., 2013; Ndehedehe et al., 2021; Ndehedehe et al., 2020b) (Table 3-4).

Dry-season baseflows facilitate connectivity between estuarine and riverine reaches. Brackish water and freshwater habitats are optimal for the growth and survival of mullet (Cardona, 2000; Whitfield et al., 2012) and lost connectivity reduces the population. Monsoon-season flood flows support the upstream and downstream migration of juvenile and adult mullet, respectively (Table 3-4). High-level flows allow access to inundated floodplain habitats for juvenile mullet and cue emigration of sub-adults and adults to the marine environment.

Constructed barriers such a weirs or dams block access to up-river habitats for juvenile mullet. Mullet are found in the riverine reaches of rivers in the Victoria catchment, so their catadromous life history remains critical to ontogenetic habitat selection over the full extent of catchment to coast (Larson et al., 2013; Waltham et al., 2013).

The ecological functions that support mullet, and their associated flow requirements, are summarised in Table 3-4.

Table 3-4 Ecological functions supporting mullet and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of riverine waterhole persistence as mullet freshwater habitat during the dry season	Dry-season duration and intensity	Low flows
Maintenance of natural ephemerality vs conversion to perennial systems	Dry-season duration and intensity	Low flows
Maintenance of water quality in riverine waterholes as mullet freshwater habitat during the dry season	Baseflows into waterholes	Low flows
Maintenance of palustrine waterhole persistence as mullet habitat during the dry season and reconnection to support migration during the wet season	Floodplain inundation extent and duration during the wet season	High flows
Riverine waterhole flush and reconnection, as well as establishing brackish estuarine conditions in the late dry season	Late dry-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue mullet immigration to riverine and palustrine habitats, their growth and survival in these habitats, and emigration to marine habitats	Moderate flood flow. Higher mullet growth and survival during wet-season high-flood flows	High flows. Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that cue mullet emigration from freshwater to marine habitats, and palustrine inundation and extensive, ephemeral mullet habitat during the wet season	Large flood flows to inundate the floodplain and maintain palustrine wetlands and productivity hotspots during the wet season; high-level mullet growth and superior survival	High flows, their frequency and seasonal reoccurrence

Pathways to change for mullet

With marine and freshwater habitat use similar to barramundi, juvenile and early-adult phase mullet prefer fresh and brackish waters, including palustrine wetlands, that support optimal growth and survival (Cardona, 2000; Whitfield et al., 2012). Seasonal rainfall and flow likely influence downstream movements (Cardona, 2000; Gillson et al., 2009). A reduction in flow volume and seasonality may negatively affect mullet populations by reducing the extent and connectivity of the estuarine and freshwater habitats (Faggotter et al., 2013; Jardine et al., 2013; O'Mara et al., 2021) and disrupting cues for spawning movements. Disrupted connectivity by built barriers may limit use of freshwater habitats (Grant and Spain, 1975; O'Mara et al., 2021; Robins and Ye, 2007; Stuart and Mallen-Cooper, 1999). Wetland 'perimeter to area ratio' and wetland 'number of patches' can be strongly related to mullet catch, suggesting the extent and connectivity of estuarine habitats, intertidal and supra-littoral areas, and creeks and channels are important to mullet production (Meynecke et al., 2008). However, some individuals occupy wholly marine habitats despite available access to nearby estuaries (Górski et al., 2015). The ecological outcomes of threatening processes on mullet in northern Australia, and their implications for changes to growth and mortality, community structure, habitat and population, are illustrated in Figure 3-10.

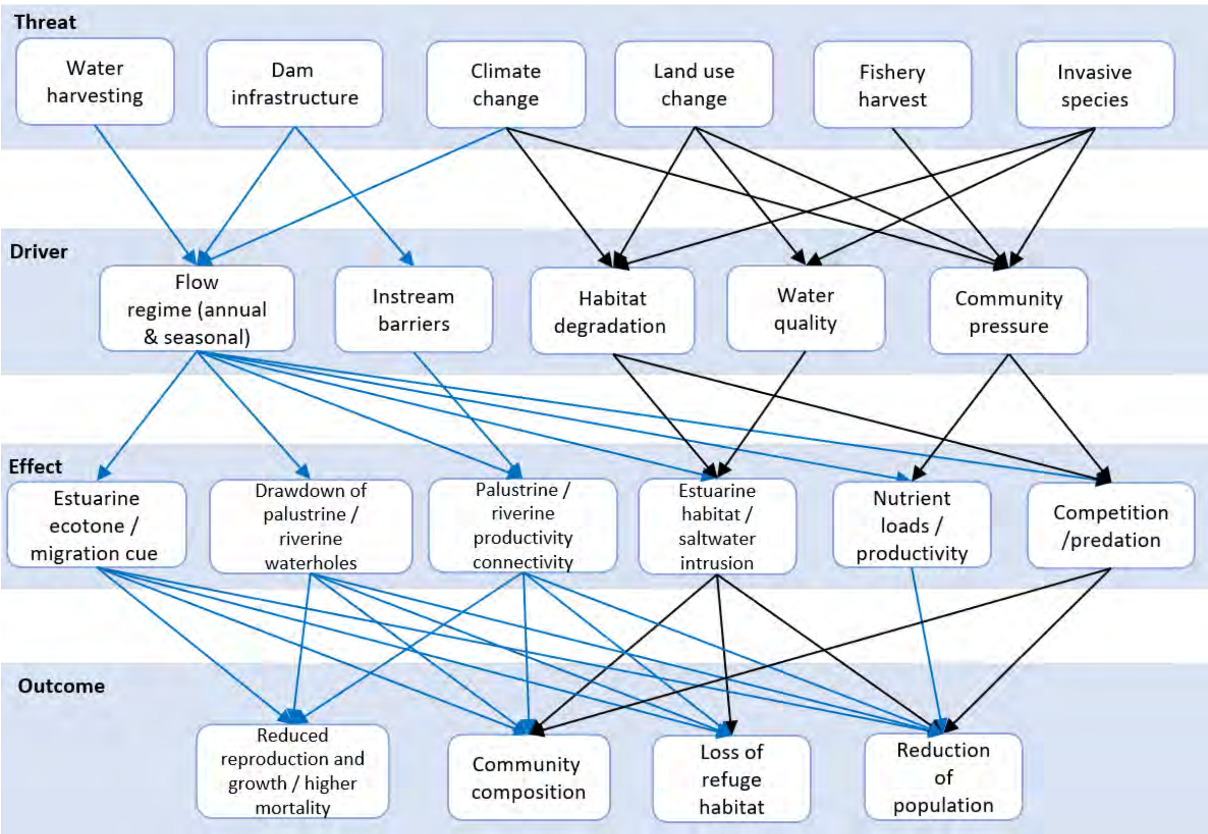

Figure 3-10 Conceptual model showing the relationship between threats, drivers, effects and outcomes for mullet in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.5 River sharks (*Glyphis* spp.)

Description and background to ecology

River sharks is the generic term given to species of the genus *Glyphis*, found in the Indo-West Pacific, each of which is Endangered or Critically Endangered (Last and Stevens, 2008; Morgan, 2011; Stevens et al., 2009). Two *Glyphis* species are found in Australian waters, the speartooth shark (*Glyphis glyphis*; Critically Endangered, EPBC Act and IUCN) and the northern river shark (*Glyphis garricki*; Endangered, EPBC Act and IUCN). The speartooth shark occurs across Cape York, the north-west coast of the Top End, inshore Joseph Bonaparte Gulf and the southern coast of Papua New Guinea (Pillans et al., 2009; White et al., 2015). The northern river shark occurs across the Kimberley and the Top End coast, as well as the Fly River, Papua New Guinea (Pillans et al., 2009; West et al., 2021; White et al., 2015). Tropical Australia and Papua New Guinea probably represent the last viable populations of the speartooth shark and the northern river shark across their global ranges (Pillans, 2014; Pillans et al., 2022).

River sharks are poorly studied, though studies of their population structure, niche partitioning, and estuarine habitat and prey have been undertaken in the past 5 years (Dwyer et al., 2019; Every et al., 2019; Feutry et al., 2020).

River sharks are of significance to the Indigenous Peoples using the freshwater and estuarine resources of the tropical rivers. Comments from elders may reflect a seasonal appearance of sharks in the freshwater reaches of the tropical rivers, which along with other species were food sources (Barber, 2013).

River sharks in the Victoria catchment and marine region

Within large tropical river systems, the speartooth shark uses the mangrove-fringed upstream portions of the estuary and the riverine habitats where the estuary blends to become the river as its primary habitat (indicative habitat salinity 1 to 28 parts per thousand (ppt) in the NT and 3 to 26 ppt in western Cape York, Queensland) (Dwyer et al., 2019; Pillans, 2014; Pillans et al., 2009). It has an ontogenetic shift in habitat preference: juveniles use the upper-estuarine and lower-freshwater reaches of rivers (up to 100 km upstream) and adults use estuarine environments (Pillans et al., 2009).

Knowledge of the reproduction of the speartooth shark is patchy; however, young are probably born from October to December in the lower estuaries or near the mouths of rivers. Mature sharks prefer highly turbid, tidal waters over fine muddy sediments. They move up and down the estuary with the flood and ebb tides (Pillans et al., 2009). Adults (likely >200 cm) are found in the lower marine-influenced estuary; none have been recorded outside rivers in marine zones (Field et al., 2013; Pillans et al., 2009). Coincident with annual monsoon floods, juvenile speartooth sharks move down-estuary within brackish conditions as their upper-estuary habitats become freshwater, and they return up-estuary during the dry season (Dwyer et al., 2019; Pillans et al., 2022).

The northern river shark has been found in Cambridge Gulf and the Daly River, respectively west and east of the Victoria River. The species uses estuarine and freshwater habitats, but is more marine in habit than the speartooth shark. The northern river shark uses rivers (salinity 2 ppt), large tropical estuarine systems (salinity 7–21 ppt), macrotidal embayments and inshore and offshore marine habitats (salinity 32–36 ppt) (Pillans et al., 2009). It is thought adults use only marine environments and may be found outside estuaries. The northern river shark likely pups prior to the annual wet season with a litter size around nine. Neonates and juveniles are found in freshwater, estuarine and marine habitats, though capture locations indicate a preference for highly turbid, tidally influenced waters over muddy substrate (Stevens et al., 2005). No *Glyphis* species have been found in isolated freshwater habitats such as billabongs or refuge waterholes in river channels (Stevens et al., 2005).

Published data on the distribution of river sharks in the Victoria River are scant. Records for northern river shark exist for Cambridge Gulf and Daly River. No published records of speartooth shark exist for regions in the Joseph Bonaparte Gulf littoral or estuarine habitats. However, Dr Richard Pillans (CSIRO Environment, Brisbane), 2022, pers. comm.) conducted surveys of freshwater elasmobranchs in the Victoria River in 2018 and 2019 and recorded both speartooth and northern river sharks in brackish-water reaches of the river (Figure 3-11, Dr Richard Pillans, CSIRO Environment, Brisbane, 2022, pers. comm.)). The presence of river sharks in the ~2 to 20 ppt ecotone as critical juvenile habitat suggests they are highly dependent on the estuarine-riverine interface of large perennial river systems where extensive sections of the river ecotone are present. These habitats are not common in many northern tropical rivers as hypersaline conditions exist in the upper estuary due to the cessation of flows during the dry season. These surveys were conducted as part of the Ord River Offset program, which inventories natural resources in the vicinity of expanded Ord River irrigation agriculture.

Dr Pillans caught three speartooth sharks and eight northern river sharks in the Victoria River upper estuary, from about 80 to 120 km upstream from Entrance Island. These new records of the presence of the two species in the Victoria River exemplify the paucity of biological data from remote tropical Australia.

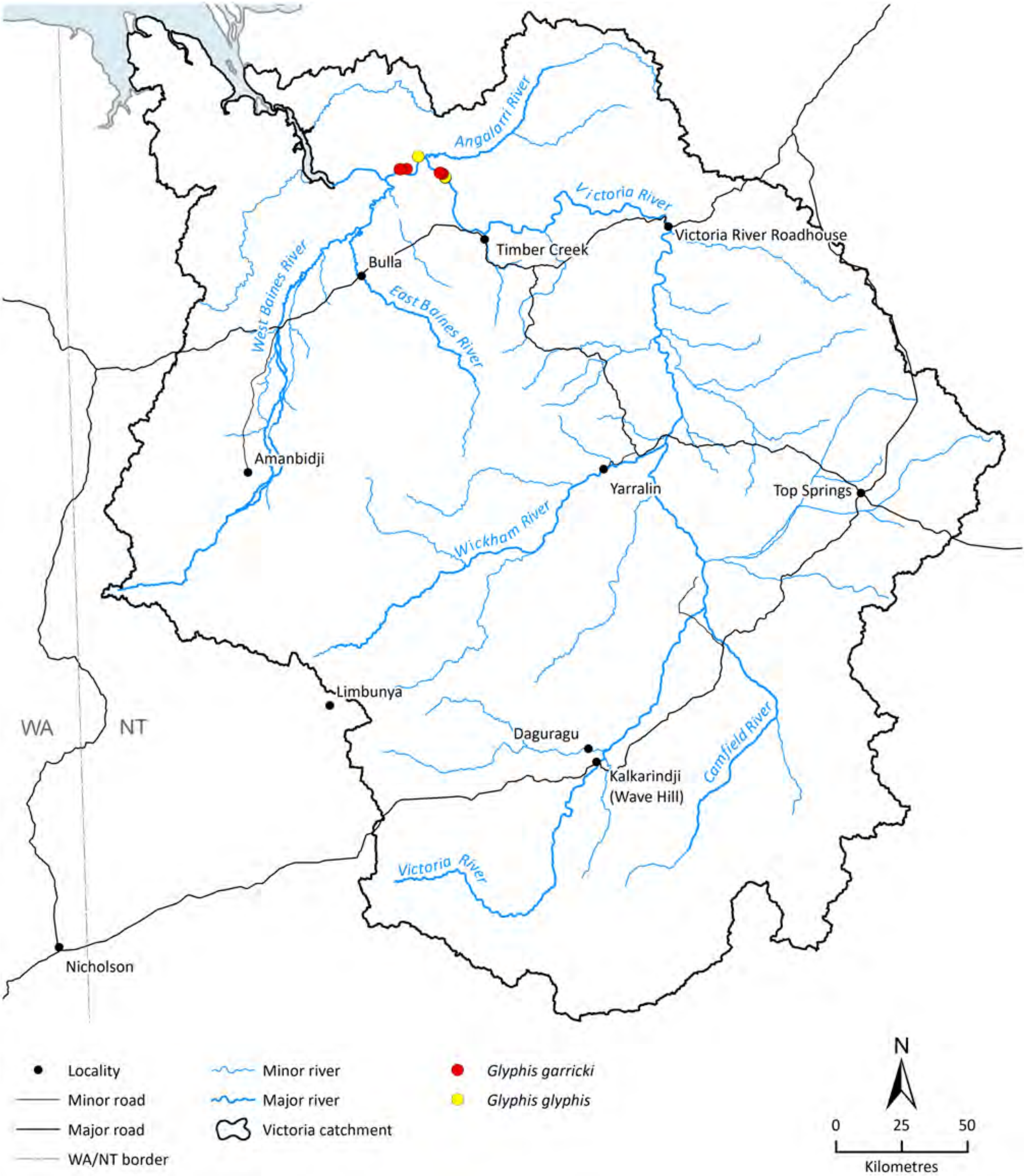

Figure 3-11 Observed locations of river sharks (*Glyphis* spp.) in the Victoria catchment

Data sources: Dr Richard Pillans, CSIRO; Pillans et al. (2022) and the Atlas of Living Australia (2023)

Flow–ecology relationships for river sharks

Both speartooth sharks and northern river sharks use brackish, turbid habitats within the upper estuary as habitat. Juvenile sharks are confined to upstream estuarine reaches of tropical rivers, and wet-season downstream movement is related to increases in freshwater flows. Adult sharks use the marine lower reaches of estuaries, as well as coastal embayments and inshore habitats. Northern river sharks are more coastal marine as adults than speartooth sharks. Both *Glyphis* species show female philopatry (i.e. return to their birthplace to reproduce), resulting in genetic isolation between rivers (Morgan et al., 2016; Thorburn, 2007; Whitty, 2017; 2009). The density of speartooth sharks is highest in the seasonally variable salinity of the brackish portion of the estuary, and their high-density distribution shifts downstream during the wet season when salinity reduces in the upper estuary (Pillans et al., 2022).

River sharks are slow growing and long lived, and there is strong evidence for population subdivision at the estuary level in both these endangered species (Feutry et al., 2017; Feutry et al., 2020; Feutry et al., 2014). Consequently, any reduction or change in seasonality of flow in any one river is likely to significantly reduce the population size and dynamics of these Endangered and Critically Endangered species (Table 3-5). Lack of gene-mixing between estuaries or estuary groups renders population stability for these species at risk given current pressures on tropical coasts (e.g. from fishing) (Pillans et al., 2022).

The ecological functions that support river sharks, and their associated flow requirements, are summarised in Table 3-5.

Table 3-5 Ecological functions supporting sawfishes and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of brackish water conditions in estuaries during the dry season that is critical habitat for juvenile river sharks	Dry-season duration and intensity	Low flows
Persistence of brackish conditions in estuaries in the early dry season that support river shark habitat and foraging (from April onwards)	Early dry-season persistent low-level flow	Low flows
Provision of brackish estuarine conditions in the late dry season that support river shark growth and survival and enhance critical habitat (prior to January wet season)	Late dry-season first low-level flow	Low flows
Early wet-season flood flows that increase the spatial extent and downstream alignment of the estuarine ecotone to support river shark growth and survival and downstream migration to forage	Early wet-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue juvenile river shark migration down-estuary	Moderate flood flows	Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that create shallow freshwater habitat conditions in estuaries and floodplains; possibly poor habitat for speartooth and northern river sharks	Large flood flows that cause freshwater estuarine habitats and scour estuaries	High flows, their frequency and seasonal reoccurrence

Pathways to change for river sharks

The interruption of wet-season low flows and the reduction of both low- and high-flood flows by water diversion or impoundment will reduce the ability of speartooth sharks and northern river sharks to access brackish-water ecotone habitats. Juvenile speartooth sharks access freshwater habitats in the lower river reaches, while northern river sharks forage for freshwater prey. Reduced river flows would reduce the spatial and temporal extent of the estuarine ecotone. Physical barriers to low-level, dry-season flows (e.g. instream dams and barrages) may render the estuary hypersaline, prohibit connectivity with freshwater riverine habitats and expose rare species to high levels of predation (possibly by crocodiles) as they accumulate below the barrier attempting to move upstream (Morgan et al., 2017b).

Barriers and flow reduction would affect the habitat extent and movement of river sharks within the estuary. The reduction of both low- and high-flood flows would reduce prey availability for these river sharks. For example, river sharks consume river prawns (*Macrobrachium equidens*), *Johnius novaeguineae*, an estuarine fish species, and barramundi (freshwater resident individuals described by Every et al. (2019)).

River sharks are vulnerable to fishing mostly as non-target catch by both recreational and commercial fishers (Kyne and Feutry, 2017). Speartooth sharks are at significant risk as bycatch in crab pots set in estuaries by both commercial and recreational fishers (Pillans et al., 2022). They enter the pots by the funnel entry to access bait and can't get out due to the funnel arrangement. In a tropical Queensland river, shark habitat use overlapped the distribution of commercial and recreational mud crab fishing effort during both the dry and wet fishing seasons. Seasonally, both the distributions of speartooth sharks and commercial crab pots moved up and down tropical estuaries with the salinity profile of the estuary. As part of a scientific survey, speartooth sharks were caught during the dry season as bycatch in pots set in brackish estuarine reaches. However, no speartooth sharks were caught during the wet season, despite sharks and fishing effort moving down the estuary following the brackish ecotone (Pillans et al., 2022). Two reasons were suggested for the low catch during the wet season: loss of appetite due to stress associated with an abrupt salinity decline, and movement of sharks out of the estuary to the shallow coastal zone (Pillans et al., 2022). In addition, speartooth sharks were vulnerable to commercial and recreational catch in the late wet to early dry season (April and May) when estuarine salinity increased due to freshwater flows trailing off and the brackish ecotone moving up-estuary to overlap with fisher access.

Reduced high-level flows during the wet season would cause the estuary to remain brackish, and sharks would remain within the estuary and exposed to risk as bycatch in crab pots. Reduced levels of salinity due to less freshwater influx would reduce stress on sharks and modify appetite loss, possibly causing sharks to enter crab pots in response to baits. Reduced freshwater flows after the main pulse of the wet season would allow saline influences within the estuary earlier than under a natural flow regime. The brackish ecotone would move upstream earlier, exposing sharks to recreational and commercial fishing effort earlier in the dry season (Pillans et al., 2022). The ecological outcomes of threatening processes on river sharks in northern Australia, and their implications for changes to growth and mortality, community structure, habitat and population, are illustrated in Figure 3-12.

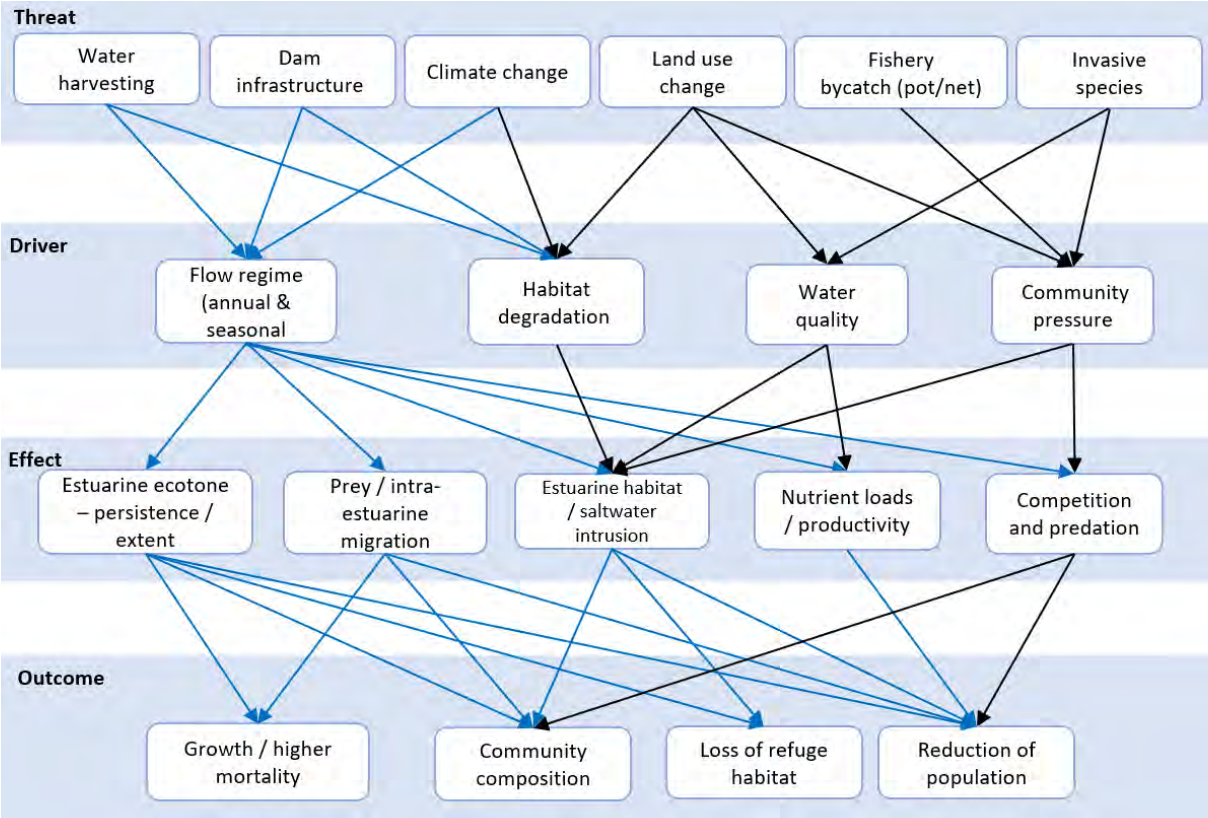

Figure 3-12 Conceptual model showing the relationship between threats, drivers, effects and outcomes for river sharks (*Glyphis* spp.) in large rivers in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.6 Sawfishes (*Pristis* spp.)

Description and background to ecology

Sawfishes are a type of ray from the order Pristiformes. They are characterised by a tooth-lined rostrum, commonly referred to as a 'saw'. As adults, sawfish can reach substantial sizes, ranging from 5 to 7 m in total length. They are widely distributed in the marine waters of northern Australia, although they are not necessarily abundant (Last and Stevens, 2008; Morgan, 2011; Stevens et al., 2009). These species can migrate at landscape and oceanic scales through their life cycle, with inshore waters, including bays and estuaries, serving as crucial nursery grounds for neonates and juvenile sawfishes until about 4 to 6 years of age (Morgan, 2011; Morgan et al., 2017a; Peverell, 2005). As adults, they predominantly inhabit tropical and subtropical coastal marine waters (Dulvy, 2016; Last and Stevens, 2008).

Globally, sawfishes rank among the most threatened marine taxa (Dulvy, 2016). In Australian waters there are four species of sawfishes, all listed of conservation significance at national and international level. The freshwater or large tooth sawfish (*Pristis pristis*), the green sawfish (*P. zijsron*) and the dwarf sawfish (*P. clavata*) are listed as Vulnerable under the EPBC Act. The narrow sawfish (*Anoxypristis cuspidata*) is listed as Endangered by the IUCN. It is listed as Migratory under the EPBC Act, but because it is also listed in Appendix I and II under the Convention on the Conservation of Migratory Species of Wild Animals (Bonn Convention), it has similar protection status under the EPBC Act. Moreover, sawfishes are of significant cultural and spiritual importance for Indigenous Australians (Ebner et al., 2016).

Sawfishes face multiple threats, partly due to their morphology (the shape of their rostra) and behaviour, and partly due to their life-history characteristics: long lives, slow growth and low reproductive rates, late maturation, relatively low abundance and high habitat specificity during different life stages (Peverell, 2005; Phillips, 2017; Stevens et al., 2009). Given the overlap of sawfish habitats with coastal fisheries and their susceptibility to be captured in gill-net and trawl fisheries, as well as in recreational fishing, they are at a high risk of negative impacts.

Sawfish rostrums have been collected as trophy items for decades (McDavitt, 1996), and there is a growing demand for live sawfish for display in public aquaria (Buckley et al., 2020; Compagno et al., 2006). Fishing mortality over recent decades has been high (Fry, 2021). Other pressures include the cumulative impacts from climate change, habitat loss, artificial passage barriers and declining water quality that may have a significant impact on the movements of sawfish between freshwater and estuarine environments.

Sawfishes in the Victoria catchment and marine region

Published data on the distribution of sawfishes in the Victoria River are scarce (Figure 3-13). Records for freshwater sawfish exist for the Ord River, Keep River and Daly River, catchments adjacent to the west and east of the Victoria River. Four records for freshwater sawfish in the Victoria River were found in the Ocean Biodiversity Information System (OBIS, 2022). However, Dr Richard Pillans conducted surveys for freshwater elasmobranchs in the Victoria River in 2018 and 2019 and recorded both freshwater and dwarf sawfish in the river's reaches (Figure 3-13, Dr Richard Pillans, CSIRO Environment, Brisbane, 2022, pers. comm.). These surveys were part of the Ord River Offset program, which inventories natural resources in the vicinity of expanded Ord River irrigation agriculture (Smolinski et al., 2010; Smolinski et al., 2015). Dr Pillans caught 28 freshwater sawfish and 29 dwarf sawfish in the freshwater reaches of the Victoria River. Freshwater sawfish were recorded throughout the freshwater river to about 400 km upstream from the river mouth. They were caught halfway between Timber Creek and Gregory, as well as upstream of Nitjpurru (Pigeon Hole). Dwarf sawfish were recorded from above the confluence with the Baines River to the vicinity of Timber Creek, about 120 km upstream from Entrance Island (in the mid-estuary). New and more extensive records of the presence of the two species in the Victoria River typifies the low level of biological inventory of remote tropical Australia (Figure 3-13).

In addition to estuarine catches of sawfish, four species have been taken as bycatch during prawn trawling in waters about 60 to 80 m deep in the Joseph Bonaparte Gulf. Narrow sawfish were commonly caught. Freshwater sawfish, green sawfish and dwarf sawfish were less common (Figure 3-13). The modelled distribution of *Pristis pristis* is shown in Figure 3-14.

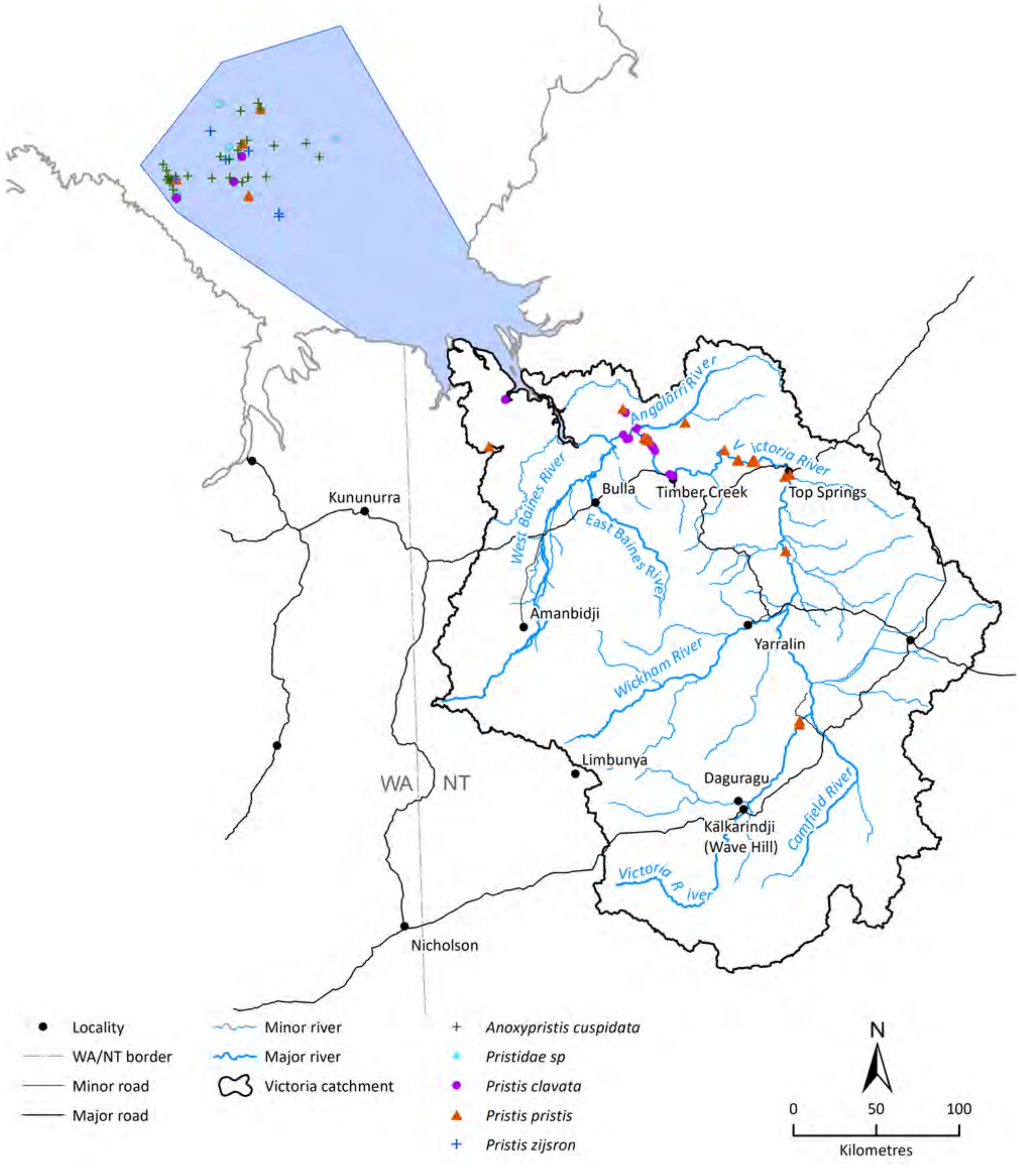

Figure 3-13 Observed locations of sawfishes in the Victoria catchment and the marine region

Data sources: Fry (2021); Kenyon et al. (2022);Atlas of Living Australia (2023); Department of Environment Parks and Water Security (2019a)

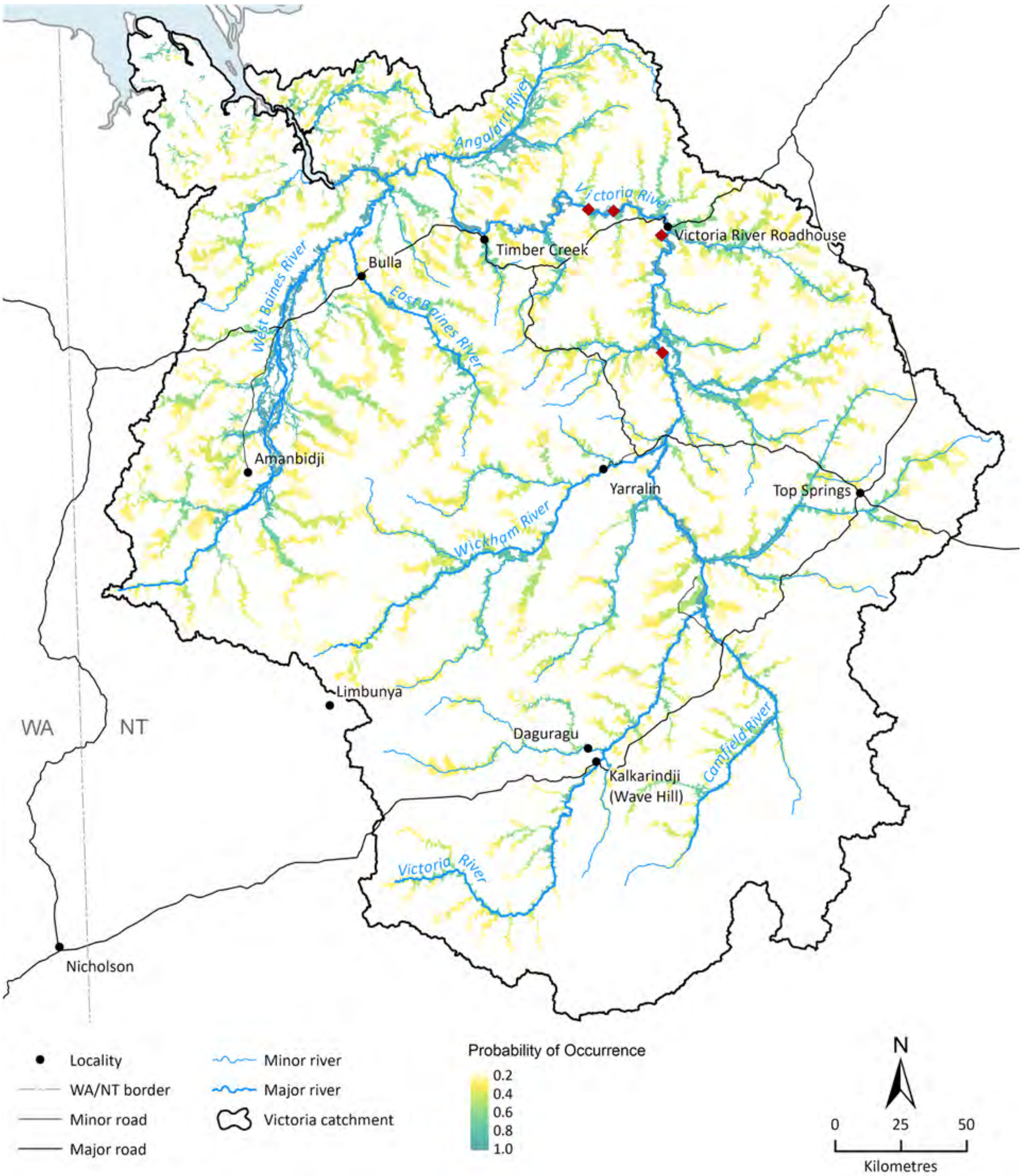

Figure 3-14 Modelled probability of occurrence of freshwater sawfish (*Pristis pristis*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for sawfishes

Sawfish species exhibit known dependencies on estuarine and riverine habitats. Juvenile largetooth sawfish inhabit both estuarine and freshwater environments, using high-flood flows to move upstream to riverine reaches and access freshwater habitats (Morgan et al., 2016; Thorburn, 2007; Whitty, 2017; 2009). They can be found over 400 km upstream in freshwater riverine reaches of large rivers. During their juvenile stage, which extends up to approximately 5 years old, they prefer refuge pools (see Section 3.4.3) during the dry season in the Australian tropics. At maturity, sawfish migrate downstream to estuarine habitats and become vulnerable to inshore gill-net fisheries, particularly in the monsoonal wet season (February to April) (Peverell, 2005). Riverine-estuarine connectivity and long-stream connectivity are critical for largetooth sawfish to access their juvenile habitats and return to estuarine breeding habitats (Table 3-6).

Dwarf sawfish use estuarine habitats and the lowermost riverine reaches seasonally, prompted by salinity changes. In the Fitzroy River of WA, dwarf sawfish were found in a single large high-salinity pool at the uppermost tidal limit in the late dry season (August–November) before migrating downstream to close proximity to the river mouth or in King Sound during the wet and early dry seasons (December–July) (Morgan et al., 2021). Their movements are influenced by freshwater cues, guiding them toward higher salinity waters. Also, the green sawfish exhibits site fidelity within the estuarine and coastal habitat matrix near the mouths of tropical rivers. They move to shallow coastal habitats during low tide and to mangrove creek habitats at high tide (Morgan et al., 2017a). During large-river-flow discharge events they emigrate from the river estuary to coastal habitats. The ecological functions and the flow requirements supporting sawfish are summarised in Table 3-6.

Table 3-6 Ecological functions supporting sawfishes and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT	FLOW COMPONENT OR ATTRIBUTE
Dispersal and migration	Frequency (number of connection events) – number of times threshold flow is met Connectivity (duration of connection and disconnection) – days above and below threshold Depth – days above threshold	Flow timing and magnitude. Seasonality of flows
Dispersal and migration	High-salinity pools	Seasonality of flows
Prey supply	Connectivity (duration of connection and disconnection) – days above and below threshold	Low flows
Habitat availability	Extent – days above threshold – high flows (summer) Connectivity (duration of connection and disconnection) – days above and below threshold Depth – days above threshold	Flow timing and magnitude. Seasonality of flows

Pathways to change for sawfishes

The potential implications of threatening processes for sawfishes in northern Australia are summarised in Figure 3-15. Changes in the depth, extent, duration and timing of flows in river reaches the sawfish inhabit can result in habitat loss and significantly affect the sawfish populations. For example, neonate recruitment may be reduced (Morgan et al., 2016), the potential growth of individuals may be affected (Hunt et al., 2012) and/or reducing abundance and survivorship (Close et al., 2014; Jellyman et al., 2016; Morgan et al., 2016). Changes may also reduce the abundance of prey species that use floodplain wetlands during their life cycle (Novak et al., 2017).

Recent research in the Fitzroy River, WA, has identified critical flow characteristics in Australia's tropical rivers that support sawfish populations. The recruitment and survival of largetooth sawfish within riverine freshwater habitats was critically dependent on large flood flows. Largetooth sawfish recruitment to riverine habitats relied on extended periods of high-level flows (14 or more consecutive days in the 98th percentile of recorded water levels) to facilitate access to the upstream freshwater river reaches that serve as their juvenile habitats (Lear et al., 2019). Importantly, persistent riverine pools are critical refugia for sawfishes during the dry season. Confined to these pools, they lose body condition over the dry season with a greater loss following low-volume wet-season flows compared to high-volume ones (Lear et al., 2021). Additionally, certain rivers are likely to function as stronghold nursery habitats for freshwater sawfish, supporting consistent and high numbers of recruits, hence water resource development on these rivers may affect sawfish habitat to a greater extent than development on other rivers (Lear et al., 2021).

The maintenance of depth and stability of river pools during the dry season is critical to the health of sawfish, and disruptions to natural flows due to water impoundment or extraction has the capacity to affect their survival (Figure 3-15) (Lear et al., 2020). During the early wet season, re-established connectivity downstream to estuarine habitats is also crucial, and modification of early-season flows or low-level flows during a poor wet season may delay or reduce riverine connectivity. Fishing mortality over recent decades has been high (Fry, 2021). Other pressures include the cumulative impacts from climate change, habitat loss, artificial passage barriers and declining water quality that may have a significant impact on the movements of sawfish between freshwater and estuarine environments.

The impact of water resource development such as the construction of dams or water harvest at several levels of extraction, on freshwater sawfish populations has been modelled (Plagányi et al., 2023). An array of water harvest and impoundment scenarios on the Mitchell, Gilbert and Flinders rivers in the eastern and southern Gulf of Carpentaria reduced the biomass of sawfish by approximately 50 to 80% depending on the water resource development scenario. The risk to the sawfish population was assessed as severe and/or extreme for three of four water resource development scenarios and moderate for the remaining one in these modelled catchments (Plagányi et al., 2023). The model outputs included an explicit representation of the dependence of sawfish on the abundance of lower estuarine prey species that also were flow-dependent, hence their decline influenced the decline in the sawfish population. Plagányi et al. (2023) showed that both the construction of dams and the harvest of river flows via pumped water extraction affect aspects of freshwater sawfish life history that limit the resilience of their population. Reduced volume and duration of high-level flows and induced variability in the seasonality and volume of low-level flows due to water resource development affect riverine–estuarine connectivity, habitat suitability, body condition, growth, survival and especially upstream migration of freshwater sawfish (Lear et al., 2019; Lear et al., 2021; Plagányi et al., 2023). The ecological outcomes of threatening processes on sawfish in large rivers in northern Australia, and their implications for changes to growth, population and community structure, are illustrated in Figure 3-15.

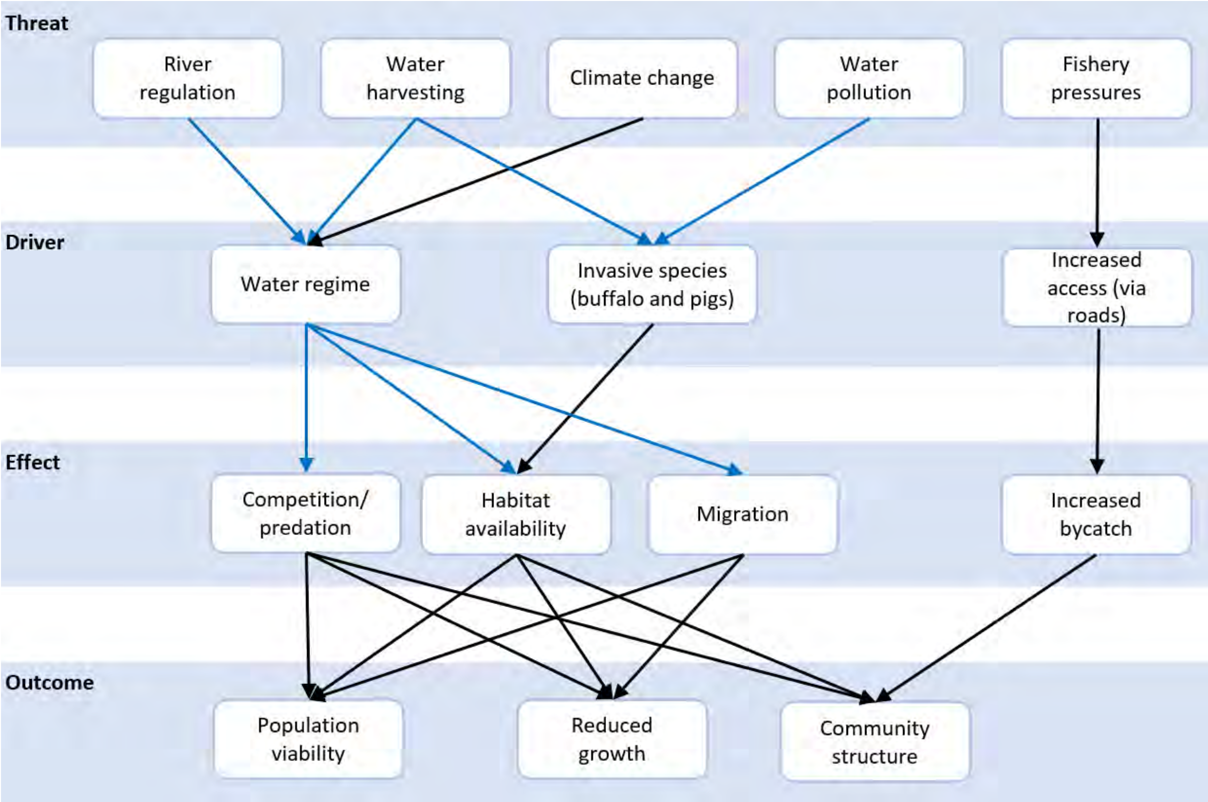

Figure 3-15 Conceptual model showing the relationship between threats, drivers, effects and outcomes for sawfish (*Pristis pristis*) in large rivers in northern Australia

The conceptual model has only been developed for *P. pristis* owing to the lack of information on the other three relevant sawfishes in relation to hydrological change. Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.1.7 Threadfin (*Polydactylus macrochir*)

Description and background to ecology

King threadfin (*Polydactylus macrochir*, formerly *P. sheridani*) is a large (>1.5 m) non-diadromous, carnivorous fish (Order Perciformes). Endemic to Australasia, it is found from the Ashburton River / Exmouth Gulf, WA, across northern Australia, southern Papua New Guinea and Irian Jaya to the Brisbane River in Queensland (Motomura et al., 2000). King threadfin is long-lived (22 years) and fast-growing. Individuals begin life as a male but change to female as they age (protandrous hermaphrodites).

Their body form and quality of flesh makes threadfin a prized table fish. It is typically the second-most important target species in the commercial inshore gill-net fisheries that principally target barramundi (Welch et al., 2010). In 2018–19, 235 t of king threadfin and blue threadfin (*Eleutheronema tetradactylus*) worth $923,000 were taken in the Northern Territory (Steven et al., 2021). Threadfin are also target species for recreational and Indigenous fisheries throughout wet-dry tropical Australia (Moore et al., 2011). King threadfin are of cultural significance for the Indigenous community, and in key localities in the vicinity of Indigenous townships in the NT they are subject to management plans specifying season and bag limits (Malak Malak: Land and Water Management, 2016).

King threadfin complete their entire life cycle in turbid coastal waters, in estuaries, mangrove creeks and inshore marine waters. They tolerate brackish water with a salinity as low as 2 ppt, but are not found in freshwater habitats (Blaber et al., 1995; Moore et al., 2012). Adults probably spawn in inshore coastal waters and lower parts of estuaries (Halliday and Robins, 2005; Welch et al., 2014). High salinity (>32 ppt) is important for survival of the pelagic eggs, and spawning occurs in marine waters away from the outflows of river mouths, avoiding lower salinity levels (Halliday et al., 2008; Robins and Ye, 2007; Welch et al., 2014). Young fish likely enter estuaries during the wet season when prawns and other prey species are seasonally abundant. Turbid waters during wet-season flows may protect young threadfin from large predators (Welch et al., 2014). King threadfin particularly inhabit the mid-to-upper estuary, but they are thought to restrict their use of estuarine habitats to permanent water areas in the main channels and tributaries of creeks and rivers (Halliday et al., 2008). Older fish inhabit estuarine and marine systems.

King threadfin are a top predator capable of modifying the estuarine fish and crustacean community in which they live (Salini et al., 1990; Salini et al., 1998). Although king threadfin are restricted to estuarine and marine conditions, the extent and patchiness of wetland and salt flat habitats are likely to be important to king threadfin production (Meynecke et al., 2008), perhaps via productivity and availability of prey. Preying on a range of fish and crustaceans in the coastal ecosystem (Blaber et al., 1995; Salini et al., 1990), king threadfin exemplify an estuary-dependent fish that hunts successfully in turbid waters (Salini et al., 1998). Threadfin are not obligate visual predators; they also use tactile sensors (pectoral filaments) to detect their dominant crustacean prey (prawns) (Pember, 2006; Salini et al., 1998). As adults, their success as a predator may be significantly affected by interruptions to the high-level natural river flows that maintain the turbid, brackish ecotone of tropical rivers within which they successfully hunt.

Threadfin in the Victoria catchment marine region

The Victoria catchment is poorly surveyed for most fish species, including king threadfin, and this is reflected in the paucity of recorded observations of this group. The large tidal range in the estuary renders the river difficult to fish commercially, so too few fishers operate in the estuary to provide commercial catch data. Fish sampling captured 534 king threadfin from the Daly River (north of the Victoria River) for age structure analyses (318 fish were within 600 to 1100 mm total length). The fish were collected opportunistically from commercial and recreational catches throughout 2007 to 2010. Despite poor records for the Victoria catchment marine region, king threadfin is believed to be common in the Victoria River estuary.

Flow–ecology relationships for threadfin

The influence of river flows on commercial catch data are evident. Halliday et al. (2012) recorded that, after adjusting for fishing effort, the annual king threadfin commercial catch from 1990 to 2009 was significantly positively correlated with spring rain lagged by 3 years. It was also significantly but negatively correlated with autumn rain in the year of catch. King threadfin do not use freshwater habitats, so the effect of flood flows on their abundance is less-well defined (Halliday et al., 2012). However, flood flows are key environmental drivers for king threadfin prey, so flow effects on threadfin populations are moderated by food webs, tide regimes, and catchment and estuarine productivity (Jinks et al., 2020).

In some tropical and subtropical rivers, the year-class strength of king threadfin was positively correlated with spring and summer flood flows (Halliday et al., 2008; Halliday et al., 2012). Baseflow in the spring and early-season low flows are used by threadfin larvae in marine habitats as cues to access estuaries. Monsoon flows create a brackish ecotone within estuaries that is prime habitat for threadfin and their prey (Cardona, 2000; Russell and Garrett, 1983; Vance et al., 1998) (Table 3-7). In addition, flood flows deliver nutrients and increase turbidity in estuaries, supporting the food chain and minimising predation; both aspects enhance the survival of juvenile threadfin. Small fish and crustaceans (including penaeid prawns, the prime prey of king threadfin), are abundant in tropical estuaries in the pre-wet and wet seasons (Jinks et al., 2020; Salini et al., 1990; Salini et al., 1998; Vance et al., 1998). In the Australian tropics, the prey community is supported by turbid wet-season flows, though turbidity is also advantageous for fish that do not rely on visual predation alone, such as threadfin using tactile sensing. The ecological functions that support threadfin, and their associated flow requirements, are summarised in Table 3-7.

Table 3-7 Ecological functions supporting threadfin and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of brackish water conditions in estuaries (especially for juveniles) and non-hypersaline water conditions in littoral habitats during the dry season that support threadfin growth and survival	Dry-season duration and intensity	Low flows
Persistence of brackish conditions in estuaries in the early dry season that supports growth and survival of juvenile threadfin (from April onwards)	Early dry-season first low-level flow	Low flows
Provision of brackish water conditions in estuaries and non-hypersaline water conditions in littoral habitats in the late dry season that support threadfin growth and survival (prior to January)	Late dry-season first low-level flow	Low flows
Early wet-season flood flows that support threadfin foraging, growth and survival in estuaries and adjacent littoral habitats	Early wet-season first-flush flows	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that provide nutrients and sediments that support foraging, growth and productivity of mangrove habitats in the estuarine zone and possibly cue migration of threadfin	Moderate flood flows	Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that create freshwater communities within estuaries and floodplains and degrade estuaries as threadfin habitats and cause migration seaward	Large flood flows that cause freshwater estuarine habitats and scour estuaries	High flows, their frequency and seasonal reoccurrence

Pathways to change for threadfin

During a 5-year study in a large Queensland subtropical estuary, king threadfin year-class strength (indicating recruitment and survival of juvenile king threadfin) was positively related to the annual levels of freshwater flow during spring and summer (Halliday et al., 2008). In the Gulf of Carpentaria and the Daly River, both commercial catch (as a measure of abundance) and year-class strength were positively related to monsoon rainfall (often year lagged) in some rivers, but not for all river flows (Halliday et al., 2012; Welch et al., 2014). The survival and growth of king threadfin is likely supported by higher estuarine productivity and abundant prey in years of high flood flow, though these relationships are not robustly studied in tropical Australia (Halliday et al., 2012; Moore et al., 2012). The frequency and duration of high-flood events supports the annual inundation and enhanced primary productivity of floodplain and estuarine supra-littoral habitats (Burford et al., 2016; Ndehedehe et al., 2020a). Carbon and nutrients that are exported to the estuarine and near-shore habitats are used by king threadfin and their prey. Reduced natural flow volumes and interrupted seasonality of monsoon floods would reduce the growth and abundance of king threadfin, as has been found for other large predatory fish that use Gulf of Carpentaria estuaries as prime habitat (Leahy and Robins, 2021). The ecological outcomes of threatening processes on threadfin in northern Australia, and their implications for changes to growth and mortality, community structure, habitat and population, are illustrated in Figure 3-16.

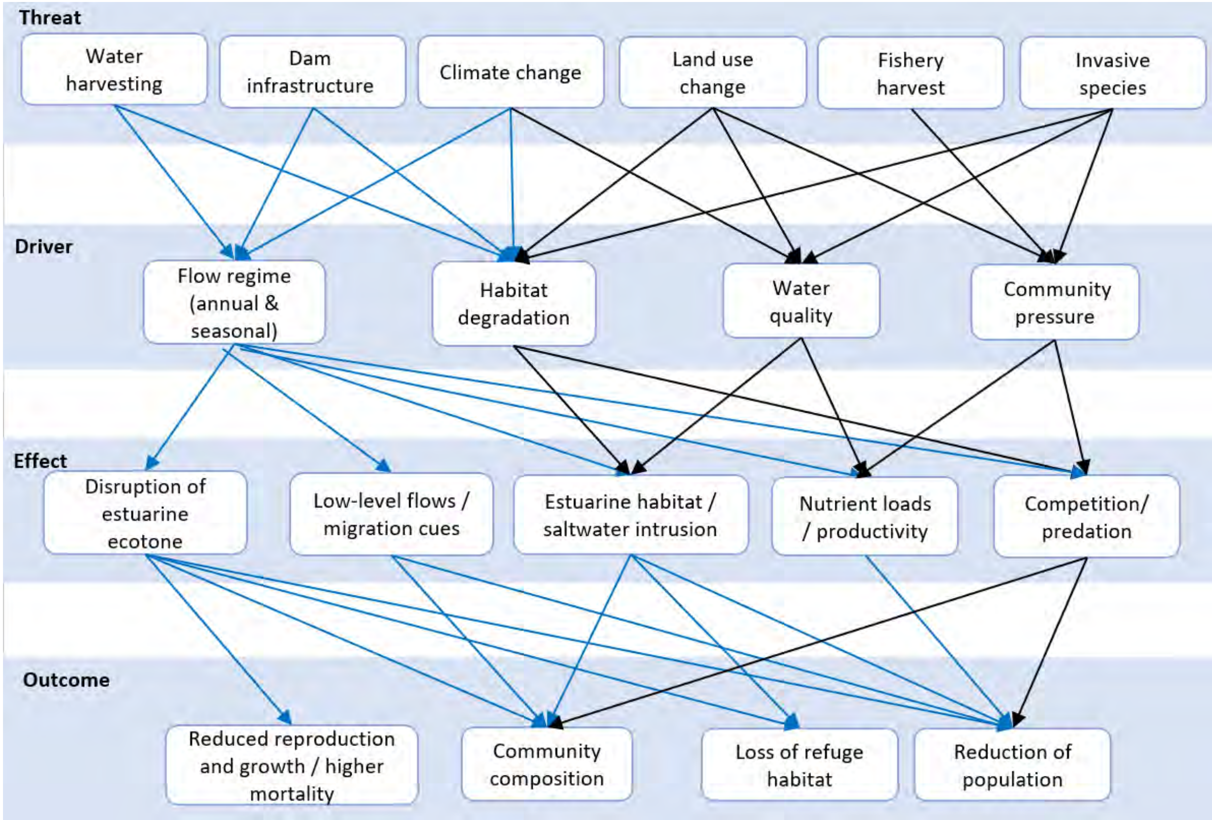

Figure 3-16 Conceptual model showing the relationship between threats, drivers, effects and outcomes for threadfin in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.2 Waterbirds

Grouping waterbirds

Freshwater and saltwater habitats throughout northern Australia are home to a diverse range of waterbird species. Waterbirds are highly dependent on the resources provided by these habitats, including food, shelter and nesting opportunities, all of which are critical for species survival and population maintenance. In most of these habitats, waterbird behaviour, movement and distribution, social organisation and reproductive ecology are largely dependent on natural flooding and rainfall events (Kingsford and Johnson, 1998). Waterbirds respond to flooding and rainfall and subsequent primary and secondary productivity by building condition, moving and breeding (Brandis et al., 2009). Consequently, waterbirds are recognised as important indicators of aquatic ecosystem quality and environmental variability (Garnett et al., 2015 ; Rahman and Ismail, 2018).

Worldwide, populations of waterbirds are in decline, with many species listed as Threatened, Endangered, or Critically Endangered. In Australia, species such as the eastern curlew (*Numenius madagascariensis*), brolga (*Antigone rubicunda, syn. Antigone rubicunda*) and Australian painted snipe (*Rostratula australis*) are listed as priority species through state, federal or international agreements and legislation (Kingsford, 2013). Waterbird population declines are primarily driven by changes in habitat and food availability and quality, driven in turn by changes including in river flow and flood regimes through construction of dams and weirs, water extraction from rivers, water harvesting from floodplains, draining of wetlands, loss of intertidal habitat, over-fishing, water quality changes and other anthropogenic impacts. Consequently, waterbirds are a focal group for the conservation and management of aquatic and semi-aquatic habitats across northern Australia (Bellio et al., 2004; Butchart et al., 2010). Their unique characteristics, visual appeal and social behaviours have historically influenced human culture and continue to engage people and communities with their environments, for example, through cultural activities, traditional stories, symbology, hunting and birdwatching (Kushlan et al., 2002).

To provide a simple basis for understanding and communicating the associated risks and opportunities for waterbirds related to potential water resource development in northern Australia, waterbird species have been divided into four high-level groups. These groups are based on foraging behaviour, nesting behaviour and habitat dependencies. Both foraging and nesting dependencies need to be taken into account, because while some species both forage and nest in northern Australia, others move nomadically or migrate annually to take advantage of foraging opportunities and avoid the northern hemisphere winter. The four waterbird groups are:

- colonial and semi-colonial nesting waders
- cryptic waders
- shorebirds
- swimmers, grazers and divers.

Group 1: Colonial and semi-colonial nesting waders (Section 3.2.1). Colonial and semi-colonial nesting wading species have a high level of dependence on flood timing, extent, duration, depth, vegetation type and condition for breeding. They are also often dependent on specific important breeding sites in Australia. They are usually easily detectable when breeding, and good datasets are available for most species. These species are typically nomadic or partially migratory.

Group 2: Cryptic waders (Section 3.2.2). Cryptic wading species have a high level of dependence on shallow temporary and permanent wetland habitats with relatively dense emergent aquatic vegetation that requires regular or ongoing inundation to survive (e.g. reeds, rushes, sedges, wet grasses and lignum). These species breed in Australia and usually nest as independent pairs, though some may occasionally nest semi-colonially. They may be sedentary, nomadic, migratory or partially migratory. Few data are available; however, habitat requirements can be used as surrogates to assess vulnerability.

Group 3: Shorebirds (Section 3.2.3). Shorebirds have a high level of dependence on end-of-system flows and large inland flood events that provide broad areas of very shallow water and mudflat-type environments. They occur across freshwater and marine habitats, are largely migratory or nomadic (mostly breeding in the northern hemisphere rather than Australia) and are a group of international concern.

Group 4: Swimmers, grazers and divers (Section 3.2.4). These are species with a relatively high level of dependence on semi-open, open and deeper water environments. These species commonly swim when foraging (including diving, filtering, dabbling, grazing) or when taking refuge. They breed in Australia and may be sedentary, nomadic, migratory or partially migratory.

To support the ecology assessment, example species from each group have been selected (see Table 3-8). Species selected are good representatives of the group as a whole, of conservation or cultural importance, and likely to be affected by water resource development. These species provide examples for synthesising the pathways to impact associated with potential water resource developments.

Table 3-8 Waterbird species groups and example representative species for northern Australia

GROUP	REPRESENTATIVE SPECIES	SCIENTIFIC NAME
Colonial and semi-colonial nesting waders	Royal spoonbill	*Platalea regia*
Cryptic waders	Australian painted snipe	*Rostratula australis*
Shorebirds	Eastern curlew	*Numenius madagascariensis*
Swimmers, grazers and divers	Magpie goose	*Anseranas semipalmata*

The primary pathways of potential water resource development impact on waterbirds include: (i) habitat loss, fragmentation and change, (ii) toxins from pollution or contaminants, (iii) disturbance from human activities, (v) predation by invasive or feral animals, and (v) changes in disease, or parasite burdens. Habitat loss, fragmentation and change are the most important drivers of changes in waterbird abundance, population size and diversity worldwide (McGinness, 2016).

The toxic effects of pollution or contaminants such as pesticides, heavy metals, nutrients and other chemicals are known to have caused declines in many populations of waterbirds worldwide (De Luca-Abbott et al., 2001; Howarth et al., 1981; Kim and Oh, 2015). Besides their direct toxic effects, pesticides and herbicides can reduce food availability for waterbirds, depending on their diet. Changes in the extent or intensity of water resource development and subsequent agricultural developments are often associated with increases in the amounts of pollution or contaminants such as pesticides, heavy metals, nutrients and other chemicals in catchments, and therefore present risks to waterbird populations.

Predation is a natural component of waterbird population biology. However the nature and importance of its impact can be changed by anthropogenic changes, in particular, the introduction of feral predators such as pigs and habitat alteration via introduced plants and herbivores (Sovada et al., 2001). Changes in water levels during nesting periods can make nests more accessible and vulnerable to predators (McGinness, 2016). Many studies have shown that predation on waterbirds occurs mainly during nesting and is dominated by egg predation; nestling and fledgling predation are also reported. Predation on adult waterbirds is relatively rare, but is probably additive to mortality due to other factors (e.g. hunting, pollution (Sovada et al., 2001)). Predators such as pigs can reduce the survival of waterbirds, and consequently population size, either through direct predation or indirectly by causing adults to desert their nests or foraging sites. They can also affect population size by competing for habitat or food, or affecting other predators and prey (Cruz et al., 2013; MacDonald and Bolton, 2008; Skorka et al., 2014).

Disturbance from human activities can affect bird behaviour and temporal and spatial distribution of waterbirds. Human disturbance can be equivalent to habitat loss or degradation because it may lead waterbirds to avoid or underuse areas (Fernandez and Lank, 2008). Temporary loss of foraging habitats can occur, and species vary in their capacity to compensate by foraging for longer periods (Sutherland et al., 2012). During the breeding season, human disturbance may also influence nest incubation and chick rearing, affecting overall nest success and eventual recruitment, which then affects population sizes and trajectories.

Disease and parasites can affect waterbird nest success, fledging rates, juvenile survival and adult survival. They are more likely to be a problem where there is insufficient habitat and birds are crowded, which can occur following changes in flood regimes and habitats due to water resource development or land development (McGinness, 2016). Infectious diseases are an important and dominant mortality factor in waterbird populations. Bacteria such as *Clostridium botulinum* and viruses such as avian influenza, West Nile virus, Newcastle disease virus, avian poxvirus, duck plague, avian bornavirus, reoviruses and adenoviruses may contribute to population declines of both wild and domestic waterbirds. The infection rate by *Plasmodium* parasites (avian malaria) is rapidly increasing in many birds, and infection rates of campylobacteria in waders are high (Sutherland et al., 2012). Ticks parasitising nestlings can reduce survival and nest success, and potentially also transmit viruses. Changes in land use and global climate may concentrate waterbirds on remaining high-quality sites, making them potentially more vulnerable to infections (Sutherland et al., 2012).

Where impacts on waterbird populations are natural processes (e.g. predation, disease), anthropogenic influences have almost always altered those processes, as described above. Consequently, such processes can become management problems, even though they are fundamentally natural. Interactions are also likely with climate change. Climate change is affecting seasonal and extreme temperatures; the timing, intensity, amount and duration of rain; and the frequency and severity of extreme weather events, all of which have the potential to influence waterbird populations positively and negatively, and directly and indirectly (Chambers et al., 2005; Sutherland et al., 2012).

3.2.1 Colonial and semi-colonial nesting wading waterbirds

Description and background to ecology

The colonial and semi-colonial nesting wading waterbirds (colonial waders) group comprises wading waterbird species with a high level of dependence on water for breeding, including requirements for flood timing, extent, duration, depth, vegetation type and vegetation condition. In northern Australia, this group comprises 21 species from 5 families, including ibis, spoonbills, herons, egrets, avocets, stilts, storks and cranes (Figure 3-17). The species in this group are often easily detectable when breeding, and relatively good datasets are available for most, unlike for other species or groups.

The species in this group often depend on specific important breeding sites (Arthur et al., 2012). Ibis, spoonbills, herons, egrets, avocets and stilts nest in loose groups or dense colonies of hundreds to tens of thousands of birds in specific vegetation types and locations, over or adjacent to water (Bino et al., 2014). Storks (such as the black-necked stork, *Ephippiorhynchus asiaticus*) and cranes including the brolga (*Antigone rubicunda*) and sarus crane (*Antigone antigone*) usually nest independently, but loose widely spaced groups of nests may occur in suitable habitat. Species in this group may travel up to thousands of kilometres to use these sites (McGinness et al., 2019), and nesting events can last several months, depending on inundation conditions (Kingsford et al., 2012). Species in this group usually have a mixed diet including fish, frogs, crustaceans and insects, and use foraging methods such as walking, stalking and striking to catch their prey. Colonial and semi-colonial nesting waders generally prefer shallow water or damp sediment with medium- to low-density vegetation for foraging (Garnett et al., 2015). These species are typically nomadic or partially migratory but may spend long periods in particular locations when conditions are suitable.

For the assessment, the species selected as representative of the colonial and semi-colonial nesting waders group is the royal spoonbill (*Platalea regia*; Figure 3-17). The royal spoonbill is a large wading species highly adapted to foraging in shallow wetlands (Marchant and Higgins, 1990). This species requires water and water-dependent vegetation for feeding, nesting, refuge, roosting and movement habitat (e.g. 'stopover' habitat for longer distance trips) (Marchant and Higgins, 1990). Spoonbills nest in loose colonies, usually in vegetation surrounded by water, including reedbeds, semi-aquatic shrubs and trees. They often nest adjacent to colonies of other species in the group.

Figure 3-17 Royal spoonbills at the nest

Royal spoonbills are a representative species of the colonial and semi-colonial nesting waders waterbird group

Photo attribution: CSIRO

Table 3-9 Species in the colonial and semi-colonial nesting wading waterbird group, and their national and international conservation status

(LC = Least concern)

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	IUCN STATUS
Australian white ibis	*Threskiornis moluccus*	Threskiornithidae	LC
Banded stilt	*Cladorhynchus leucocephalus*	Recurvirostridae	LC
Black-winged stilt (pied stilt)	*Himantopus leucocephalus* (*Himantopus himantopus*)	Recurvirostridae	LC
Cattle egret	*Bubulcus ibis* (syn. *Ardea ibis*)	Ardeidae	LC
Eastern reef egret	*Egretta sacra*	Ardeidae	LC
Glossy ibis	*Plegadis falcinellus*	Threskiornithidae	LC
Great egret (eastern great egret)	*Ardea alba modesta* (syn *Ardea modesta*)	Ardeidae	LC
Great-billed heron	*Ardea sumatrana*	Ardeidae	LC
Plumed egret (Intermediate egret)	*Ardea plumifera (formerly Ardea intermedia)*	Ardeidae	LC
Little egret	*Egretta garzetta*	Ardeidae	LC
Nankeen night-heron	*Nycticorax caledonicus*	Ardeidae	LC
Pied heron	*Egretta picata* (formerly *Ardea picata*)	Ardeidae	LC
Red-necked avocet	*Recurvirostra novaehollandiae*	Recurvirostridae	LC
Royal spoonbill	*Platalea regia*	Threskiornithidae	LC
Sarus crane	*Grus antigone*	Gruidae	Vulnerable
Straw-necked ibis	*Threskiornis spinicollis*	Threskiornithidae	LC
White-faced heron	*Egretta novaehollandiae*	Ardeidae	LC
White-necked heron	*Ardea pacifica*	Ardeidae	LC
Yellow-billed spoonbill	*Platalea flavipes*	Threskiornithidae	LC
Black-necked stork	*Ephippiorhynchus asiaticus*	Ciconiidae	LC
Brolga	*Antigone rubicunda*	Gruidae	LC

Colonial and semi-colonial nesting waders in the Victoria catchment

Colonial and semi-colonial nesting waders are found throughout the Victoria catchment (Figure 3-18 and Figure 3-19). Waterbird surveys have been conducted in the Joseph Bonaparte Gulf area and the Legune Station area in the north-west of the catchment (Chatto, 2000; 2003; 2006). The 2006 survey found six significant breeding sites within the survey block. Species at these sites included the pied heron (*Egretta picata*), little egret (*Egretta garzetta*), great egret (*Ardea alba*), intermediate egret (*Ardea intermedia*), nankeen night-heron (*Nycticorax caledonicus*), Australian white ibis (*Threskiornis moluccus*) and royal spoonbill (*Platalea regia*) breeding in significant numbers (Chatto, 2006). Other birds within this group that were found in significant numbers include the brolga (*Antigone rubicunda*), glossy ibis (*Plegadis falcinellus*), black-winged stilt (*Himantopus himantopus*) and egret species (*Egretta* spp.) (Chatto, 2006). The modelled distribution of the royal spoonbill is shown in Figure 3-20.

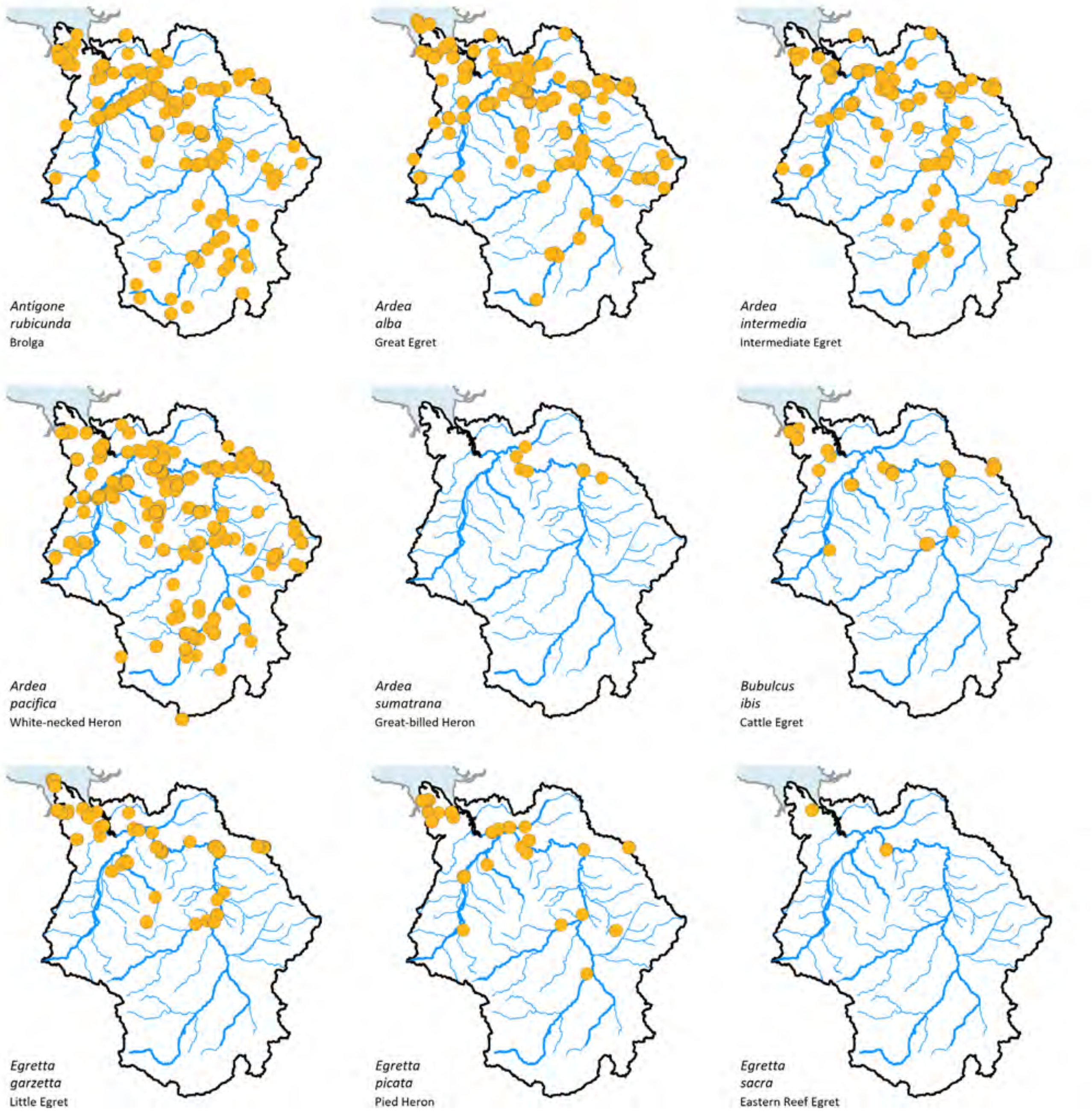

Figure 3-18 Observed locations of colonial and semi-colonial nesting wading waterbirds in the Victoria catchment in alphabetic order of species name: *Antigone rubicunda* (brolga) to *Egretta sacra* (eastern reef egret)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

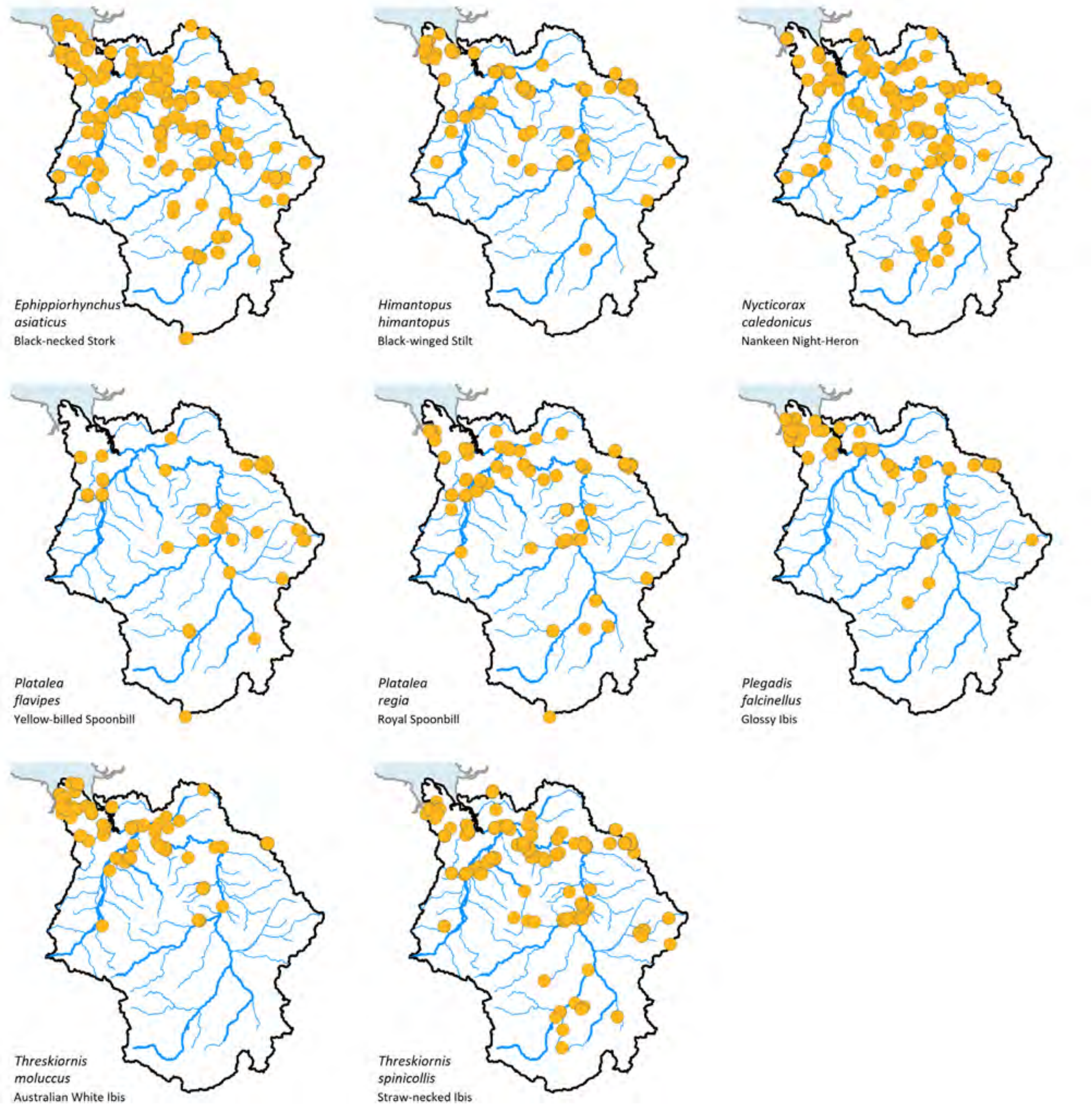

Figure 3-19 Observed locations of colonial and semi-colonial nesting wading waterbirds in the Victoria catchment in alphabetic order of species name: *Ephippiorhynchus asiaticus* (black-necked stork) to *Threskiornis spinicollis* (straw-necked ibis)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

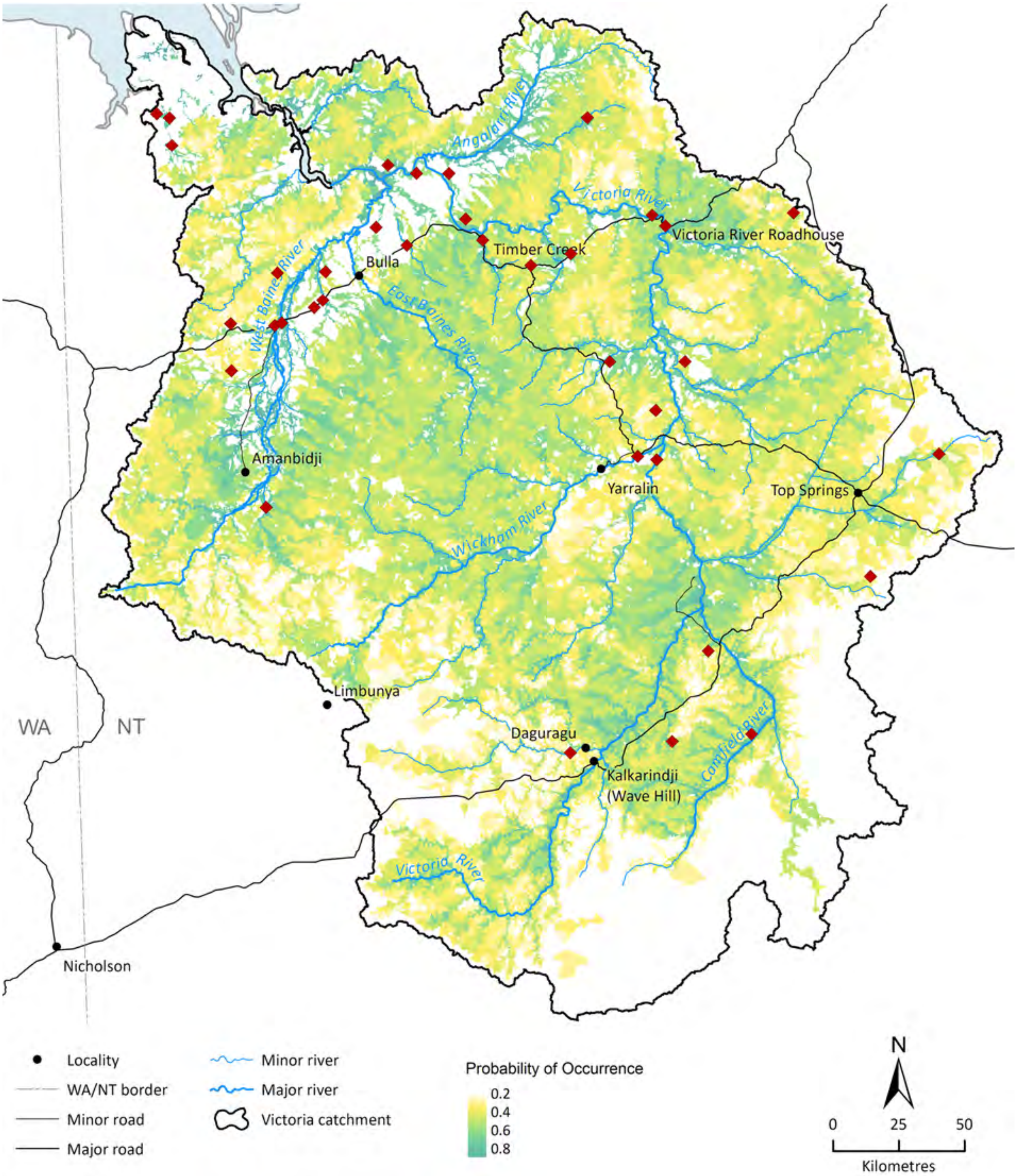

Figure 3-20 Modelled probability of occurrence of royal spoonbill (*Platalea regia*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for colonial and semi-colonial nesting wading waterbirds

Waterbird species in the colonial and semi-colonial nesting waders group are sensitive to changes in the depth, extent and duration of shallow wetland environments, particularly during nesting events. Colonial nesting waders nest when and where weather, water and vegetation provide optimal conditions, including suitable vegetation structure and water around nests for protection from predation and weather (Kingsford and Norman, 2002) and sufficient food resources (Figure 3-21) (O'Brien and McGinness, 2019). Completion of a full nesting cycle can take several months. During this time, changes in water depth, water extent, water duration or food availability can force adults to abandon their nests or expose nests to predation, resulting in nest failure, and in the long term can result in abandonment of regular breeding sites (Brandis, 2010; Brandis et al., 2011). Adults of these species may not breed every year, and recruitment rates post-breeding are frequently low because of this dependence on suitable hydrological and weather conditions to support food resources and habitats. Nesting failures may have a serious impact on population sizes and trajectories (Kingsford and Norman, 2002). The ecological functions that support colonial and semi-colonial nesting waders, and their associated flow requirements, are summarised in Table 3-10.

Table 3-10 Ecological functions supporting colonial and semi-colonial nesting waders and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW REQUIREMENT OR ATTRIBUTE
Meeting water requirements for foraging, roosting and nesting	Foraging: Damp sediment, shallow water or the edges of deeper water habitats. Ephemeral habitats preferred due to greater food availability. Roosting trees within 2 km Nesting: Vegetation standing in or next to water 0.5–2.5 m deep. No sudden changes in water depth during nesting periods	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, reproduction timing, recruitment timing Rate of change in flow events – rate of change in depth
Water regime to support required vegetation types and condition	Dense semi-aquatic vegetation such as reeds and shrubs, and fringing trees. Vegetation type depends on the long-term flow regime to support these communities, while the short-term flow regime affects vegetation condition	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Water regime to support suitable water quality	Low salinity, low turbidity, low toxic algae and cyanobacteria levels, low nutrient (e.g. no eutrophication)	Magnitude of flow events – inundation extent, inundation depth, flow rate when connected Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth and flow rate

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW REQUIREMENT OR ATTRIBUTE
Water regime to support food availability	Suitable abundance of fish, crustaceans, molluscs, invertebrates, frogs, tadpoles	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Competition and predation and diseases and parasites Water regime to reduce risk	Water regime to provide sufficient habitat extent to avoid overcrowding and provide multiple alternative site options to avoid predators	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth

Figure 3-21 Egret hunting among water lilies

Egrets are species of the colonial and semi-colonial nesting waders group.

Photo attribution: CSIRO

Pathways to change for colonial and semi-colonial nesting wading waterbirds

The primary pathways of potential water resource development impact on colonial waders are habitat loss, fragmentation and change (Figure 3-22). Because of the specific needs of colonial waders regarding water regimes in suitable nesting habitats, colony sites in areas subject to changes in flood regimes due to water resource developments (e.g. river regulation through dams or weirs, water extraction from rivers, floodplain water harvesting) are at high risk of damage or loss, with implications for population maintenance (Brandis et al., 2011). Unnatural or unexpected changes in the depth, extent, frequency and duration of inundation in wetland habitats used by colonial and semi-colonial nesting waders for nesting and foraging can have significant impacts on nesting, nest success, juvenile recruitment and adult survival (Bino et al., 2014; Brandis et al., 2018; Brandis et al., 2011; Kingsford et al., 2011). Changes can also reduce water quality and food availability, and increase rates of competition, predation and disease (McGinness, 2016). Changes can occur when flood peaks are reduced by water extraction or dams (e.g. by reducing flood extent, frequency, duration or depth), when floodwater is captured on floodplains (e.g. by dams, levees or roads), or when the time between inundation events that create these habitats is extended (Kingsford and Thomas, 2004). The life histories of many of these species have evolved to expect natural flooding regimes, so they are affected when these regimes are changed.

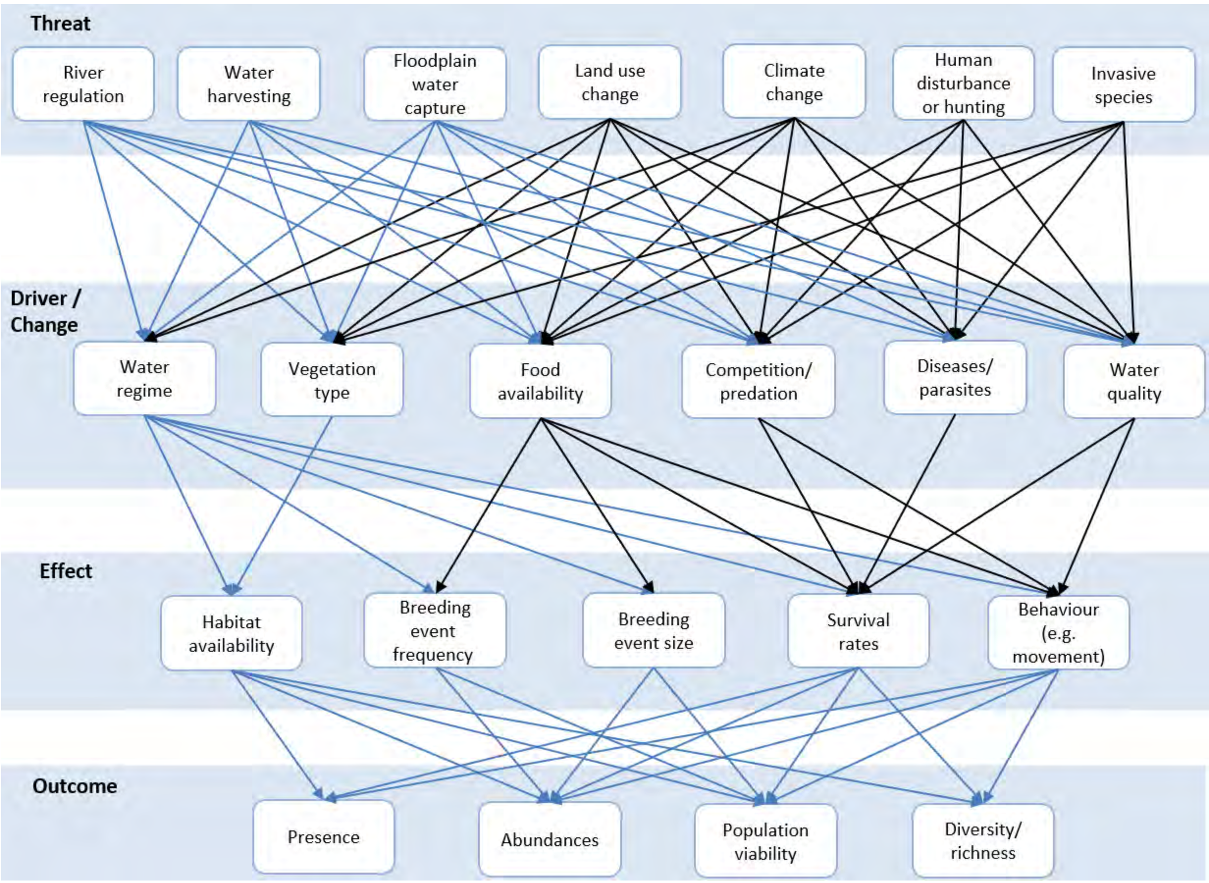

Figure 3-22 Conceptual model showing the potential relationship between threats, drivers, effects and outcomes for colonial and semi-colonial nesting wading waterbird species

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.2.2 Cryptic wading waterbirds

Description and background to ecology

The cryptic waders group comprises wading waterbird species that are relatively difficult to detect and have a high level of dependence on shallow temporary and permanent wetland habitats with relatively dense emergent aquatic vegetation (Figure 3-23) that requires regular or ongoing inundation to survive (e.g. reeds, rushes, sedges, wet grasses). In northern Australia, this group comprises 13 species from four families, including bitterns, crakes, rails and snipe (Table 3-11).

Figure 3-23 Dense aquatic and semi-aquatic vegetation used as habitat by cryptic wading waterbirds
This habitat provides protection from predators and weather.
Photo attribution: CSIRO

Species from this group are often present in low numbers and are difficult to detect even when breeding; consequently, datasets are generally sparse, and a lack of incidental records does not necessarily mean the species is absent. Cryptic wader species usually nest as independent pairs, though some may nest semi-colonially (Marchant and Higgins, 1990). Nesting generally occurs seasonally. They may be sedentary, nomadic, migratory or partially migratory (Garnett et al., 2015; Marchant and Higgins, 1990). Movements between sites are likely to be partly dependent on the availability of suitable wetland habitats between origin and destination sites for shelter and feeding.

Species in this group usually have an invertivorous or omnivorous diet and use foraging methods such as walking, stalking, striking and probing to catch their prey (Barker and Vestjens, 1989). Cryptic waders generally prefer shallow water or damp sediment with medium to high-density vegetation (Garnett et al., 2015). For nesting, some species require deeper water environments with dense vegetation, while others require very shallow water or recently dried wetland environments (Marchant and Higgins, 1990). Changes in water depth, water extent, water duration or food availability may result in nest exposure to predation or reduced food availability, resulting in nest failure (McGinness, 2016).

For the purpose of this Assessment, the endangered Australian painted snipe (*Rostratula australis*) is a representative species for the cryptic waders group and is rarely seen throughout its range (Rogers et al., 2004). It is a shy species that spends most of its time hidden in vegetation or woody debris in shallow-water areas. The population is small and has declined significantly across much of its range, most likely due to loss and degradation of inland floodplain wetland habitats and in particular breeding habitats (Rogers et al., 2004).

Table 3-11 Species in the cryptic wading waterbird group and their national and international conservation status (LC = Least concern)

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	IUCN STATUS
Australian little bittern	*Ixobrychus dubius (syn. Ixobrychus minutus)*	Ardeidae	LC
Australian painted snipe	*Rostratula australis*	Rostratulidae	Endangered
Australian spotted crake	*Porzana fluminea*	Rallidae	LC
Baillon's crake	*Porzana pusilla (Zapornia pusilla)*	Rallidae	LC
Black bittern	*Ixobrychus flavicollis*	Ardeidae	LC
Buff-banded rail	*Hypotaenidia philippensis*	Rallidae	LC
Chestnut rail	*Eulabeornis castaneoventris*	Rallidae	LC
Latham's snipe	*Gallinago hardwickii*	Scolopacidae	LC
Lewin's rail	*Lewinia pectoralis*	Rallidae	LC
Red-necked crake	*Rallina tricolor*	Rallidae	LC
Spotless crake	*Zapornia tabuensis (Porzana tabuensis)*	Rallidae	LC
Striated heron	*Butorides striatus (Butorides striata)*	Ardeidae	LC
White-browed crake	*Amaurornis cinerea (Poliolimnas cinereus)*	Rallidae	LC

Cryptic wading waterbirds in the Victoria catchment

Cryptic waders are found throughout the Victoria catchment (Figure 3-24). Suitable habitat for this group includes floodplain areas, the Bradshaw Field Training Area nationally significant wetland (see Section 3.4.1 for the floodplain wetlands asset description) and deeper stretches of river that form waterholes during the dry season (Section 3.4.2 for the waterholes asset description). The modelled distribution of the Australian painted snipe is shown in Figure 3-25.

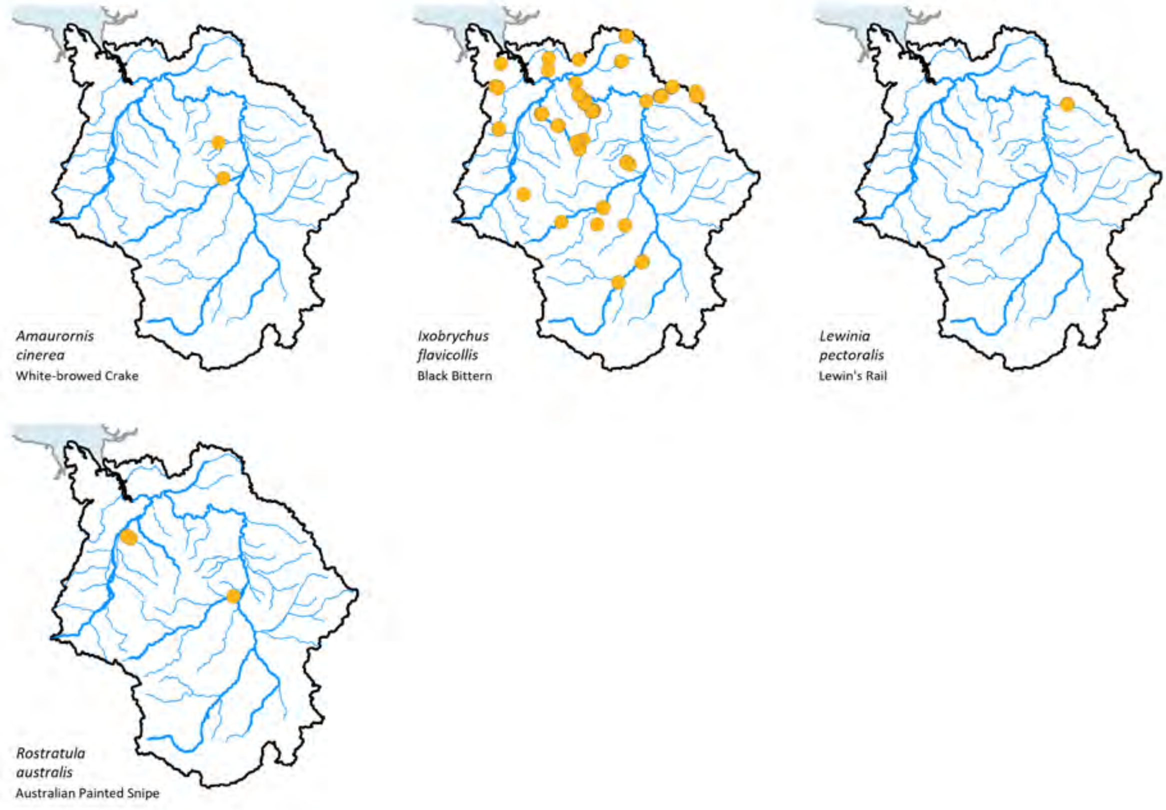

Figure 3-24 Observed locations of selected cryptic wading waterbirds in the Victoria catchment

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

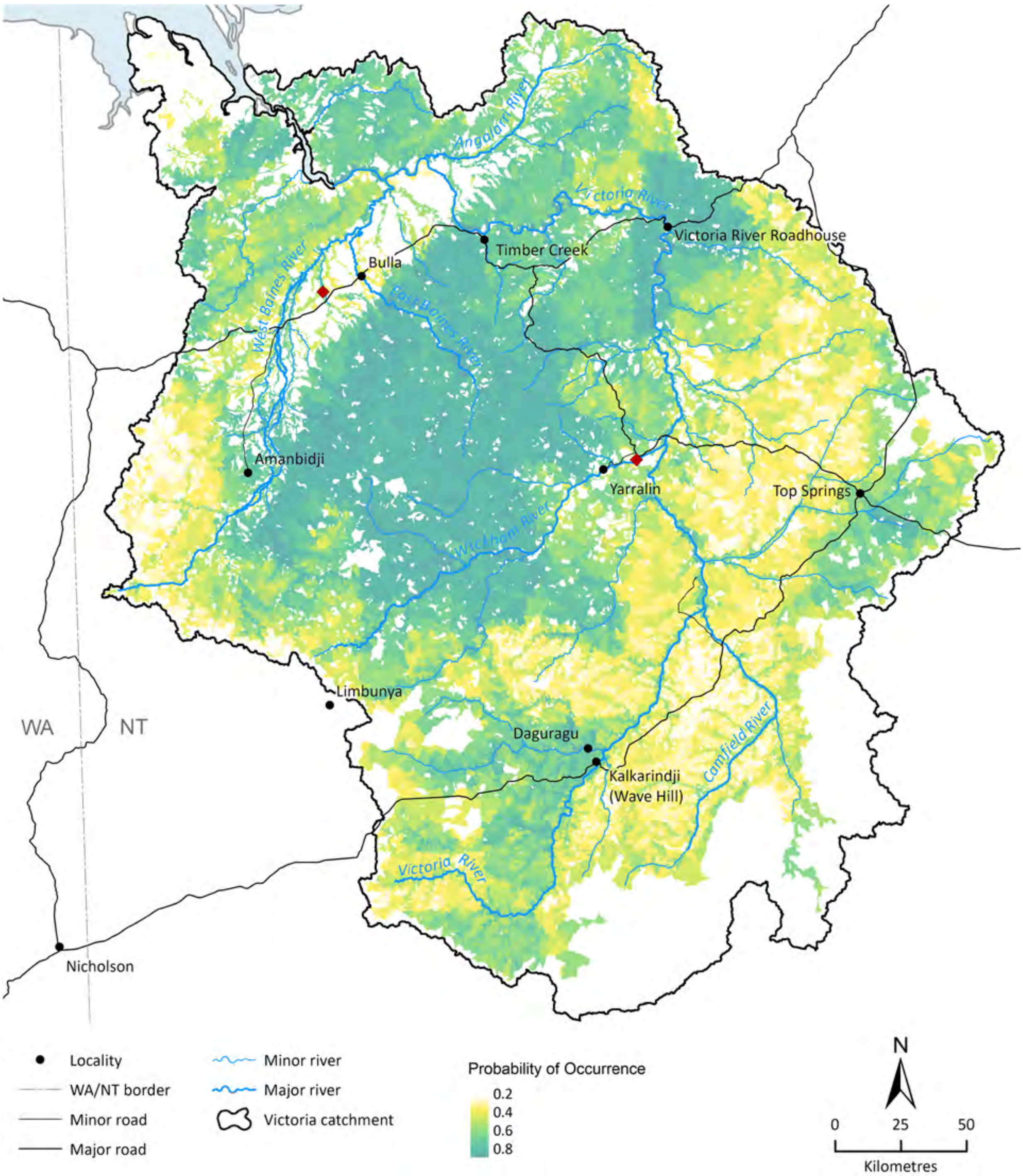

Figure 3-25 Modelled probability of occurrence of Australian painted snipe (*Rostratula australis*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for cryptic wading waterbirds

Waterbird species in the cryptic waders group are sensitive to changes in the depth, extent and duration of shallow wetland environments and the fringes of deeper-water habitats such as waterholes (Kingsford and Norman, 2002; Marchant and Higgins, 1990; McGinness, 2016). Most species nest on the ground or in low vegetation, so nests are at risk when water levels change

(Garnett et al., 2015; Marchant and Higgins, 1990). Cryptic waders are particularly sensitive to changes in the type, density or extent of emergent aquatic and semi-aquatic vegetation in and around these habitats. Besides changing foraging, nesting and refuge habitat, such changes can also reduce water quality and food availability and increase rates of competition, predation and disease (McGinness, 2016). Such changes can occur when water is directly extracted from these habitats or when the time between inundation events that create these habitats is extended (Brandis et al., 2009; Kingsford and Norman, 2002). Climate change and climate change−driven extremes are likely to interact with changes induced by water resource development, including inundation of freshwater habitats by seawater and inundation of nests by extreme flood events or seawater intrusion. The ecological functions that support cryptic wading waterbirds, and their associated flow requirements, are summarised in Table 3-12.

Table 3-12 Ecological functions supporting cryptic wading waterbirds and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Meeting water requirements for foraging, roosting and nesting	Damp sediment, shallow water or the edges of deeper water habitats. Ephemeral to permanent water. No sudden changes in water depth during nesting periods	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, reproduction timing, recruitment timing Rate of change in flow events – rate of change in depth
Water regime to support required vegetation types and condition	Dense aquatic and semi-aquatic vegetation for refuge or shelter, interspersed with more open areas for foraging. Vegetation type depends on the long-term flow regime to support these communities, while the short-term flow regime affects vegetation condition	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Water regime to support suitable water quality	Low salinity, low turbidity, low toxic algae and cyanobacteria levels, low nutrient (e.g. no eutrophication)	Magnitude of flow events – inundation extent, inundation depth, flow rate when connected Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth and flow rate
Water regime to support food availability	Fish, crustaceans, molluscs, invertebrates, frogs, tadpoles, aquatic and semi-aquatic plants	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Competition and predation and diseases and parasites Water regime to reduce risk	Water regime to provide sufficient habitat extent to avoid overcrowding and provide multiple alternative site options to avoid predators	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth

Pathways to change for cryptic wading waterbirds

Few data are available for cryptic waders, but habitat requirements can be used as surrogates to assess vulnerability and pathways to change. The cryptic wader group's need for appropriate vegetation and shallow-water environments makes them sensitive to changes in both water regimes and vegetation throughout their life cycles (Marchant and Higgins, 1990). Thus, the primary pathways of potential water resource development impacts on cryptic waders are habitat loss, fragmentation and change through changes in the timing, extent, depth and duration of inundation, which in turn change vegetation (Kingsford and Norman, 2002; McGinness, 2016; McKilligan, 2005) (Figure 3-26). In addition to direct disturbance from changes in hydrology and vegetation, species are also at risk from increased disturbance from human activities and predation (Kingsford and Norman, 2002). Human disturbance can be equivalent to habitat loss or degradation because it may lead waterbirds to avoid or underuse areas. During the breeding season, disturbance and predation may influence nest incubation and chick rearing, affecting overall nest success and eventual recruitment, which then affects population sizes and trajectories (McGinness, 2016). Changes in water regimes and vegetation can change predation pressure through increased exposure of cryptic waders and their nests (Sovada et al., 2001). Increased predation due to such changes can reduce the survival of cryptic waders and consequently population size either directly or indirectly by causing adults to desert their nests or foraging sites. Predators can also affect population size by competing for habitat or food, or affecting other predators and prey (Cruz et al., 2013; MacDonald and Bolton, 2008; Skorka et al., 2014).

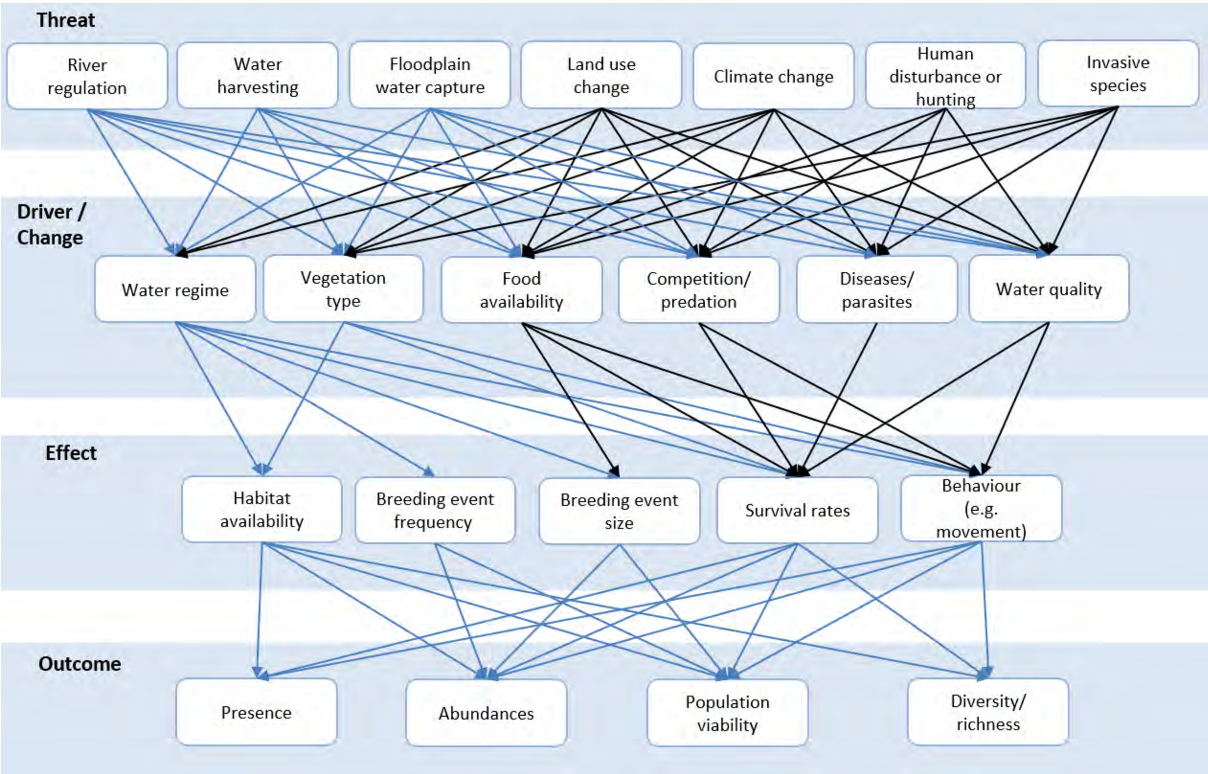

Figure 3-26 Conceptual model showing the relationship between threats, drivers, effects and outcomes for cryptic wading waterbirds in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.2.3 Shorebirds

Description and background to ecology

The shorebirds group consists of waterbirds with a high level of dependence on end-of-system flows and large inland flood events that provide broad areas of shallow water and mudflat environments. Flood events trigger production of significant food resources for these species – resources that are critical for fuelling long-distance migrations. Shorebirds generally eat fish or invertebrates. Most species walk and wade when foraging, probing sediment, rocks or vegetation for prey (Garnett et al., 2015; Marchant and Higgins, 1990).

Shorebirds are largely migratory, mostly breeding in the northern hemisphere. They are in significant decline and are of international concern. Shorebirds depend on specific shallow-water habitats in distinct geographic areas, including northern hemisphere breeding grounds, southern hemisphere non-breeding grounds and stopover sites along migration routes such as the East Asian-Australasian Flyway (Bamford, 1992; Hansen et al., 2016). As the group is of international concern, various management and conservation strategies have been implemented (DAWE, 2021), including bilateral migratory bird agreements with China (CAMBA), Japan (JAMBA), and Korea (ROKAMBA), the Bonn Convention on the Conservation of Migratory Species of Wild Animals (Bonn), and the Ramsar Convention on Wetlands of International Importance.

In northern Australia, this group comprises approximately 55 species from four families, including sandpipers, godwits, curlew, stints, plovers, dotterel, lapwings and pratincoles (Table 3-13). Approximately 35 species are common, regular visitors or residents. Several species in this group are endangered globally and nationally, including the bar-tailed godwit (*Limosa lapponica*), curlew sandpiper (*Calidris ferruginea*), eastern curlew, great knot (*Calidris tenuirostris*), lesser sand plover (*Charadrius mongolus*) and red knot (*Calidris canutus*).

The eastern curlew is listed as Critically Endangered under the EPBC Act and recognised through multiple international agreements as requiring habitat protection in Australia. Eastern curlews rely on food sources along shorelines, mudflats and rocky inlets, as well as roosting vegetation. Developments and disturbances, such as recreational, residential and industrial use of these habitats, have restricted habitat and food availability for the eastern curlew, contributing to population declines.

The red-capped plover (*Charadrius ruficapillus*; Figure 3-27) is a shorebird that breeds in Australia rather than in the northern hemisphere. It is a small species that is widespread and common both inland and along the coast. It prefers open flat sediment areas such as mudflats and beaches for foraging and eats a range of small invertebrates including crustaceans. It breeds in response to flooding or rain inland, and seasonally on the coasts.

Photo attribution: CSIRO

Table 3-13 Species in the shorebirds group and their national and international conservation status
(LC = Least concern)

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Australian pratincole	*Stiltia isabella*	Glareolidae	Australian	LC	LC
Beach stone-curlew	*Esacus magnirostris*	Burhinidae	Australian	Near Threatened	LC
Masked lapwing	*Vanellus miles*	Charadriidae	Australian	LC	LC
Red-capped plover	*Charadrius ruficapillus*	Charadriidae	Australian	LC	LC
Black-fronted dotterel	*Elseyornis melanops*	Charadriidae	Endemic	LC	LC
Inland dotterel	*Charadrius australis (Peltohyas australis)*	Charadriidae	Endemic	LC	LC
Red-kneed dotterel	*Erythrogonys cinctus*	Charadriidae	Endemic	LC	LC
Banded lapwing	*Vanellus tricolor*	Charadriidae	Endemic	LC	LC
Bar-tailed godwit	*Limosa lapponica*	Scolopacidae	Non-breeding migrant	LC	Critically Endangered
Black-tailed godwit	*Limosa limosa*	Scolopacidae	Non-breeding migrant	Near Threatened	Near Threatened
Broad-billed sandpiper	*Limicola falcinellus*	Scolopacidae	Non-breeding migrant	LC	LC
Common greenshank	*Tringa nebularia*	Scolopacidae	Non-breeding migrant	LC	LC
Common sandpiper	*Actitis hypoleucos*	Scolopacidae	Non-breeding migrant	LC	LC
Curlew sandpiper	*Calidris ferruginea*	Scolopacidae	Non-breeding migrant	Vulnerable	Critically Endangered
Eastern curlew	*Numenius madagascariensis*	Scolopacidae	Non-breeding migrant	Vulnerable	Critically Endangered
Great knot	*Calidris tenuirostris*	Scolopacidae	Non-breeding migrant	Vulnerable	Critically Endangered
Greater sand plover, Large sand plover	*Charadrius leschenaultii*	Charadriidae	Non-breeding migrant	LC	LC
Grey plover	*Pluvialis squatarola*	Charadriidae	Non-breeding migrant	LC	LC
Grey-tailed tattler	*Tringa brevipes*	Scolopacidae	Non-breeding migrant	Near Threatened	LC
Lesser sand plover	*Charadrius mongolus*	Charadriidae	Non-breeding migrant	LC	Critically Endangered
Little curlew	*Numenius minutus*	Scolopacidae	Non-breeding migrant	LC	LC

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Long-toed stint	*Calidris subminuta*	Scolopacidae	Non-breeding migrant	LC	LC
Marsh sandpiper	*Tringa stagnatilis*	Scolopacidae	Non-breeding migrant	LC	LC
Oriental plover, Oriental dotterel	*Charadrius veredus*	Charadriidae	Non-breeding migrant	LC	LC
Oriental pratincole	*Glareola maldivarum*	Glareolidae	Non-breeding migrant	LC	LC
Pacific golden plover	*Pluvialis fulva*	Charadriidae	Non-breeding migrant	LC	LC
Red knot	*Calidris canutus*	Scolopacidae	Non-breeding migrant	Vulnerable	Critically Endangered
Red-necked stint	*Calidris ruficollis*	Scolopacidae	Non-breeding migrant	LC	Near Threatened
Ruddy turnstone	*Arenaria interpres*	Scolopacidae	Non-breeding migrant	Near Threatened	Near Threatened
Sanderling	*Calidris alba*	Scolopacidae	Non-breeding migrant	LC	LC
Sharp-tailed sandpiper	*Calidris acuminata*	Scolopacidae	Non-breeding migrant	LC	LC
Swinhoe’s snipe	*Gallinago megala*	Scolopacidae	Non-breeding migrant	LC	LC
Terek sandpiper	*Xenus cinereus*	Scolopacidae	Non-breeding migrant	LC	LC
Whimbrel	*Numenius phaeopus*	Scolopacidae	Non-breeding migrant	LC	LC
Wood sandpiper	*Tringa glareola*	Scolopacidae	Non-breeding migrant	LC	LC
Asian dowitcher	*Limnodromus semipalmatus*	Scolopacidae	Non-breeding migrant	Near Threatened	Near Threatened
Common redshank, Redshank	*Tringa totanus*	Scolopacidae	Non-breeding migrant	LC	LC
Double-banded plover	*Charadrius bicinctus*	Charadriidae	Non-breeding migrant	LC	LC
Pectoral sandpiper	*Calidris melanotos*	Scolopacidae	Non-breeding migrant	LC	LC
Wandering tattler	*Tringa incana (Heteroscelus incanus)*	Scolopacidae	Non-breeding migrant	LC	LC
Little ringed plover	*Charadrius dubius*	Charadriidae	Non-breeding migrant	LC	LC
Pin-tailed snipe	*Gallinago stenura*	Scolopacidae	Non-breeding migrant	LC	Data Deficient
Red-necked phalarope	*Phalaropus lobatus*	Scolopacidae	Non-breeding migrant	LC	LC

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Ruff (reeve)	*Calidris pugnax (Philomachus pugnax)*	Scolopacidae	Non-breeding migrant	LC	LC
Baird's sandpiper	*Calidris bairdii*	Scolopacidae	Vagrant	LC	LC
Caspian plover	*Charadrius asiaticus*	Charadriidae	Vagrant	LC	LC
Green sandpiper	*Tringa ochropus*	Scolopacidae	Vagrant	LC	LC
Buff-breasted sandpiper	*Tryngites subruficollis*	Scolopacidae	Vagrant	Near Threatened	LC
Dunlin	*Calidris alpina*	Scolopacidae	Vagrant	LC	LC
Grey (red) phalarope	*Phalaropus fulicaria*	Scolopacidae	Vagrant	LC	LC
Lesser yellowlegs	*Tringa flavipes*	Scolopacidae	Vagrant	LC	LC
Little stint	*Calidris minuta*	Scolopacidae	Vagrant	LC	LC
Common ringed plover	*Charadrius hiaticula*	Charadriidae	Vagrant	LC	LC
Spotted redshank	*Tringa erythropus*	Scolopacidae	Vagrant	LC	LC
White-rumped sandpiper	*Calidris fuscicollis*	Scolopacidae	Vagrant	LC	LC

Figure 3-27 Red-capped plover walking along a shore
The red-capped plover is a member of the shorebirds group.

Shorebirds in the Victoria catchment

The shorebirds group favours habitats that are open with very shallow water, including the edges of inland floodplains, lakes, estuarine and coastal mudflats, and sandflats (Figure 3-28, Figure 3-29, Figure 3-30 and Figure 3-31). Saline coastal wetlands are located around the Joseph Bonaparte Gulf in the north-western part of the catchment. However, Chatto (2006) found that these areas were rarely used by shorebirds; extensive surveys in the area, including the Legune Station, were unable to identify any shorebird species. However, previous surveys by Chatto (2003) identified a number of shorebirds, including the Terek sandpipers (*Xenus cinereus*), greater sand plover (*Charadrius leschenaultii*), bar-tailed godwit and the red-necked stint (*Calidris ruficollis*) as being the most common shorebirds. The large estuarine area at the mouth of the Victoria River is also likely to be suitable habitat for this group. The modelled distribution of the red-capped plover is shown in Figure 3-32.

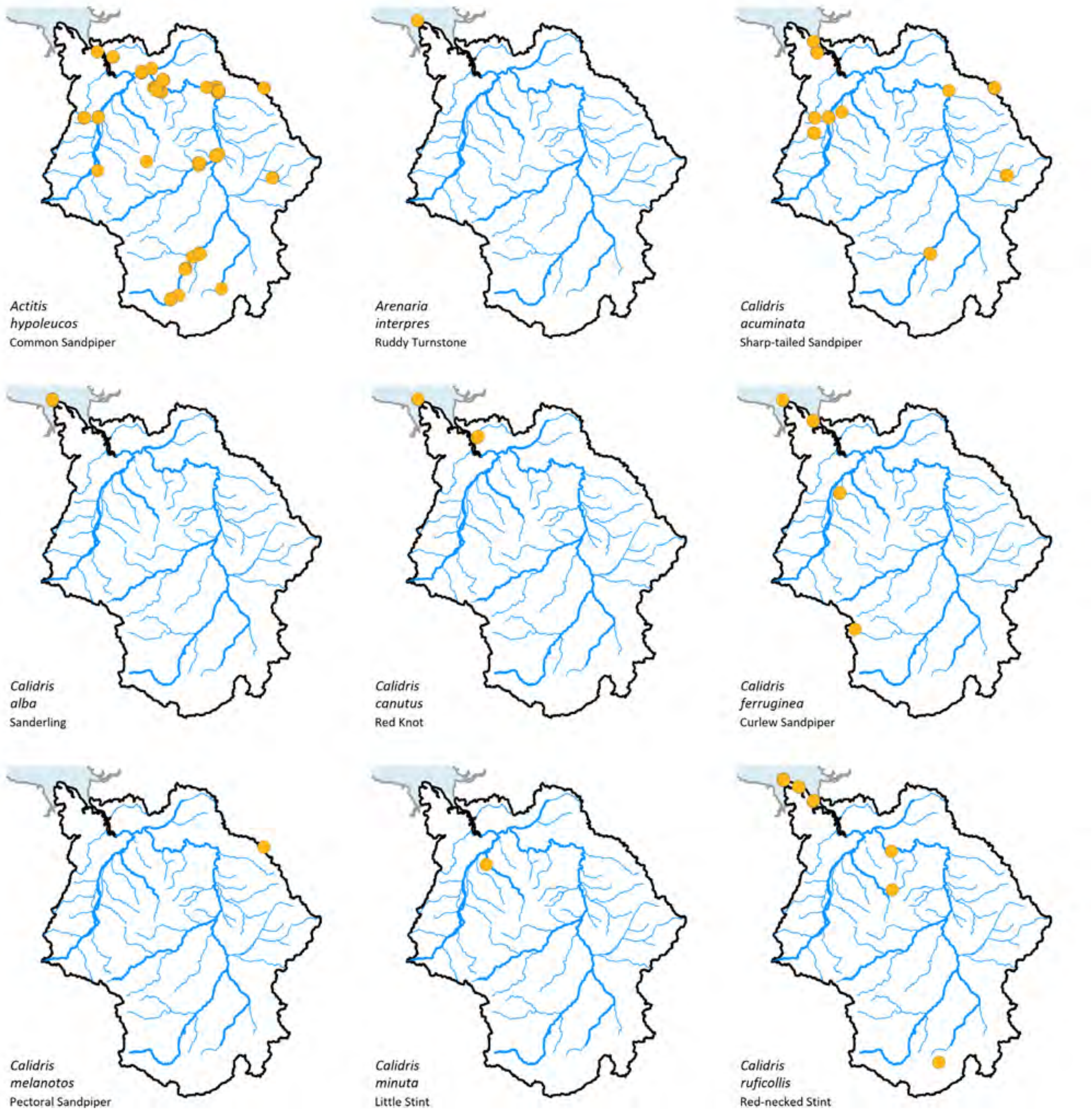

Figure 3-28 Observed locations of shorebirds in the Victoria catchment in alphabetic order of species name: *Actitis hypoleucos* (common sandpiper) to *Calidris ruficollis* (red-necked stint)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

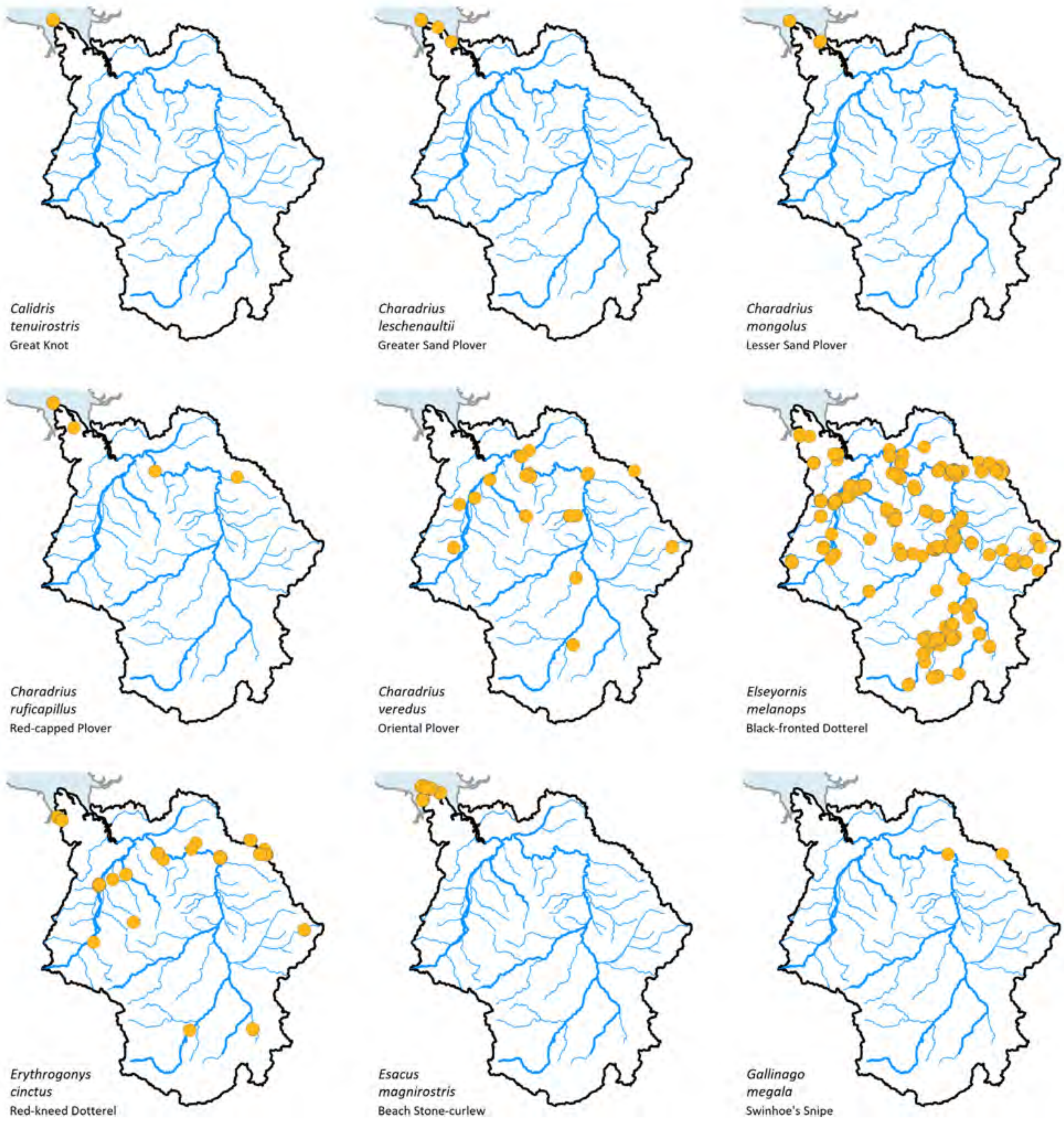

Figure 3-29 Observed locations of shorebirds in the Victoria catchment in alphabetic order of species name: *Calidris tenuirostris* (great knot) to *Gallinago megala* (Swinhoe's snipe)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

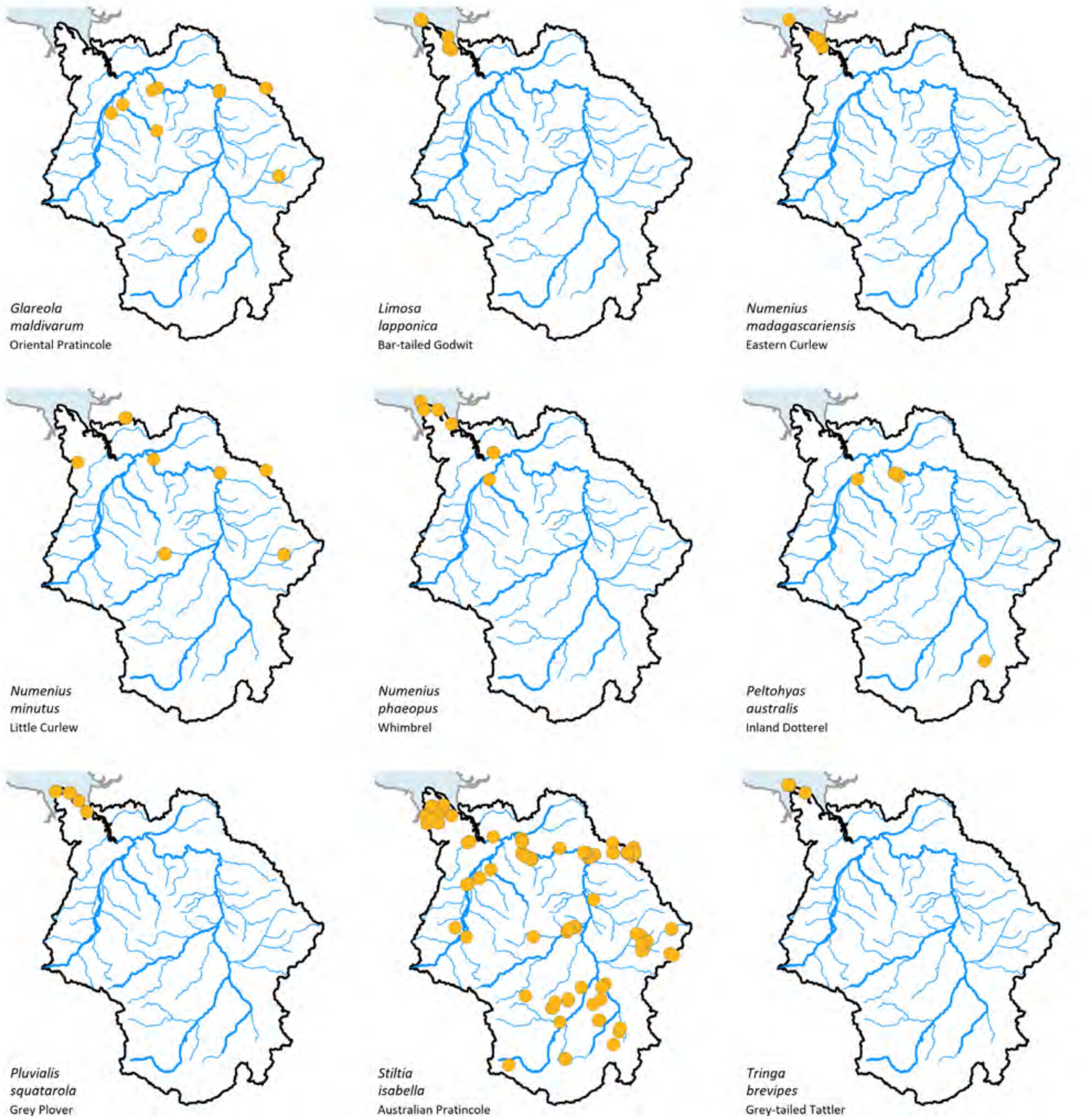

Figure 3-30 Observed locations of shorebirds in the Victoria catchment in alphabetic order of species name: *Glareola maldivarum* (Oriental pratincole) to *Tringa brevipes* (grey-tailed tattler)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

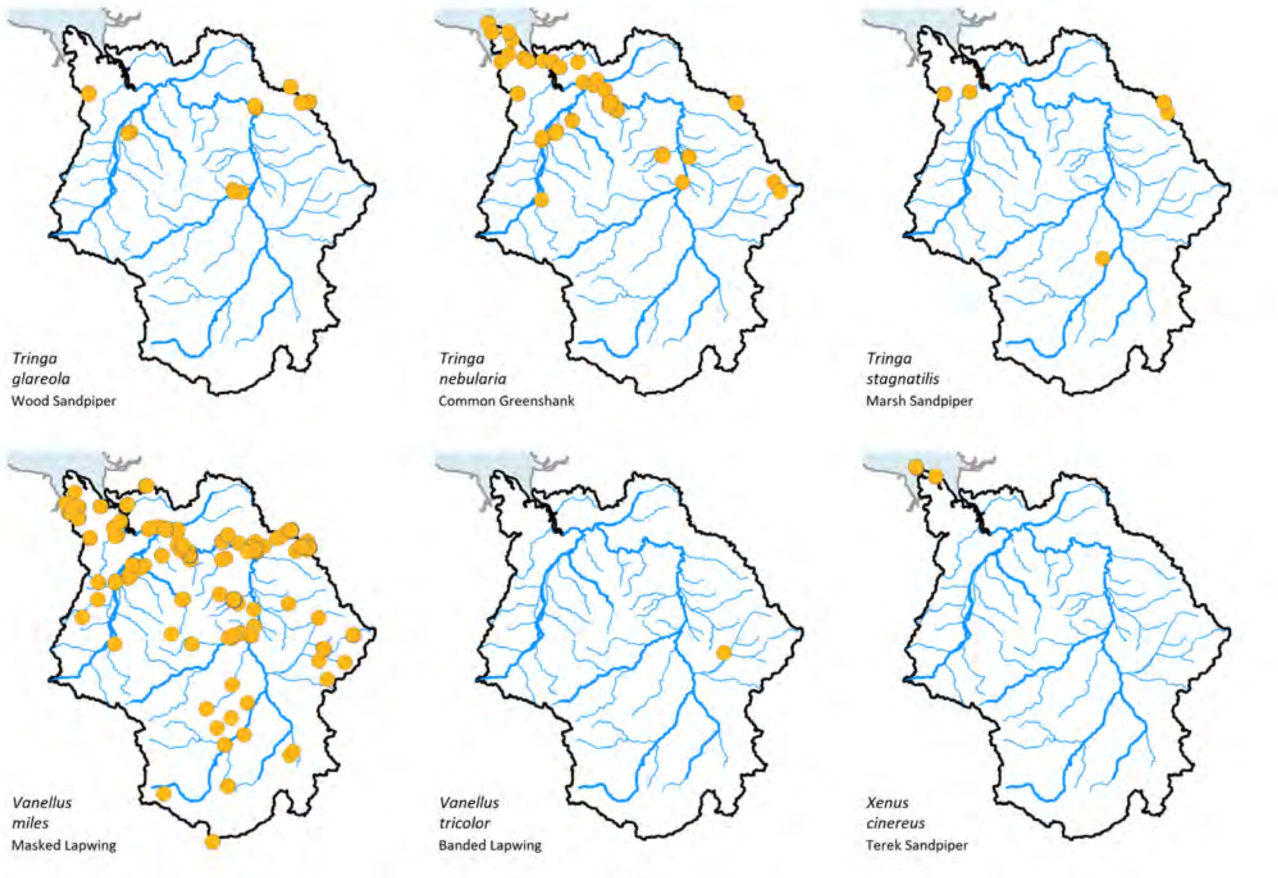

Figure 3-31 Observed locations of shorebirds in the Victoria catchment in alphabetic order of species name: *Tringa glareola* (wood sandpiper) to *Xenus cinereus* (Terek sandpiper)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

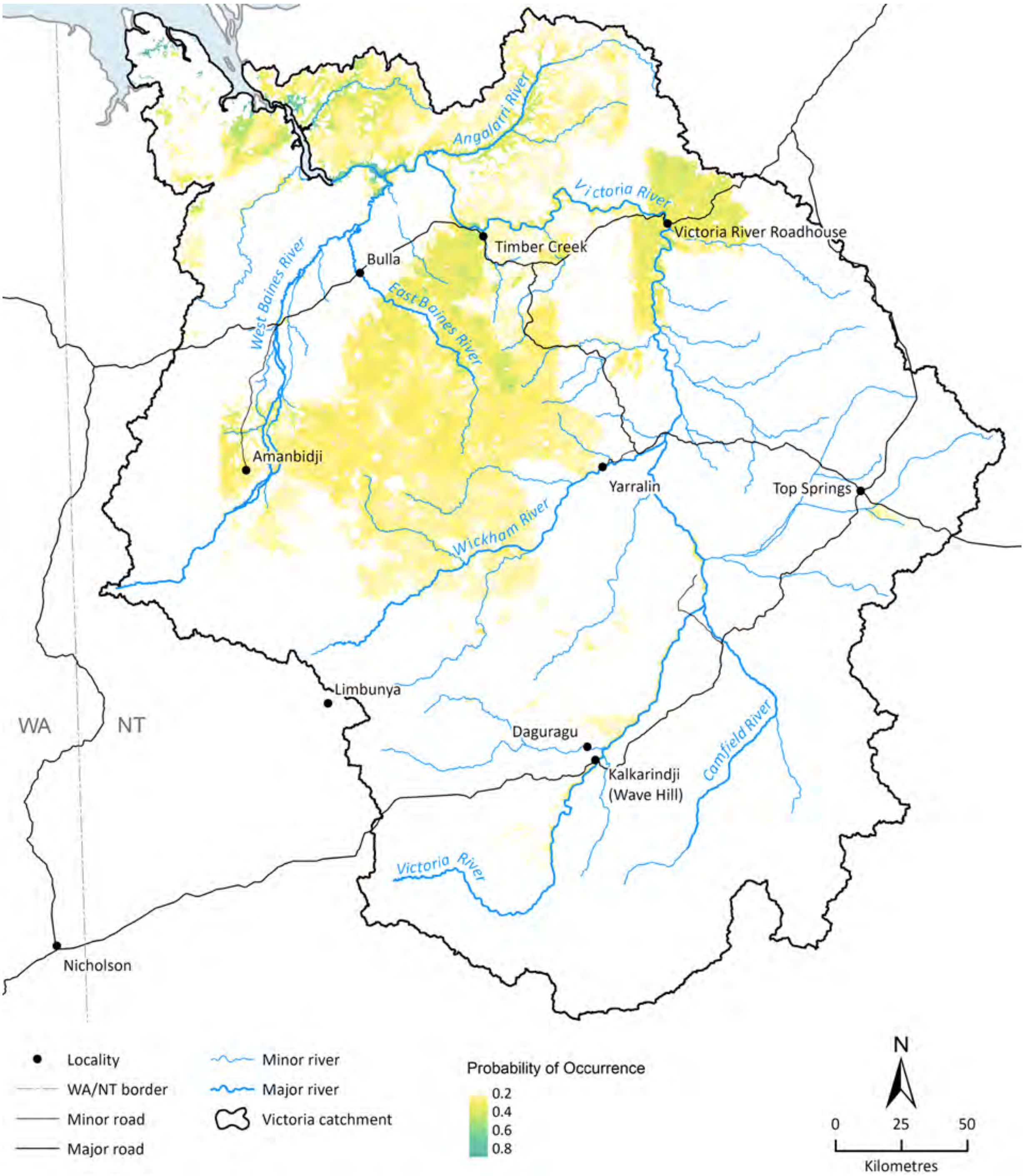

Figure 3-32 Modelled probability of occurrence of eastern curlew (*Numenius madagascariensis*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for shorebirds

Waterbird species in the shorebirds group are sensitive to changes in the depth, extent and duration of inundation of open very shallow water environments, including the edges of inland floodplains and lakes and estuarine and coastal mudflats and sandflats (Albanese and Davis, 2015; Donnelly et al.; Fernandez and Lank, 2008; Ge et al., 2009; Jackson et al., 2019; Schaffer-Smith et al., 2017). Their preference for open flat areas and good visibility when foraging means that encroachment of dense vegetation or human activity can prevent their use of a site (Baudains and Lloyd, 2007; Ge et al., 2009; Tarr et al., 2010). These species require abundant and spatially dense food, the latter being dependent on good water quality, high productivity of freshly inundated floodplain areas, and end-of-system flows to estuaries and coasts (Saint-Beat et al., 2013; Taft and Haig, 2005; 2006; Tjorve et al., 2008). The ecological functions that support shorebirds, and their associated flow requirements, are summarised in Table 3-14.

Table 3-14 Ecological functions supporting shorebirds and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT	FLOW COMPONENT OR ATTRIBUTE
Meeting water requirements for foraging, roosting and nesting	Damp sediment, shallow water or the edges of deeper water habitats. Ephemeral habitats preferred due to greater food availability	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, reproduction timing, recruitment timing Rate of change in flow events – rate of change in depth
Water regime to support required vegetation types and condition	Very low density to no vegetation	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Water regime to support suitable water quality	Low salinity, low turbidity, low toxic algae and cyanobacteria levels, low nutrient (e.g. no eutrophication)	Magnitude of flow events – inundation extent, inundation depth, flow rate when connected Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth and flow rate
Water regime to support food availability	Suitable abundance of crustaceans, invertebrates, small fish	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Competition and predation and diseases and parasites Water regime to reduce risk	Water regime to provide sufficient habitat extent to avoid overcrowding and provide multiple alternative site options to avoid predators	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth

Pathways to change for shorebirds

Shorebirds use habitats such as mudflats, sandflats, coastal bays or inlets to recover from migration flights (Atkinson, 2003; Jackson et al., 2019). Quality sites are able to support large numbers of shorebirds by providing abundant food, minimal human disturbance and shelter to rest (Goodenough et al., 2017; Pfister et al., 1992). Throughout the non-breeding season, shorebirds must increase their food intake to fuel their migration back to northern breeding sites (Goodenough et al., 2017). They require undisturbed and productive feeding areas to ensure minimal energy expenditure (Anderson et al., 2019). They rely on the inundation of shallow flat areas such as mudflats and sandflats to provide invertebrates and other food sources (Aharon-Rotman et al., 2017; Galbraith et al., 2002). Without inundation events, these habitats cannot support high densities of shorebird species, and lack of food can increase mortality rates both on-site and during and after migrations (Aharon-Rotman et al., 2017; Goss-Custard, 1977; Rushing et al., 2016).

The primary pathways of potential water resource development impact on shorebirds include: habitat loss, fragmentation and change; toxins from pollution or contaminants; and disturbance from human activities (Aharon-Rotman et al., 2016). Habitat loss and disturbance from human activities is of particular concern for shorebird species worldwide. Shorebirds may waste time and energy responding to human disturbance, which may cause temporary loss of foraging habitats. The capacity to compensate by foraging for longer periods may vary between individuals and species (Glover et al., 2011; Pfister et al., 1992; Rogers et al., 2006; St Clair et al., 2010; Tarr et al., 2010; West et al., 2002). During the breeding season, human disturbance may also influence nest incubation and chick rearing, affecting overall nest success and eventual recruitment, which then affect population sizes and trajectories (McGinness, 2016). Climate change is also affecting habitat availability and quality among other factors for shorebirds, including changing freshwater inflows and the availability of mudflats and similar environments (Bellisario et al., 2014; Iwamura et al., 2013). The ecological outcomes of threatening processes on wetlands in the Victoria catchment, and their implications for changes to biodiversity and ecosystem function, are illustrated in Figure 3-33.

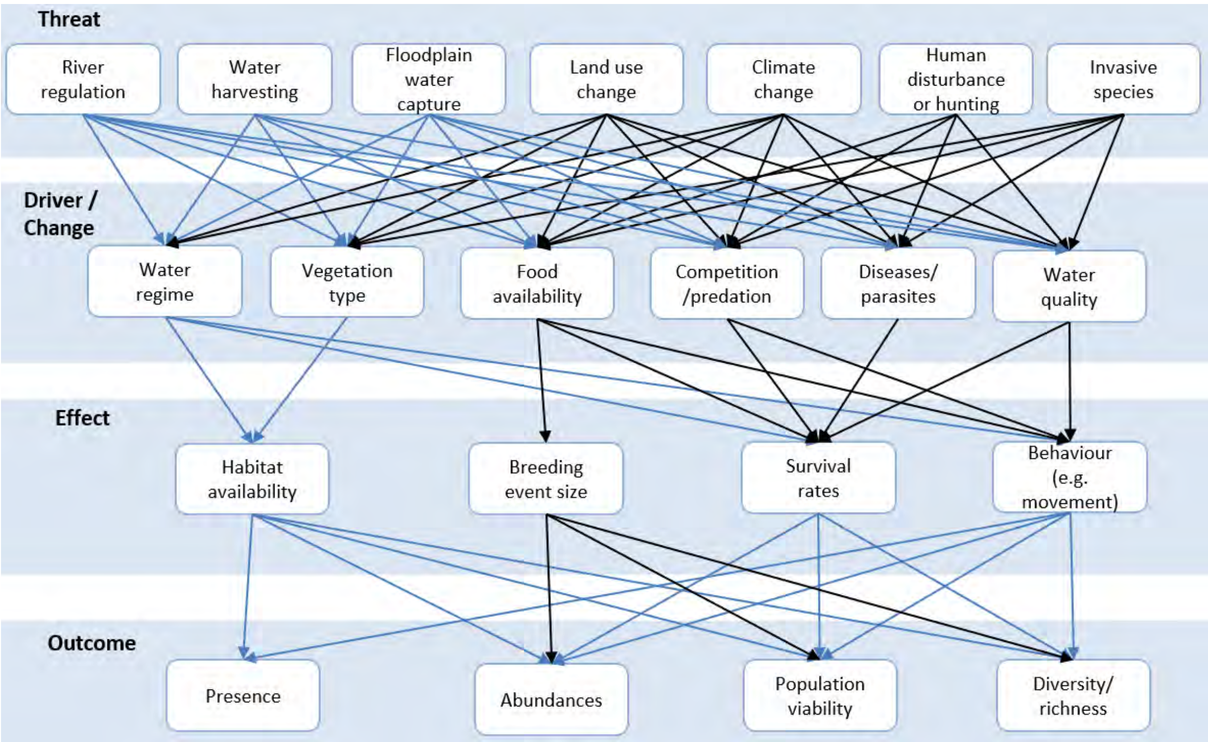

Figure 3-33 Conceptual model showing the relationship between threats, drivers, effects and outcomes for the shorebirds group in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.2.4 Swimming, grazing and diving waterbirds

Description and background to ecology

The swimmers, grazers and divers group comprises species with a relatively high level of dependence on semi-open, open and deeper water environments. These species commonly swim when foraging (including diving, filtering, dabbling, grazing) or when taking refuge. In northern Australia, this group comprises 49 species from 11 families, including ducks, geese, swans, grebes, pelicans, darters, cormorants, shags, swamphens, gulls, terns, noddies and jacanas (Table 3-15). This group can be further broken down into the subgroups:

- diving swimmers – e.g. cormorants, pelicans, grebes
- aerial divers – e.g. terns, gulls, noddies
- grazing swimmers – e.g. swans, coots, swamphens, ducks, geese.

These species breed in Australia and may be sedentary, nomadic, migratory or partially migratory. Nesting generally occurs seasonally, usually in dense vegetation such as emergent macrophytes, trees and shrubs (Garnett et al., 2015). Nests are usually independent or semi-colonial, and while breeding is usually seasonal, it can be stimulated by flooding or large rainfall events (Kingsford and Norman, 2002). Species diets may be piscivorous, omnivorous or herbivorous (Barker and Vestjens, 1990). Changes in water depth, water extent or water duration can expose nests to predation or drowning of nests, or reduce food availability, resulting in nest failure (McGinness, 2016; Poiani, 2006).

The magpie goose (*Anseranas semipalmata*; Figure 3-34) is one example of the swimmers, grazers and divers group, and while it is an iconic species in northern Australia, it is also the source of some conflict with humans when resources are limited (Corriveau et al., 2022; Frith and Davies, 1961; Traill et al., 2010). The magpie goose is an ancient and unique species of particular importance to Indigenous Peoples, providing eggs, meat and feathers. This species feeds on aquatic vegetation and often nests colonially (Marchant and Higgins, 1990). While currently abundant in northern Australia, wild magpie goose populations have largely disappeared from southern Australia due to human-driven change such as habitat destruction and hunting (Nye et al., 2007), and climate change is likely to exacerbate the impacts of such change on magpie geese in northern Australia (Poiani, 2006; Traill et al., 2009a).

Table 3-15 Species in the swimming, grazing and diving waterbirds group, and their national and international conservation status

(LC = Least concern)

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Australian (Australasian) shoveler	*Anas rhynchotis (Spatula rhynchotis)*	Anatidae	Australian	LC	LC
Australian wood duck (maned duck)	*Chenonetta jubata*	Anatidae	Endemic	LC	LC
Spotted whistling-duck	*Dendrocygna guttata*	Anatidae	Australian	LC	LC
Garganey, Garganey teal	*Spatula querquedula (Anas querquedula)*	Anatidae	Non-breeding migrant	LC	LC
Freckled duck	*Stictonetta naevosa*	Anatidae	Endemic	LC	LC
Chestnut teal	*Anas castanea*	Anatidae	Endemic	LC	LC
Grey teal	*Anas gracilis*	Anatidae	Australian	LC	LC
Pacific black duck	*Anas superciliosa*	Anatidae	Australian	LC	LC
Hardhead	*Aythya australis*	Anatidae	Australian	LC	LC
Black swan	*Cygnus atratus*	Anatidae	Endemic	LC	LC
Wandering whistling-duck	*Dendrocygna arcuata*	Anatidae	Australian	LC	LC
Plumed whistling-duck	*Dendrocygna eytoni*	Anatidae	Australian	LC	LC
Pink-eared duck	*Malacorhynchus membranaceus*	Anatidae	Endemic	LC	LC
Cotton pygmy-goose	*Nettapus coromandelianus*	Anatidae	Australian	LC	LC
Green pygmy-goose	*Nettapus pulchellus*	Anatidae	Australian	LC	LC
Blue-billed duck	*Oxyura australis*	Anatidae	Endemic	LC	LC
Radjah shelduck	*Radjah radjah (Tadorna radjah)*	Anatidae	Australian	LC	LC
Australian shelduck	*Tadorna tadornoides*	Anatidae	Endemic	LC	LC
Australasian darter	*Anhinga novaehollandiae*	Anhingidae	Australian	LC	LC
Magpie goose	*Anseranas semipalmata*	Anseranatidae	Australian	LC	LC
Comb-crested jacana	*Irediparra gallinacean*	Jacanidae	Australian	LC	LC

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Common noddy	*Anous stolidus*	Laridae	Australian	LC	LC
Whiskered tern	*Chlidonias hybrida*	Laridae	Australian	LC	LC
White-winged black tern	*Chlidonias leucopterus*	Laridae	Non-breeding migrant	LC	LC
Silver gull	*Chroicocephalus novaehollandiae*	Laridae	Australian	LC	LC
Australian gull-billed tern	*Gelochelidon macrotarsa*	Laridae	Endemic (breeding only)	LC	LC
Common gull-billed tern	*Gelochelidon nilotica*	Laridae	Non-breeding migrant	LC	LC
Caspian tern	*Hydroprogne caspia*	Laridae	Australian	LC	LC
Bridled tern	*Onychoprion anaethetus*	Laridae	Australian	LC	LC
Sooty tern	*Onychoprion fuscatus (Onychoprion fuscata)*	Laridae	Australian	LC	LC
Roseate tern	*Sterna dougallii*	Laridae	Australian	LC	LC
Common tern	*Sterna hirundo*	Laridae	Non-breeding migrant	LC	LC
Black-naped tern	*Sterna sumatrana*	Laridae	Australian	LC	LC
Little tern	*Sternula albifrons*	Laridae	Australian	LC	LC
Lesser crested tern	*Thalasseus bengalensis*	Laridae	Australian	LC	LC
Crested tern	*Thalasseus bergii*	Laridae	Australian	LC	LC
Australian pelican	*Pelecanus conspicillatus*	Pelecanidae	Endemic (breeding only)	LC	LC
Little pied cormorant	*Microcarbo melanoleucos*	Phalacrocoracidae	Australian	LC	LC
Great cormorant	*Phalacrocorax carbo*	Phalacrocoracidae	Australian	LC	LC
Little black cormorant	*Phalacrocorax sulcirostris*	Phalacrocoracidae	Australian	LC	LC
Pied cormorant	*Phalacrocorax varius*	Phalacrocoracidae	Australian	LC	LC
Great crested grebe	*Podiceps cristatus*	Podicipedidae	Australian	LC	LC
Hoary-headed grebe	*Poliocephalus poliocephalus*	Podicipedidae	Endemic	LC	LC
Australasian grebe	*Tachybaptus novaehollandiae*	Podicipedidae	Australian	LC	LC
Pale-vented bush-hen, Bush-hen	*Amaurornis moluccana*	Rallidae	Australian	LC	LC

SPECIES NAME	SPECIES SCIENTIFIC NAME	FAMILY SCIENTIFIC NAME	POPULATION TYPE	IUCN STATUS	AUSTRALIAN CONSERVATION STATUS
Eurasian coot	*Fulica atra*	Rallidae	Australian	LC	LC
Dusky moorhen	*Gallinula tenebrosa*	Rallidae	Australian	LC	LC
Purple swamphen	*Porphyrio porphyrio*	Rallidae	Australian	LC	LC
Black-tailed native-hen	*Tribonyx ventralis*	Rallidae	Endemic	LC	LC

Figure 3-34 Magpie goose perched on a fallen tree branch

Magpie geese are a representative species of the swimmers, grazers and divers group.

Photo attribution: CSIRO

Swimmers, grazers and divers in the Victoria catchment

In the Victoria catchment, the Legune Homestead Swamps (part of the Legune Wetlands) are important habitat for a range of waterbirds, including as a breeding site for magpie geese (Department of Agriculture, Water and the Environment, 2021). The maximum water depth in the Legune Homestead Swamps ranges from 1.5 to 2 m (Department of Agriculture, Water and the Environment, 2021), making the habitat suitable for birds in this group. Aerial surveys by Chatto (2006) found that magpie geese were the most abundant waterbird in the Legune coastal floodplain area. Other common birds found in these surveys included wandering whistling-duck (*Dendrocygna arcuata*), grey teal (*Anas gracilis*), plumed whistling-duck (*Dendrocygna eytoni*), hardhead (*Aythya australis*), white-winged black tern (*Chlidonias leucopterus*), Pacific black duck (*Anas superciliosa*) and the purple swamphen (*Porphyrio porphyrio*) (Chatto, 2006). Within the broader Victoria catchment, deeper stretches of river are also suitable for this group (Figure 3-35, Figure 3-36, Figure 3-37 and Figure 3-38). The modelled distribution of the magpie goose is shown in Figure 3-39.

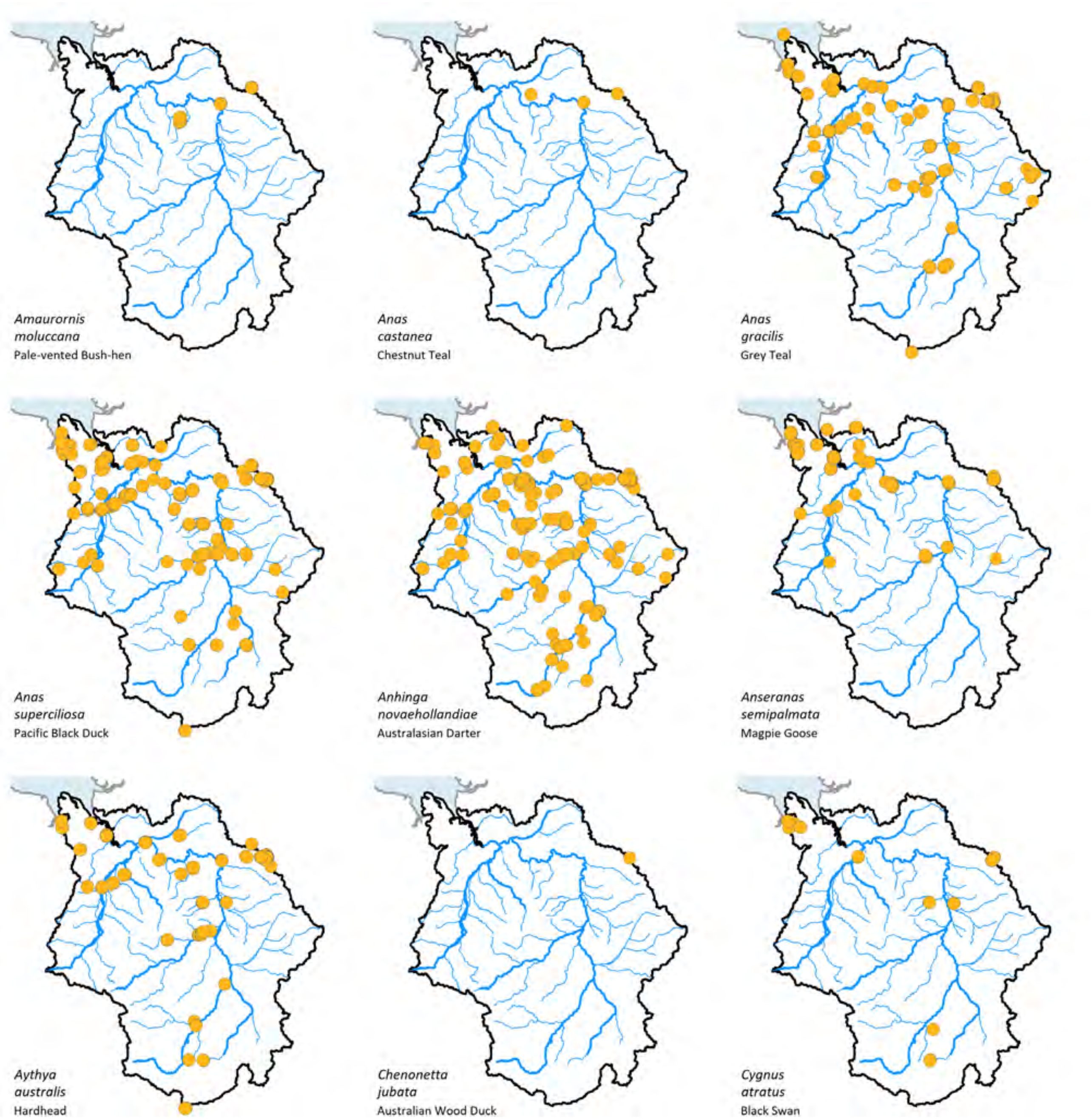

Figure 3-35 Observed locations of swimming, grazing and diving waterbirds in the Victoria catchment in alphabetic order of species name: *Amaurornis moluccana* (pale-vented bush-hen) to *Cygnus atratus* (black swan)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

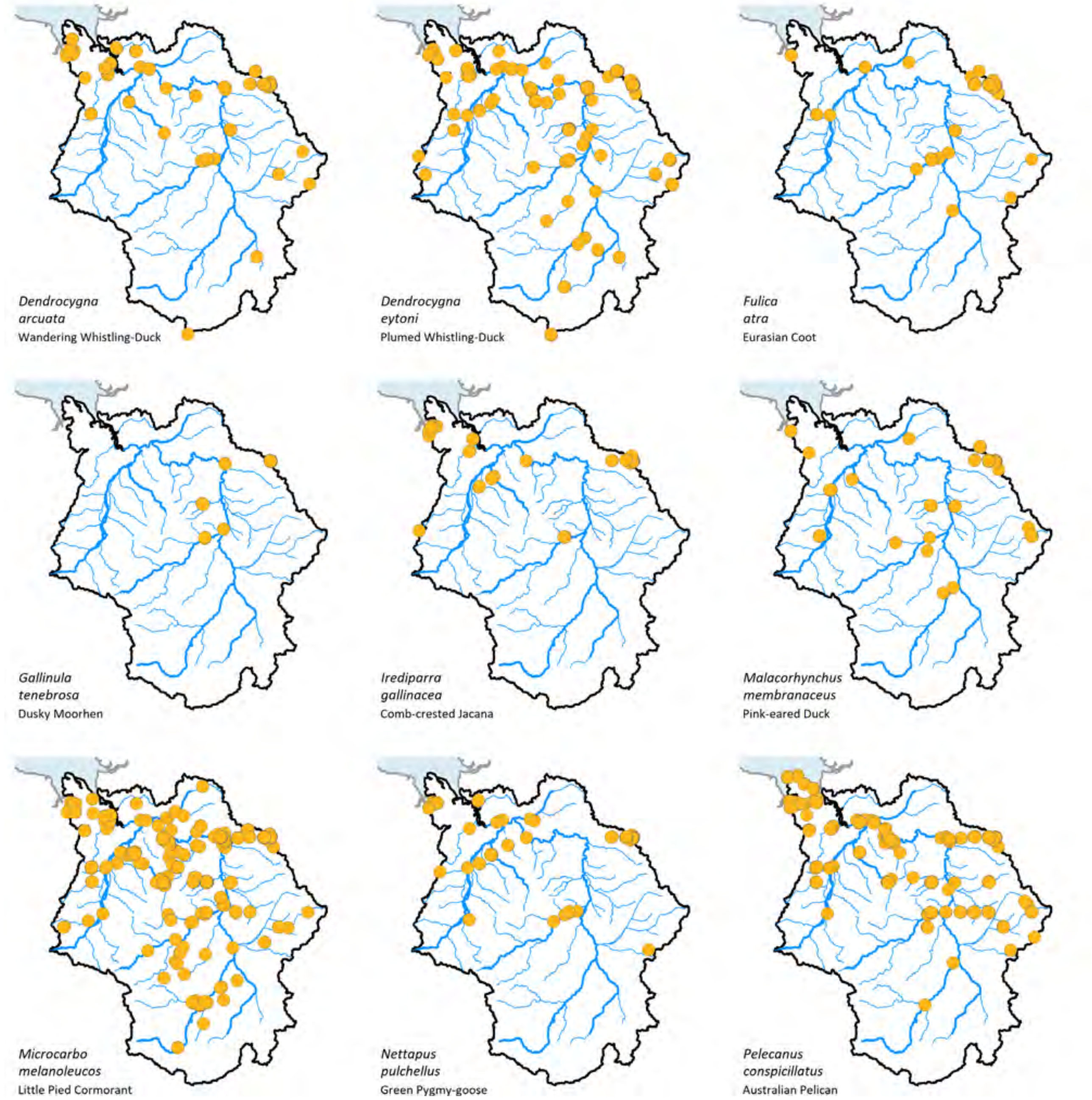

Figure 3-36 Observed locations of swimming, grazing and diving waterbirds in the Victoria catchment in alphabetic order of species name: *Dendrocygna arcuata* (wandering whistling-duck) to *Pelecanus conspicillatus* (Australian pelican)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

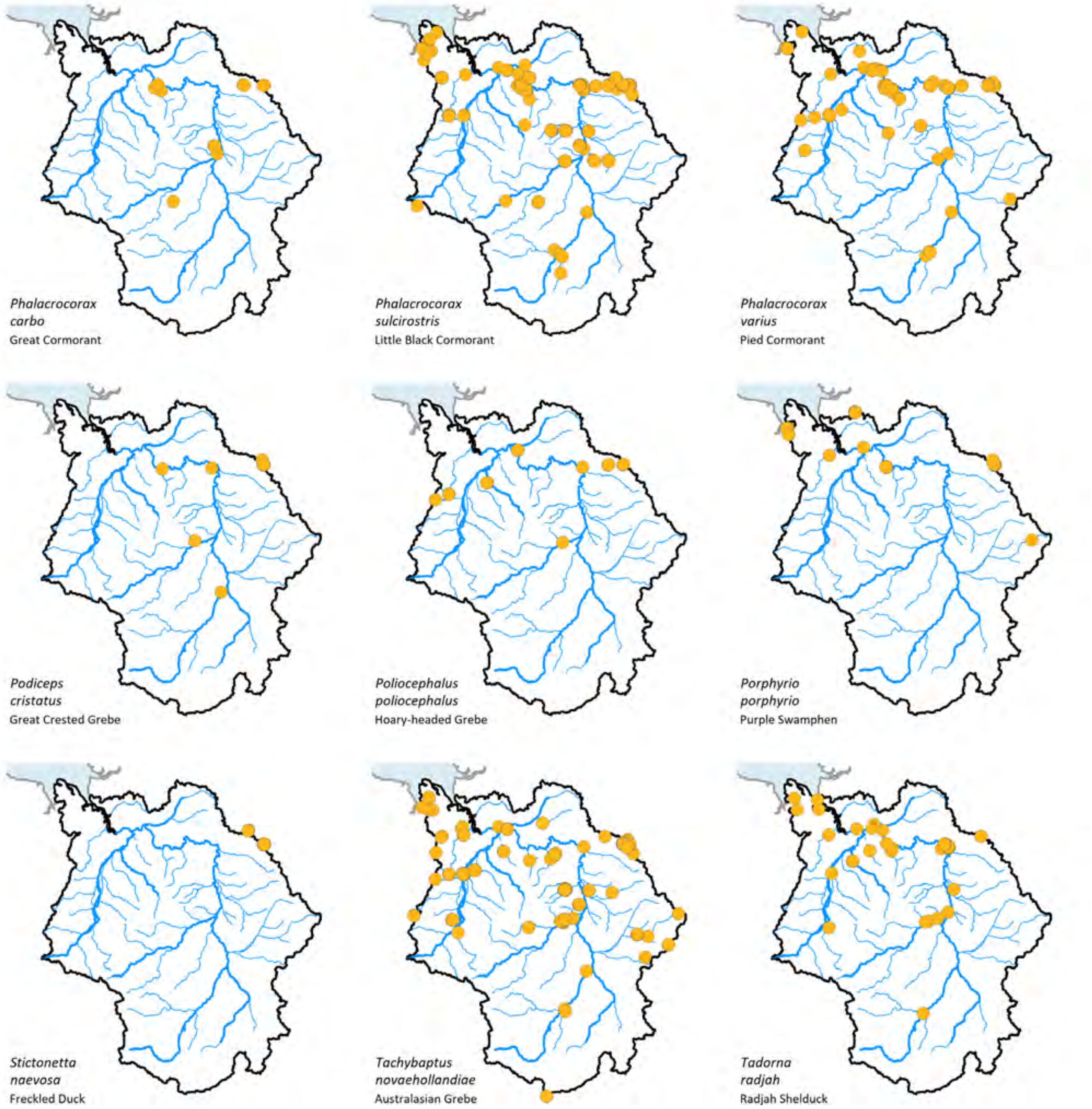

Figure 3-37 Observed locations of swimming, grazing and diving waterbirds in the Victoria catchment in alphabetic order of species name: *Podiceps cristatus* (great crested grebe) to *Tadorna radjah* (Radjah shelduck)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

Figure 3-38 Observed locations of swimming, grazing and diving waterbirds in the Victoria catchment: *Tribonyx ventralis* (black-tailed native-hen)

Map tiles include species for which there is data in the ALA.

Data source: Atlas of Living Australia (2023)

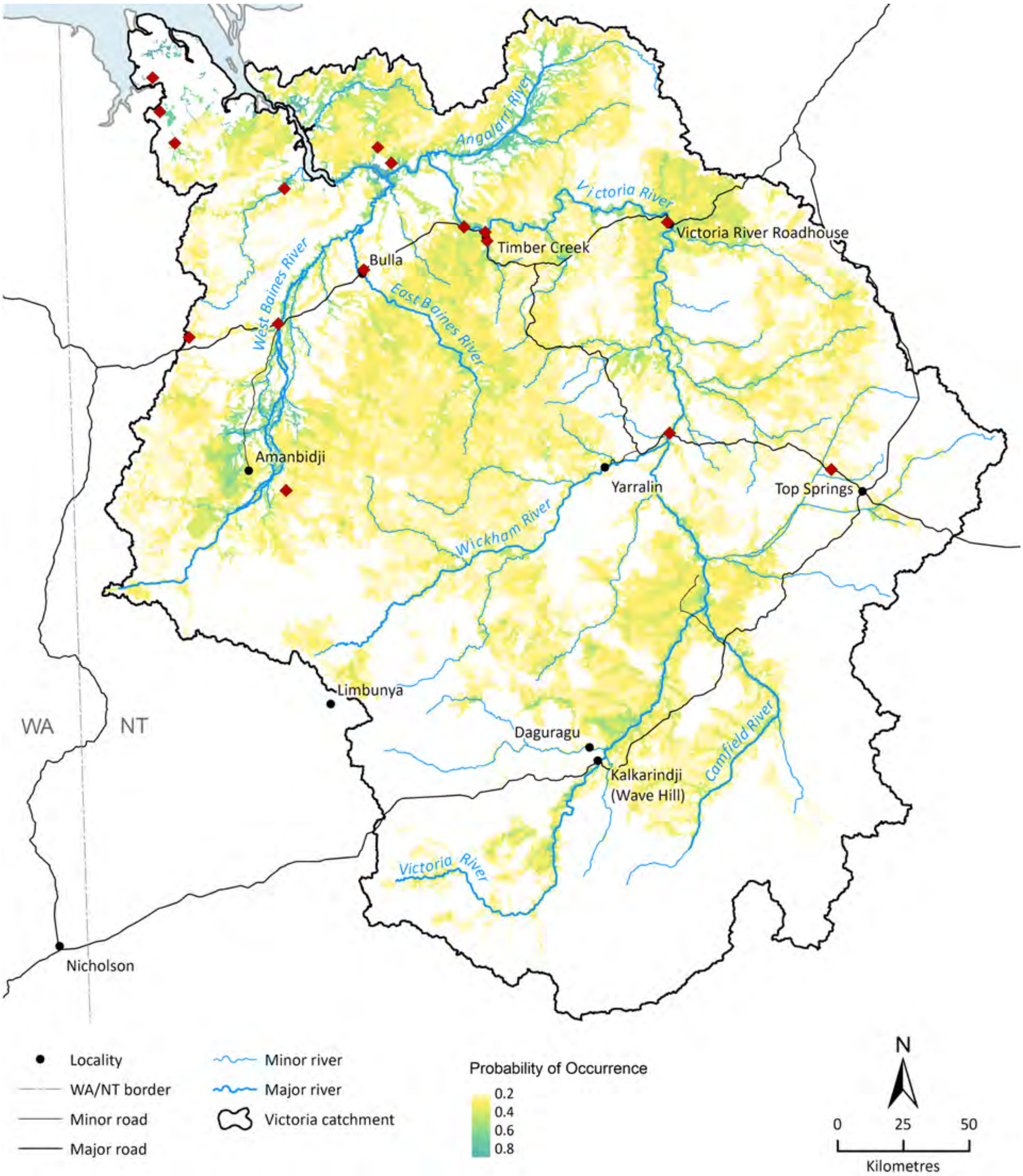

Figure 3-39 Modelled probability of occurrence of magpie goose (*Anseranas semipalmata*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for swimming, grazing and diving waterbirds

Species in the swimming, grazing and diving waterbirds group are sensitive to changes in the depth, extent and duration of perennial semi-open and open deeper water environments such as waterholes and wetlands (Table 3-16) (Marchant and Higgins, 1990; McGinness, 2016). They can also be sensitive to changes in the type, density or extent of the fringing aquatic or semi-aquatic vegetation in and around these habitats. Besides changing foraging, nesting and refuge habitat, such changes can also reduce water quality and food availability and increase rates of competition, predation and disease (Douglas et al., 2005; McGinness, 2016). Such changes can occur when water is extracted directly from these habitats or when the time between connecting flows or rainfall events that fill these habitats is extended (Kingsford and Norman, 2002). Climate change and extremes are likely to interact with changes induced by water resource development, including inundation of freshwater habitats by seawater and inundation of nests by extreme flood events or seawater intrusion (Nye et al., 2007; Poiani, 2006; Traill et al., 2009a; Traill et al., 2009b). The ecological functions that support swimming, grazing and diving waterbirds, and their associated flow requirements, are summarised in Table 3-16.

Table 3-16 Ecological functions supporting swimming, grazing and diving waterbirds and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT	FLOW COMPONENT OR ATTRIBUTE
Meeting water requirements for foraging, roosting and nesting	Deep, semi-permanent to permanent. No sudden changes in water depth during nesting periods	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, reproduction timing, recruitment timing Rate of change in flow events – rate of change in depth
Water regime to support required vegetation types and condition	Dense fringing and medium density submerged and floating aquatic and semi-aquatic	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Water regime to support suitable water quality	Low salinity, low turbidity, low toxic algae and cyanobacteria levels, low nutrient (e.g. no eutrophication)	Magnitude of flow events – inundation extent, inundation depth, flow rate when connected Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth and flow rate
Water regime to support food availability	Suitable abundance of fish, crustaceans, molluscs, invertebrates, frogs, tadpoles, aquatic and semi-aquatic plants	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth
Competition and predation and diseases and parasites Water regime to reduce risk	Water regime to provide sufficient habitat extent to avoid overcrowding and provide multiple alternative site options to avoid predators	Magnitude of flow events – inundation extent, inundation depth Frequency of flow events – frequency that habitat is dry, shallow or deep Duration of flow events – duration wet, duration dry, shallow or deep Timing of flow events – season, growth periods, reproduction timing Rate of change in flow events – rate of change in depth

Pathways to change for swimming, grazing and diving waterbirds

The primary pathways of potential water resource development impact on waterbirds in the swimmers, grazers and divers group include: (i) habitat loss, fragmentation and change, including water depth changes and weed invasion changing habitats, (ii) climate change and extremes – including inundation of freshwater habitats by seawater when river flows are reduced and inundation of nests by extreme flood events, (iii) toxins from pollution or contaminants, (iv) disturbance and hunting from human activities, (v) predation by invasive or feral animals, and (vi) changes in disease or parasite risk or burdens (Bayliss, 1989; Corbett and Hertog, 1996; Douglas et al., 2005; Morton, 1990; Nye et al., 2007; Poiani, 2006; Traill et al., 2010; Traill et al., 2009a; Traill et al., 2009b) (Figure 3-40). Reduced extent, depth and duration of inundation of waterhole and other deep-water environments are likely to reduce habitat and food availability for this group, increasing competition and predation and also increasing risk of disease and parasite spread. Conversely, species in this group that nest at water level or just above, such as magpie geese, are particularly at risk of nests drowning when water depths increase unexpectedly (Douglas et al., 2005; Poiani, 2006; Traill et al., 2010; Traill et al., 2009a; Traill et al., 2009b).

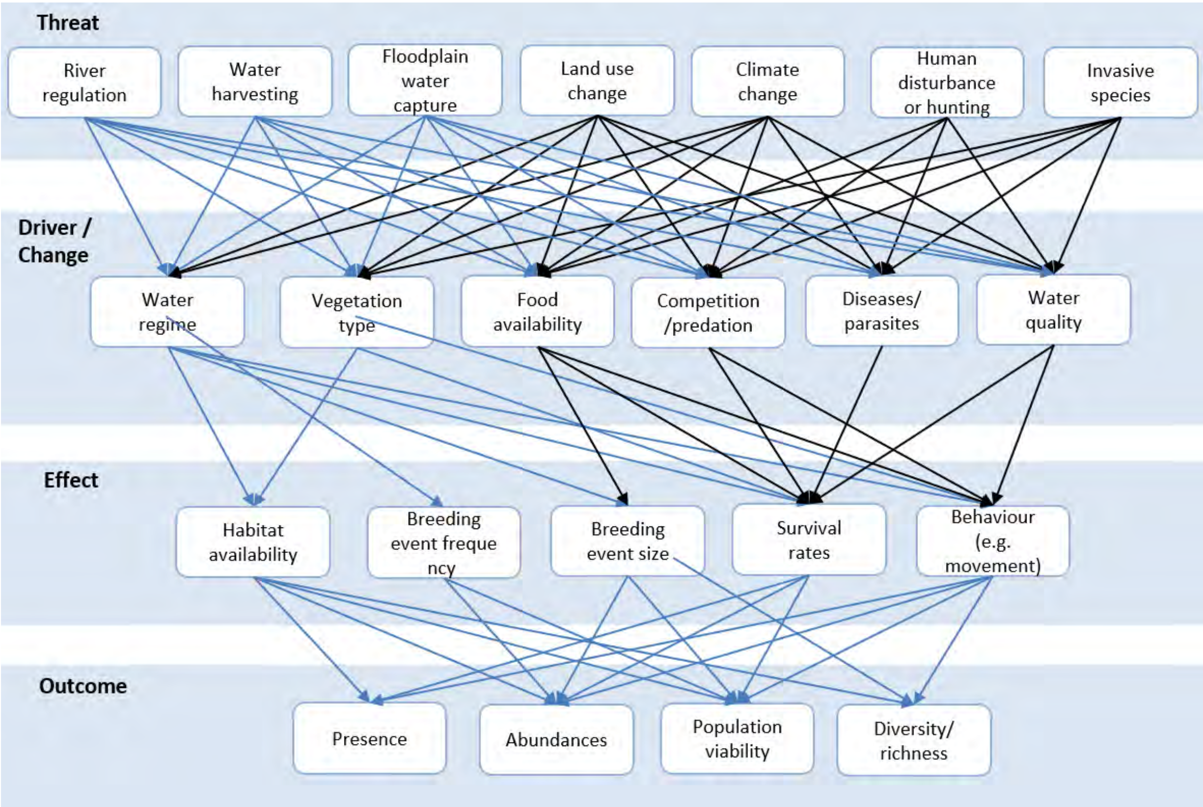

Figure 3-40 Conceptual model showing the relationship between threats, drivers, effects and outcomes for the swimming, grazing and diving waterbirds group in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.3 Turtles, prawns and other species

3.3.1 Banana prawns (*Penaeus indicus* and *P. merguiensis*)

Description and background ecology

Banana prawns are large-bodied decapod crustaceans around 80 g in weight of the family Penaeidae that are found throughout the Indo-West Pacific. They are a prized fishery target species throughout their geographic distribution. Two species of banana prawns are found in Australia: the common banana prawn (*Penaeus merguiensis*) and the redleg banana prawn (*Penaeus indicus*). Both banana prawn species are globally widespread throughout the Indian Ocean and south-east Asian and west Pacific coastal habitats. In the Joseph Bonaparte Gulf and the estuary of the Victoria River, redleg banana prawns are the dominant species (Kenyon et al., 2004; Plagányi et al., 2021). Joseph Bonaparte Gulf and the western Tiwi Island region in north-west Australia are the south-eastern limit of the worldwide distribution of redleg banana prawns (Grey et al., 1983). In contrast, across tropical and subtropical Australia, common banana prawns are widespread in tropical and subtropical coastal waters (Grey et al., 1983).

Post-larval banana prawns settle in the mudbank and mangrove forest matrix in the upper reaches of estuarine tributaries (Kenyon et al., 2004; Vance et al., 1996; 2002). They occupy mangrove forest habitats (see Section 3.4.4) and are forced from the mangroves on each ebb tide, to return on the next flood tide (Vance et al., 2002). Mangrove prop roots and trunks are critical to juvenile banana prawn survival; they provide shelter and refuge from predation (Meager et al., 2005). The substrates within the forest and on the intertidal banks support microflora and meiofauna (algae, molluscs, crustaceans and annelid worms), which they consume on each tide (Burford et al., 2012; Duggan et al., 2014; Vance et al., 2002; Wassenberg and Hill, 1993).

Using commercial catch as a measure of population abundance, large flood flows cue the prolific estuarine population of juvenile redleg banana prawns to emigrate to offshore zones where they rely on marine habitats for enhanced growth and survival (Plagányi et al., 2021). However, emigration of redleg banana prawns was not determined by high rainfall and flood flows alone, a Southern Oscillation Index value greater than 7 (typical of La Niña years) in combination with high wet-season rainfall was a good predictor of higher catches of redleg banana prawns (Plagányi et al., 2021). Freshwater flows alone are more clear as a migration cue for common banana prawns (Broadley et al., 2020; Duggan et al., 2019; Lucas et al., 1979).

Juvenile redleg banana prawns do not tolerate freshwater estuaries (Kumlu and Jones, 1995). In the Ord River, runoff from the Ord River Irrigation Area constantly flowed downstream and caused the Ord River estuary to be perennially fresh to brackish. Compared to similar estuaries nearby, the Ord River estuary had a much lower abundance of juvenile redleg banana prawns (Kenyon et al., 2004). In all likelihood, the tolerance of juvenile redleg banana prawns to estuarine habitats would be similar to that of common banana prawns, which declines as prawns grow (Dall, 1981). Under salinity declines due to flood flows, fewer large juvenile prawns can tolerate the low salinity waters to reside in estuaries. Juvenile common banana prawns with a carapace length (CL) greater than 12 mm emigrate when salinity is about 30 to 35 ppt, while prawns less than or equal to 8 mm CL emigrate when salinity drops to about 5 ppt or lower, particularly when the decline was abrupt (Staples and Vance, 1986). Emigrants move offshore to reside on muddy sediments in deeper waters (Staples, 1980; Staples and Vance, 1986; Vance et al., 1998).

Adult banana prawn distribution is adjacent to their juvenile estuarine mangrove habitats (Plagányi et al., 2021; Staples et al., 1985; Zhou et al., 2015). Adult redleg banana prawns are found in waters 60 to 80 m deep in the north-western portion of the Victoria catchment marine region. They emigrate large distances from their juvenile estuarine habitat locations such as the Victoria River and Cambridge Gulf (Plagányi et al., 2021). Adult common banana prawns occupy soft-sediment substrates in relatively shallow waters within the south-west, south-east and eastern Gulf of Carpentaria, and along the Top End / Arnhem Land coastline. Banana prawns are managed by limited effort (licence to fish) and by spatial and temporal closures. The fishing season opens on 1 April annually and continues until catch rates decline to a trigger level defined in the Northern Prawn Fishery Harvest Strategy (AFMA, 2022).

Common banana prawns grow to about 55 mm CL for females (50 mm CL = ~85 g) and about 47 mm CL for males (40 mm CL = ~50 g). In Joseph Bonaparte Gulf, redleg banana prawns were a similar size to the common banana prawns: a growth and mortality study in the gulf found the largest female prawn tagged was 46.3 mm CL and the largest male prawn was 35.7 mm CL (Kenyon et al., 1999). The tagged prawns would not be the largest individuals in the offshore habitat.

Banana prawns in the Victoria catchment marine region

Redleg banana prawns are fished in deeper waters in north-west Joseph Bonaparte Gulf (~60–80 m), offshore from the Victoria River (Plagányi et al., 2021) (Figure 3-41). Over the past 16 years, the average tonnage of redleg banana prawns was 286 t, comprising 6.3% of the total banana prawn catch within the Northern Prawn Fishery (Laird, 2021). The estuary of the Victoria River and nearby coastal creeks in the vicinity of the Fitzmaurice and the Keep rivers in eastern coastal Joseph Bonaparte Gulf supported high abundances of juvenile redleg banana prawns (Kenyon et al., 2004). The prawns are also found in river estuaries flowing into Cambridge Gulf.

From October to December, high abundances of juvenile redleg banana prawns were found in mangrove forest and mudbank tributary habitats of the Victoria River and the adjacent Forsyth Creek (Kenyon et al., 2004). No seasonal studies of juvenile prawn abundance have been undertaken in the Victoria River, though recruitment trends of co-generic species in the Northern Prawn Fishery suggest post-larval and juvenile prawns would be abundant in the estuaries from September to March annually. The estuarine and coastal waters in the Victoria catchment marine region are highly turbid, and the mangrove prop-root structure and high turbidity provide protection from predators (Kenyon et al., 2004; Meager et al., 2005).

Juvenile common banana prawns were also found in the Victoria River Forsyth Creek and the Keep River, which form part of an extensive mangrove forest / estuarine creek delta habitat within the Victoria catchment marine region. However, common banana prawns made up less than 4% of the total juvenile banana prawn catch within coastal habitats (Kenyon et al., 2004), and in some estuaries less than 2%. Adult common banana prawns were caught within the offshore fishing zone within the Victoria catchment marine region Figure 3-42, but their proportion of the catch was low (Vance and Rothlisberg, 2020). Compared to the Gulf of Carpentaria, studies of the Joseph Bonaparte Gulf prawn stocks are scant, and the two species are not separated in the commercial catch. However, a 2009 and 2010 study of the biology of adult redleg banana prawns with a prawn tagging component showed that common banana prawns made up a small proportion of the catch (<0.1%) (R. Kenyon, 2023, unpublished data). Common banana prawns were caught in western

Joseph Bonaparte Gulf outside the Victoria catchment marine region (Laird, 2021; Vance and Rothlisberg, 2020).

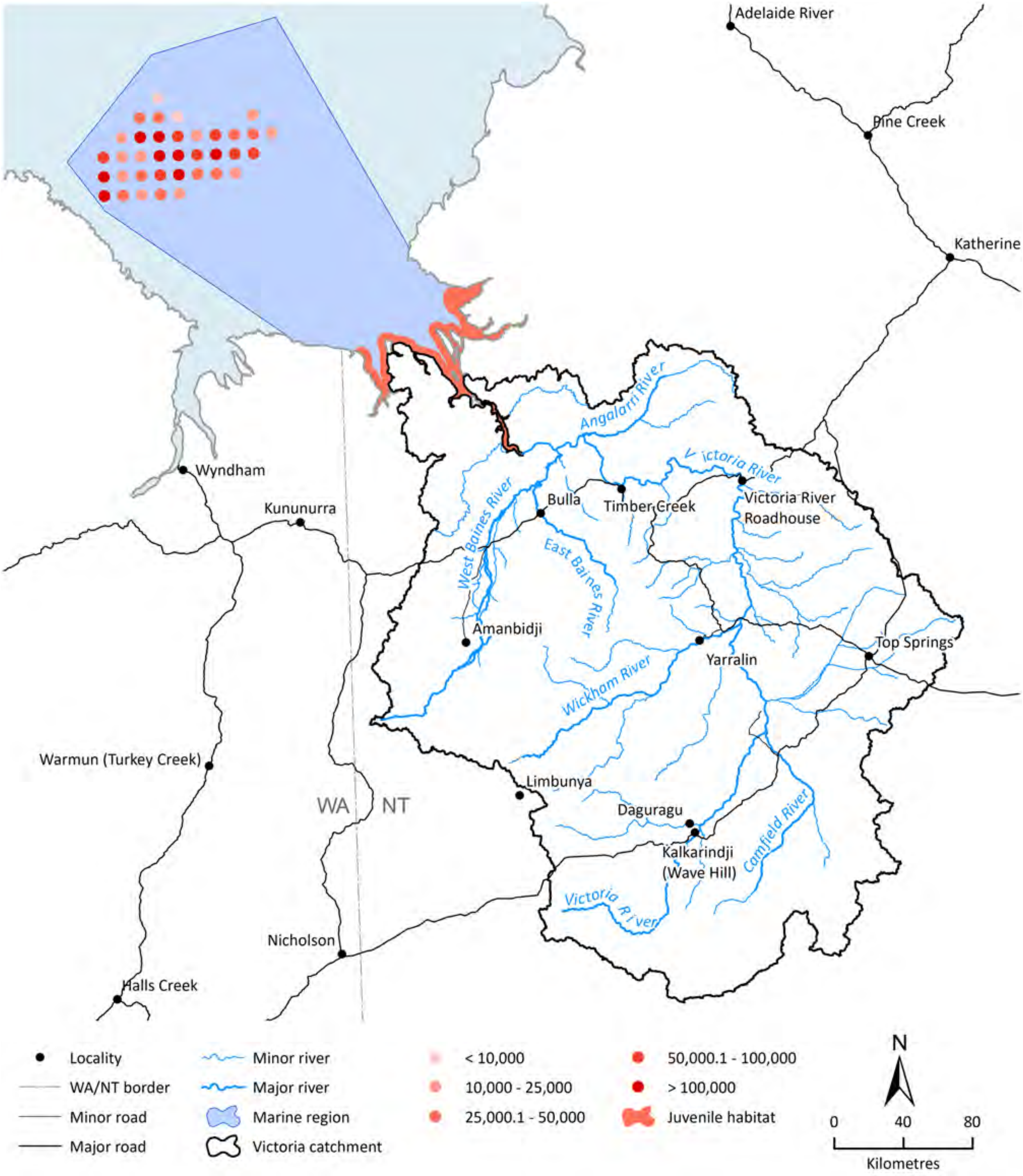

Figure 3-41 Fisheries catch of redleg banana prawns in the Victoria catchment marine region

Redleg banana prawn juveniles use tropical river estuaries as nursery habitat, particularly the mangrove forest / tributary creek matrix. Adult redleg banana prawns are caught offshore in water about 60 to 80 m deep in the marine habitat, approximately 200 km distant from their juvenile habitat.

Units are kilograms as total catches for the 10-year period 2011 to 2020.

Data sources: Kenyon et al. (2022); Kenyon et al. (2004)

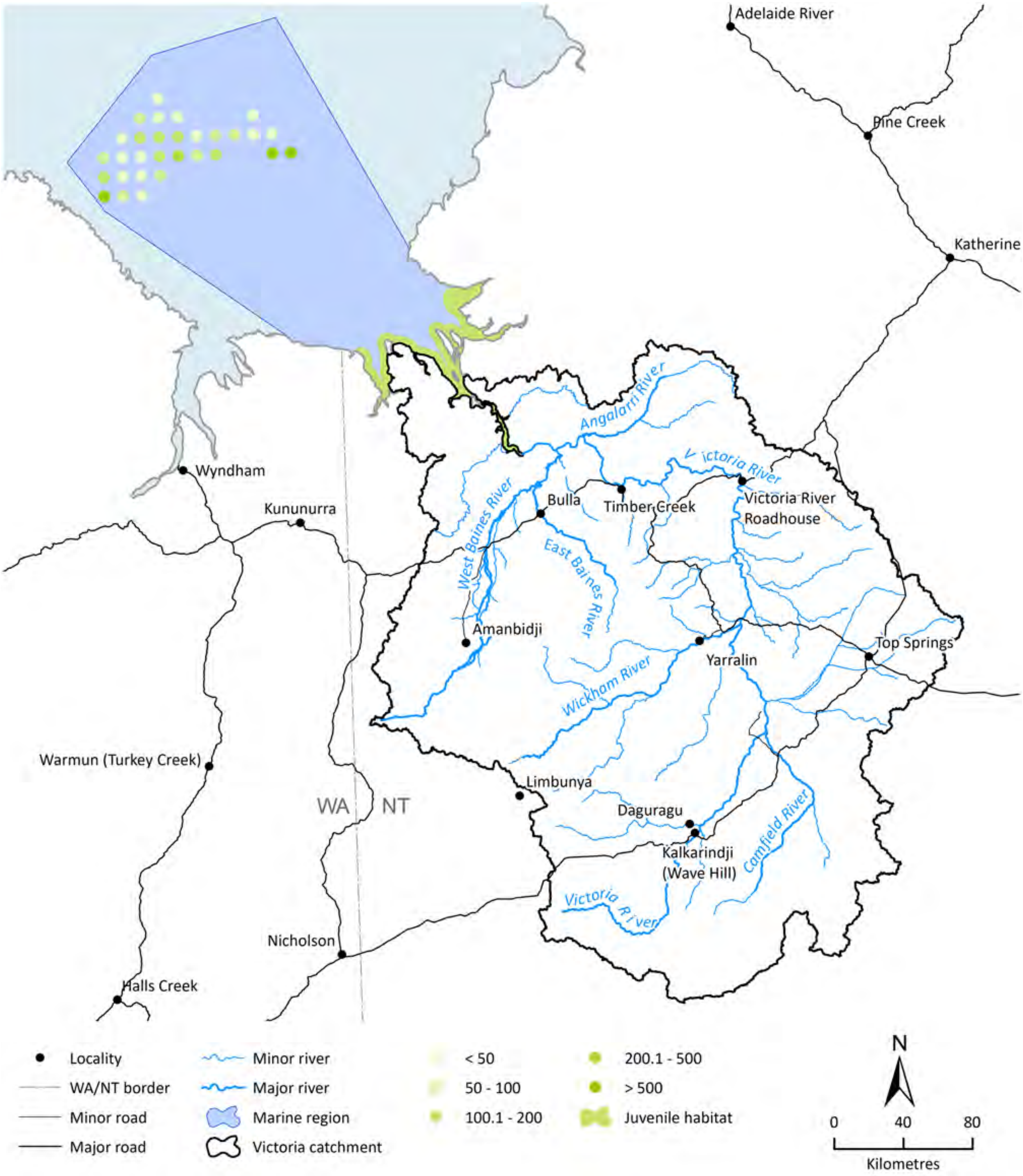

Figure 3-42 Fisheries catch distribution of white banana prawns in the Victoria catchment marine region

Banana prawn juveniles inhabit tropical river estuaries in the marine / brackish zone, particularly the mangrove forest / tributary creek matrix within river estuaries. and adult prawns are caught offshore in water about 10 to 40 m deep in the marine habitat, adjacent to their juvenile habitats.

Units are kilograms as total catches for the 10-year period 2011 to 2020.

Data sources: Kenyon et al. (2022); Kenyon et al. (2004)

Flow–ecology relationships for banana prawns

The life-history strategy of banana prawns renders them critically dependent on the natural flow regime in the Australian wet-dry tropics. Adult prawns spawn at sea, and pelagic eggs and larvae occupy the marine habitat, before post-larvae use currents to move shoreward to river estuaries (Vance and Rothlisberg, 2020). Prior to the annual wet season, post-larvae settle to benthic habitats in the estuarine mangrove forest and mudbank matrix, particularly upper tributary mangrove forests (at high tide) and creeks (Vance et al., 2002; Vance et al., 1990). They shelter and grow within the estuary, and a brackish ecotone supports lower mortality and faster growth (Staples and Heales, 1991; Vance et al., 1998; Wang and Haywood, 1999). Predation by fish within the estuary is high, and a significant proportion of the estuarine population is lost (Wang and Haywood, 1999).

Floodwaters cue juvenile banana prawns to emigrate, though the cue for redleg banana prawns is less clear. In the Victoria catchment marine region, the emigration of juvenile redleg banana prawns is enhanced in La Niña years with the Southern Oscillation Index value greater than 7 and high cumulative rainfall during January and February (Plagányi et al., 2021). In general, the larger the flood the greater the emigration event and the lower the estuarine salinity the smaller the prawns that emigrate (Kumlu and Jones, 1995; Staples and Vance, 1986) (Table 3-17). Emigrant juveniles and sub-adults move to the near-shore zone (Staples, 1980) and probably benefit from nutrient deposition within the flood plume (Burford et al., 2012; Burford and Faggotter, 2021). In addition, mortality is lower in marine habitats than in the estuary (Gwyther, 1982). The biological outcome is a larger adult population of banana prawns in coastal marine habitats that are cued by higher flood flows from adjacent estuaries (Duggan et al., 2019; Plagányi et al., 2021).

High-level pulsed flood flows during the monsoon season, low-level early-season flows, sustained flows during the wet season and persistent wet-season flows all have important effects on the estuarine population of both species of banana prawns. During the September to December recruitment window for juvenile prawns, estuaries within the Gulf of Carpentaria and Joseph Bonaparte Gulf ecosystems are stressed, and habitats are often hypersaline during the latter part of the dry season (Kenyon et al., 2004; Vance et al., 1990). The estuaries are a refuge habitat for many fish and crustaceans living under severe environmental conditions prior to the onset of the wet season, usually January to March (Babcock et al., 2019; Robins et al., 2020).

Early low-level flows that might occur during November and December condition tropical estuaries to brackish, cooler habitats that are more favourable to the growth and survival of crustaceans and fish within them, including juvenile banana prawns (Leahy and Robins, 2021; Ruscoe et al., 2004; Staples and Heales, 1991) (Table 3-17). Once an abundant estuarine population of juvenile banana prawn is established, high-level flood flows cue emigration and result in a large prawn population offshore. Persistent flows in the latter portion of the wet season continue to facilitate both a brackish estuary to support the growth of small juveniles and emigration of the larger juvenile population (Duggan et al., 2014; Staples and Vance, 1986). The ecological functions that support banana prawns, and their associated flow requirements, are summarised in Table 3-17.

Table 3-17 Ecological functions supporting banana prawns and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of brackish water conditions in estuaries during the dry season that support banana prawn growth and survival	Dry-season duration and intensity	Low flows
Persistence of brackish conditions in estuaries in the early dry season that support prawn growth and survival (from April onwards)	Early dry-season persistent low-level flow	Low flows
Provision of brackish estuarine conditions in the late dry season that support prawn growth and survival (prior to January)	Late dry-season first low-level flow	Low flows
Early wet-season flood flows that support small banana prawn growth and survival and cue emigration of large juvenile prawns	Early wet-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that cue banana prawn emigration	Moderate flood flows	Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that cue banana prawn emigration en masse	Large flood flows that cause freshwater estuarine habitats and scour estuaries	High flows, their frequency and seasonal reoccurrence

Pathways to change for banana prawns

The life history of banana prawns would be significantly affected by interruptions to the natural flows of northern Australian rivers. Large flows during the wet season cue emigration of banana prawns from estuarine habitats to the near-shore zone and further offshore. The emigration cues triggered by the annual monsoon-season flow regime renders banana prawns particularly vulnerable to water resource development. During high-flow years (strong wet season), banana prawns emigrate en masse from the estuary and commercial catches of prawns (as an indicator of abundance) are high (Broadley et al., 2020; Plagányi et al., 2022). During low-flow years (drier wet season), a proportion of banana prawns remains within the estuary and is subject to predation and mortality. Therefore, maintaining estuarine brackish habitats, diversity of river flow regimes and high-pulse flood flows enhances the populations of banana prawns and inshore and offshore habitat connectivity. Water resource development has the capacity to reduce the population of banana prawns (Broadley et al., 2020; Plagányi et al., 2021; Plagányi et al., 2022; Plagányi et al., 2023).

Extraction from, or impounding, low-level flows removes a large proportion of early-season low-level river flows, with subsequent impacts on estuarine banana prawns. Interrupting early-season low-level flows reduces the capacity of freshwater inputs to the estuary to create brackish habitats and may render an estuary continuously hypersaline. A hypersaline estuary is a stressful habitat for juvenile banana prawns during the annual recruitment window from September to January. Threshold levels of river flow as a trigger to water extraction can sustain the provision of flow, and hence the ecosystem services to the estuary, during this window of possible low-level flows before the onset of the bulk of wet-season precipitation during January to March (Plagányi et al., 2022). Significant extraction or impoundment of pulsed high-level flood flows from January to March reduces the emigration cue for juvenile banana prawns, reducing the proportion of the population reaching offshore habitats (Broadley et al., 2020; Plagányi et al., 2022).

The impact of water resource development, such as the construction of dams or water harvest at several levels of extraction, on coastal common banana prawn populations has been modelled (Plagányi et al., 2023). The biomass and commercial catches of the common banana prawn were predicted to decrease by 4 to 40% depending on the extent of water extraction or impoundment from the Mitchell, Gilbert and Flinders rivers in the eastern and southern Gulf of Carpentaria. The risk to the commercial fishery for banana prawns was assessed as negligible for one of four water resource development scenarios, moderate for two, and major for the remaining scenario (Plagányi et al., 2023). Similar work modelling redleg banana prawn response is yet to be undertaken, though the documented environmental drivers supporting the redleg banana prawn population suggest its response to reduced flows due to water resource development would be population decline, similar to common banana prawns. The common banana prawn model outputs included an explicit representation that the decline in the banana prawn population flowed on to detrimental effects on their predators. Plagányi et al. (2023) showed that both the construction of dams and the harvest of river flows via pumped water extraction affects aspects of common banana prawn life history that limit the resilience of the population. Anthropogenic reduction in the volume and duration of high-level flows, and variability in the seasonality and volume of low-level flows induced by water resource development, affect estuarine habitat suitability (brackish conditions preferred), growth, survival and emigration of banana prawns (Plagányi et al., 2023; Vance and Rothlisberg, 2020). The ecological outcomes of threatening processes on banana prawns in northern Australia, with their implications for changes to growth and mortality, community structure, habitat and population, are illustrated in Figure 3-43.

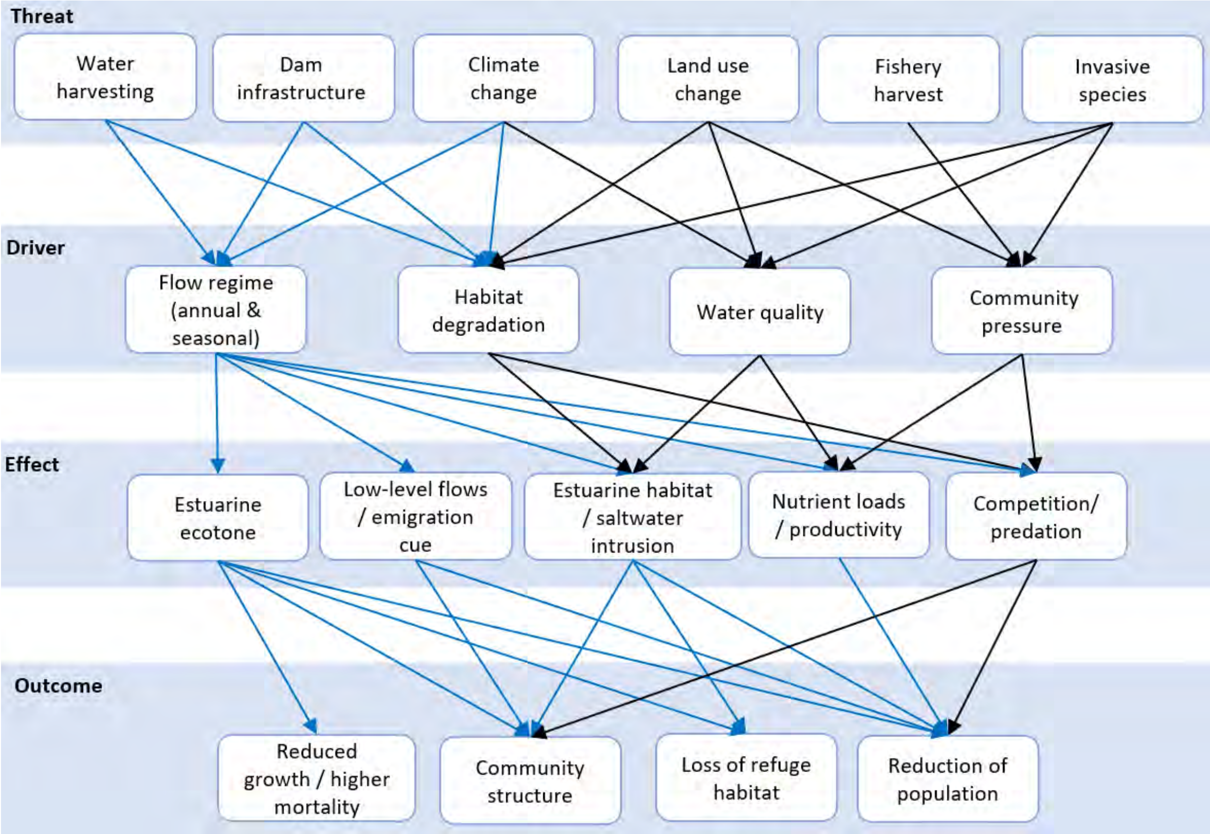

Figure 3-43 Conceptual model showing the relationship between threats, drivers, effects and outcomes for banana prawns in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.3.2 Freshwater turtles (family Chelidae)

Description and background to ecology

Freshwater turtles stand among the world's more endangered taxonomic groups, with 52% of the global species facing extinction or Threatened status (Böhm et al., 2013; Van Dijk, 2014). In Australia, freshwater turtles fall into three families: Chelidae (32 species), Trionychidae (two species), and Carettochelyidae (one species) (Georges and Thomson, 2010). Chelids, part of the Chelidae family, are highly aquatic, possessing webbed feet and the ability to remain submerged for long periods of time. These turtle species retract their necks sideways into their shells, and their dietary preferences vary between genera. Long-necked species, such as *Chelodina* spp. are predominantly carnivorous, feeding on fish, invertebrate and gastropods (Legler, 1982; Thomson, 2000). In contrast, short-necked species, such as *Elseya*, are herbivorous or specialise to eat fruits (Kennett, 1993). Freshwater turtles depend upon flooded wetland systems for breeding, nesting, food provision and refuge. Threatening processes such as changes in regional hydrology, habitat loss and climate change pose significant risks to their survival (Stanford et al., 2020).

In northern Australia, turtles inhabit various aquatic habitats, including river and floodplain wetland habitats like the main channel, waterholes, floodplain wetlands and oxbow lakes (Cann and Sadlier, 2017; Thomson, 2000). Many of the turtle species in this region have adaptations that help them to survive the interannual variation between the wet and dry seasons, such as synchronising hatching with the onset of the wet season (Cann and Sadlier, 2017). During the dry season, the movements of the freshwater turtles on and off the floodplain are limited, making them more vulnerable to changes in water quality, invasive species and habitat degradation (Cann and Sadlier, 2017; Doupe et al., 2009).

Australian freshwater turtles hold both ecological and cultural significance, including the consumption of some species by Indigenous Peoples as a seasonal source of protein (Jackson et al., 2012). Indigenous Peoples' connections to freshwater turtles through songlines and ceremonies are widespread, and certain people have roles as custodians and caretakers according to the kinship system. Knowledge holders share seasonal knowledge insights related to freshwater turtle hunting, behaviour, diet and physiology, including aestivation (i.e. dry-season torpor), fatness and breeding cycles. For example, knowledge holders said the dry (cold) season is the time to hunt for northern snake-necked turtle (*Chelodina oblonga*; formerly *Chelodina rugosa*). Indigenous Peoples have identified natural predators (including birds of prey such as eagles and hawks, crocodiles, goannas and dingoes), feral animals (such as pig, buffalo, horse, donkey, cattle and cane toad) and climate change (e.g. lower rainfall) as major threats to freshwater turtles (Russell et al., 2021).

Freshwater turtles in the Victoria catchment

Ten species of freshwater turtles are described for the NT (Department of Environment Parks and Water Security, 2019a). From those, only three freshwater turtle species have been collected in the Victoria catchment: the sandstone snake-necked turtle (*Chelodina burrungandjii*), the northern snake-neck turtle and northern snapping turtle (*Elseya dentata*) (Figure 3-44). Records for the Victoria catchment remain sparse compared to other Australian regions. Currently, *E*. *dentata* and *C. oblonga* are listed as Least concern by the Northern Territory Government, while *C. burrungandjii* is listed as data deficient (Department of Environment Parks and Water Security, 2019b).

The sandstone snake-necked turtle inhabits permanent bodies of water, like lakes and rivers, billabongs and swamps with substantial water plant beds. The northern snapping turtle lives in permanent riverine habitats, often subjected to rapid rises in water level, and migrates from deep waterholes to shallow, flooded wetlands to feed. Its distribution in the lowlands can overlap with the sandstone snake-necked turtle. Similarly, the northern snake-neck turtle is found in lakes, billabongs and swamps with substantial water plant beds, occupying permanent rivers and migrating from deep waterholes to shallow flooded wetlands for feeding, often overlapping in distribution with the sandstone snake-necked turtle. The modelled distribution of *Chelodina oblonga* is shown in Figure 3-45 and *Elseya dentata* in Figure 3-46.

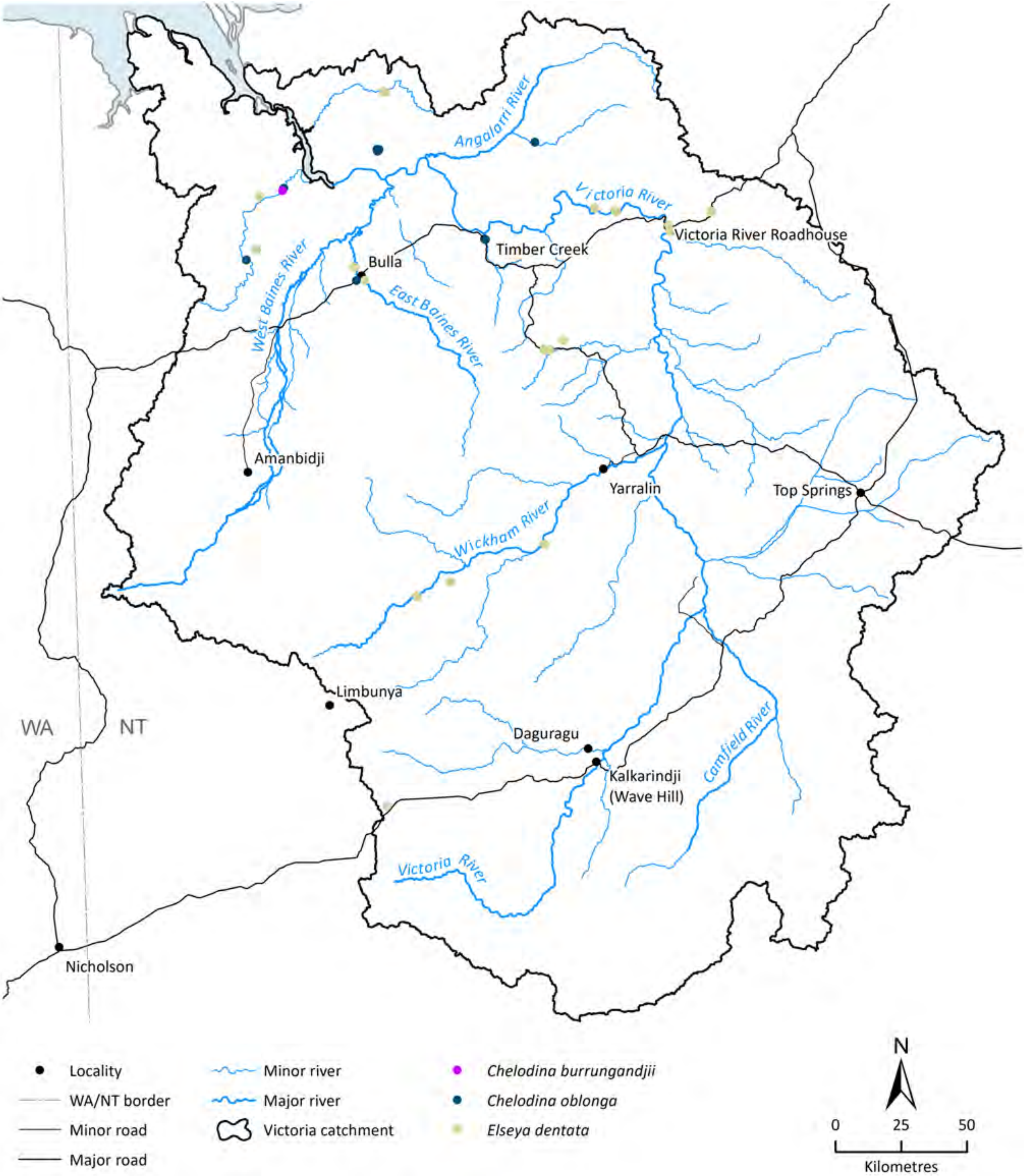

Figure 3-44 Observed locations of freshwater turtles within the Victoria catchment

Data sources: Atlas of Living Australia (2023); Department of Environment Parks and Water Security (2019a)

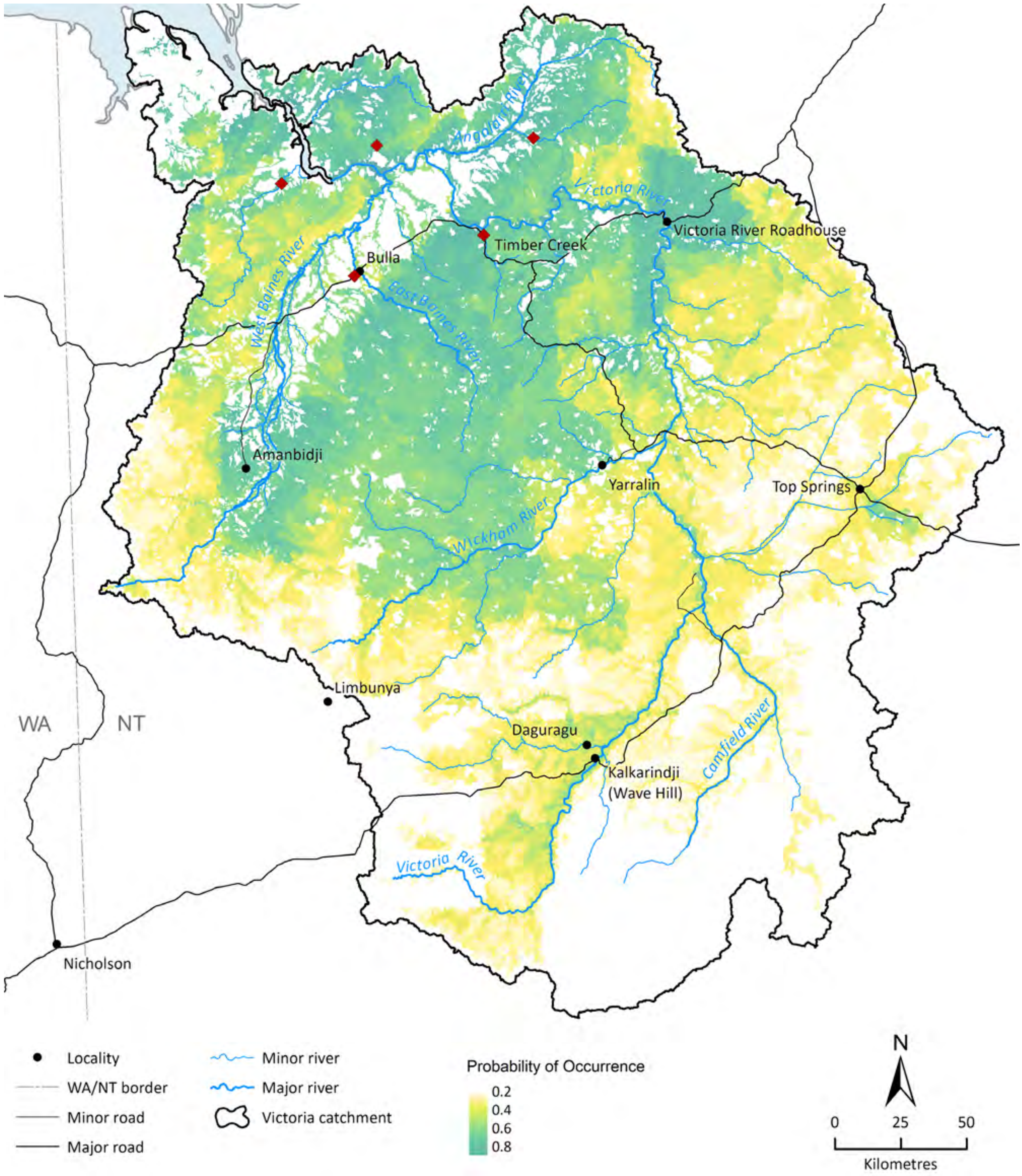

Figure 3-45 Modelled probability of occurrence of northern snake-necked turtle (*Chelodina oblonga*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

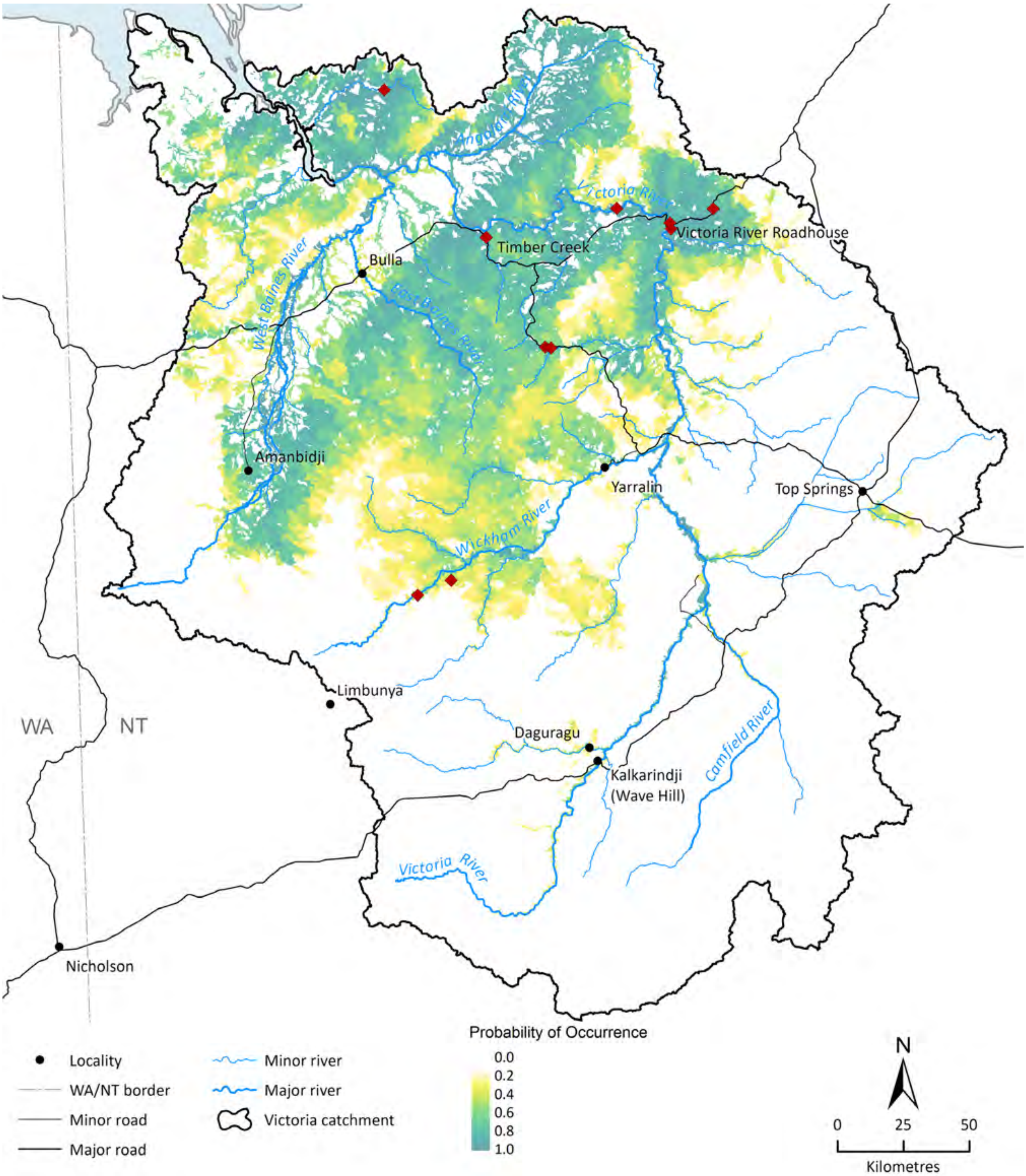

Figure 3-46 Modelled probability of occurrence of northern snapping turtle (*Elseya dentata*) in the Victoria catchment

Probability of occurrence is based upon a general linear model with model predictors provided in Appendix A. For the SDMs, only records later than 1960 that intersected with polygons that contain waterways and that had a stated coordinate uncertainty <5 km were used. Red points show locations from Atlas of Living Australia.

Data inputs: Atlas of Living Australia (2023)

Flow–ecology relationships for freshwater turtles

Freshwater turtles rely on water and wetland systems for breeding and nesting, food provision and refuge (Cann and Sadlier, 2017) (Table 3-18). In Australia, some species exhibit an indirect flow dependency due to the habitat being influenced by the flow regime. For other species, flow dependency is crucial for supporting critical phases of their life history, for example, the emergence of northern snapping turtle hatchings that coincides with the onset of the wet season (Cann and Sadlier, 2017). Nesting activities occur during the dry season in nests located within 4 m of the water, typically in alluvial soils, sand or soil mix on steep to gently sloping banks. Nesting is observed at dispersed locations along the watercourse (Cann and Sadlier, 2017).

Many turtle species in northern Australia have developed adaptive traits to thrive in the highly variable wet-dry environment (Cann and Sadlier, 2017). The nesting behaviour of the northern snake-neck turtle commences in February (wet season) and concludes by July (mid-dry season). Eggs are laid in the mud, under shallow water, surrounded by flooded waterholes. Embryo development halts while the eggs are in the water and continues once water recedes. Hatchling emergence coincides with the onset of the wet season (Cann and Sadlier, 2017).

During dry periods, dispersal of freshwater turtles is limited, which makes them more vulnerable to changes in water quality, invasive species and habitat degradation (Cann and Sadlier, 2017). Turtles often move to the shallows to aestivate during the dry season. The weeks immediately before drying pose the highest risk of predation on turtles. Introduced feral pigs pose a significant threat to turtles and their eggs (Fordham, 2006; Pusey and Kennard, 2009). Feral pigs also negatively affect turtle habitat, causing disturbances in aquatic ecosystems by disturbing sediments, destroying aquatic vegetation, creating anaerobic and acidic conditions, and enriching wetlands with nutrients. Additionally, turbid conditions limit visibility, compromising the turtles' hunting opportunities. Destruction of vegetation significantly alters production and respiration regimes, causing anoxic conditions and pH imbalances (Doupe et al., 2009).

Freshwater turtles use large riparian zones for various aspects of their life cycle, including nesting. Altering or eliminating these riparian habitats could reduce nest survival and, consequently, juvenile recruitment into the breeding population. It would also affect adult survival through lack of feeding areas and refuge habitat for the dry season, thereby increasing the risk of extinction for freshwater turtle populations (Bodie, 2001). However, more comprehensive data for freshwater turtles are needed, particularly regarding the timing and extent of riparian use. The ecological functions and the supporting flow requirements for freshwater turtles are summarised in Table 3-18.

Table 3-18 Ecological functions supporting freshwater turtles and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT	FLOW COMPONENT OR ATTRIBUTE
Recruitment, survival and population size	Flow regime change (reduced flow volume / persistence) During the transition from the wet to dry seasons, intermittent rivers contract to waterholes	Flow timing and magnitude. Seasonality of flows
Recruitment, survival and population size	Early wet-season first flush	Flow timing and magnitude. Seasonality of flows
Dispersal	Dry-season duration and intensity	Flow timing and magnitude. Seasonality of flows
Habitat availability	Large flood flows to scour and maintain bed structure to occur at suitable frequency	Low-flow days – number of days at or below threshold, duration of days at or below threshold No flow days – number of no flow days, duration of no flow days

Pathways to change for freshwater turtles

Aquatic and riparian habitats play a crucial role as feeding and breeding areas for freshwater turtles (Cann and Sadlier, 2017; Cosentino et al., 2010; Gibbons et al., 2000; Marchand and Litvaitis, 2004). Fragmentation and habitat loss can make freshwater turtles more vulnerable by disrupting nesting sites and refugia, and restricting emigration and dispersal among wetlands (Bodie and Semlitsch, 2000; Bowne et al., 2006). Similarly, changes to hydrological patterns (timing, velocity, persistence and flow extent) by creating barriers and extracting water could lead to changes to the distribution of freshwater species, population growth and reproduction (Hunt et al., 2013). Understanding how these drivers affect turtles is critical for improving environmental management and conservation at landscape scales (Bodie and Semlitsch, 2000). Given that turtle species from northern Australia are less studied than those in eastern Australia and elsewhere, mainly due to the remoteness of their habitats (Cann and Sadlier, 2017), much of their flow requirements and responses to flow are inferred from research on eastern turtle species. The ecological outcomes of threatening processes on freshwater turtles in northern Australia, and their implications for changes to community structure, population viability and biodiversity and ecosystem function are discussed below and shown in Figure 3-47.

Movement is critical for freshwater turtles to access across breeding, feeding, aestivation and refuge habitats (Ocock et al., 2018). Access to water and connectivity between suitable habitats are vital for allowing the movement of the turtles within the river channels. Threats that reduce river–wetland connectivity, such as water harvesting, dam infrastructure or climate change, are some of the key threatening processes for the freshwater turtles in northern Australia (Figure 3-47) (Stanford et al., 2020). During wet to dry season transitions, freshwater turtles move on and off the floodplains. In perennial rivers, reduced dry-season baseflows (due to extraction) could decrease the availability of suitable habitat supported by flows. Such a baseflow reduction could even shift the rivers from perennial to intermittent status, potentially limiting turtles' access to freshwater shelters during the dry season (Hunt et al., 2013). Disconnections caused by a reduced baseflow hamper the freshwater turtles' movements on and off the floodplain during the transition from wet to dry season (Warfe et al., 2011). Activities like impoundment, regulation and channelisation of riverbanks and beaches can reduce nesting and feeding habitat for most turtle species. Long-lived freshwater turtles respond slowly to environmental changes, and their impacts may not be evident until many years after the alterations occur (Tucker et al., 2012; Waltham et al., 2013). Interrupting migratory routes, especially nesting sites, disrupts gene flow between populations, reducing the genetic diversity of these turtle populations (Alho, 2011; Lees et al., 2016).

Changes to the inundation and flow regimes reduce freshwater turtles' feeding grounds and suitable habitats such as waterholes (Warfe et al., 2011), intensifying resource competition (Chessman, 1988; DSITIA, 2014). High turtle abundance can reduce hatchling survival due to direct predation and resource competition between adults and juveniles (Trembath, 2005).

Channelising rivers and shoreline hardening may eliminate nesting and basking areas and alter the hydrodynamic processes that maintain critical nesting habitat (Roosenburg, 2014). Removing exposed logs and snags to promote recreational boating eliminates critical basking sites and prey habitat (Lindeman, 1999). Changes in flow regimes due to water use and regulation can also affect freshwater turtles by disrupting breeding cues and reducing feeding and/or nesting grounds. For instance, higher dry-season flows can reduce or eliminate emergent sandbars (Tracy-Smith, 2006), affecting the availability of preferred nesting sites and potentially resulting in less successful recruitment (Bodie, 2001). Fluctuating water levels through water management infrastructure can inundate freshwater turtles' nests, resulting in egg mortality and the loss of optimal nesting habitat (Waltham et al., 2013). Consequently, reduced breeding success, survival and population size of freshwater turtles have an overarching impact on the community and population structure (Georges et al., 1993; Tucker et al., 2012). Similarly, rapid shifts in temperature might preclude successful gradual responses that functioned historically, like active modification of geographic range. Also, early nesting and early egg maturity resulting from temperature rises may lead to the eggs perishing in the ground, while late nesting risks the eggs being prematurely flooded by rising waters (Jolly, 2008).

During extended drought periods, refuge habitats become critical, as aestivation − limited by fat reserves and dehydration − rarely lasting more than 7 months (Roe et al., 2008). Water extraction may diminish waterhole sizes, numbers and persistence, potentially delaying their reconnection between seasons (Warfe et al., 2011). Structure like reservoirs, weirs and barrages can reduce inundation in floodplains. They can also act as barriers to downstream sediment transmission, resulting in smaller and fewer sandbars (Pusey and Kennard, 2009). Groundwater discharge through springs can be important for maintaining perennial river baseflow in the dry season. Persistent surface water remains essential as refuge habitat for freshwater turtles (Warfe et al., 2011).

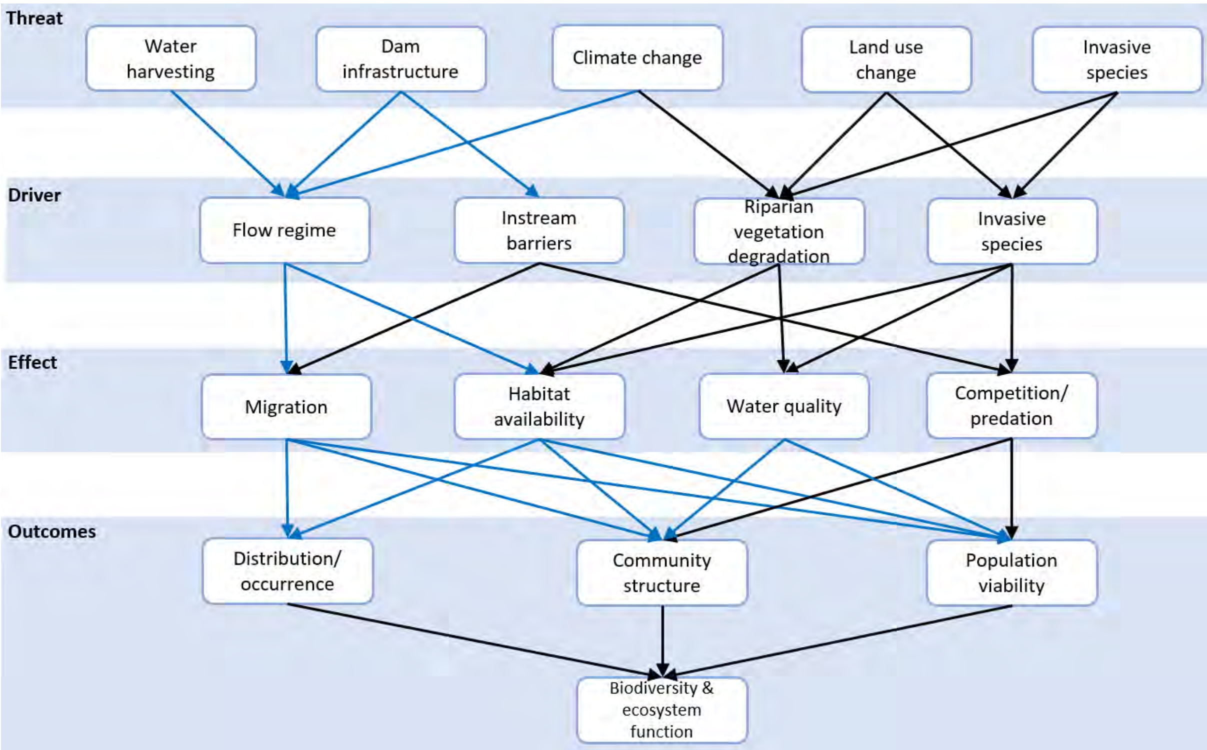

Figure 3-47 Conceptual model showing the relationship between threats, drivers, effects and outcomes for freshwater turtles in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.3.3 Mud crabs (*Scylla serrata*)

Description and background to ecology

Mud crabs are large-bodied, large-clawed, short-lived, fast-growing decapod crustaceans (>200 mm carapace width) that inhabit the estuarine and shallow subtidal community along tropical and subtropical coastlines, especially mangrove-dominated habitats (Figure 3-48). They are targeted throughout their range as a commercial, recreational and Indigenous fishery resource and a prized table species (commercial catch 40,000 t worldwide in 2012) (Alberts-Hubatsch et al., 2016). Two species of mud crab are found in tropical Australia (*Scylla serrata* and *S. olivacea* (Alberts-Hubatsch et al., 2016; Robins et al., 2020)) but only *S. serrata* is found in the marine region of the Victoria catchment. Mud crabs are distributed across the Indo-Pacific; though in Australia, *S. serrata* is the dominant commercial species by abundance (Robins et al., 2020). *Scylla olivacea* is found only in the north-east Gulf of Carpentaria in the Weipa region (Alberts-Hubatsch et al., 2016; Robins et al., 2020).

Figure 3-48 Mangrove and intertidal habitat typical of mud crab habitat in northern Australia

Photo attribution: CSIRO

The Northern Territory mud crab catch in 2018–19 was 270 t and valued at $7,881,000 (all crab species; Steven et al. (2021)). At the Sydney Fish Market, the price for mud crabs averaged about $34/kg in 2018–19, making them a high-value regional resource (Robins et al., 2020). The mud crab's high fecundity, high natural mortality and relatively short life span suggest that they are a moderately resilient species suitable for sustainable harvest. The high market price commanded by mud crabs supports their fishery within, and transport from, remote coastal locations in tropical Australia, including the Gulf of Carpentaria regions.

Mud crabs occupy mangrove forest (see Section 3.4.4) and nearby shallow subtidal habitats within estuarine and coastal ecosystems (Alberts-Hubatsch et al., 2016); hence, they use the estuary and shallow-water coasts in the Victoria catchment marine region as habitat. Mud crabs are an important ecological species, being both predator and prey in the coastal ecosystem. As small juveniles, mud crabs are detritivores; as large juveniles and as adults they are benthic predators feeding on crustaceans, molluscs and fish. Estimates suggest that the mud crab population consumes 650 kg biomass per hectare per year in the mangrove forest and 2100 kg biomass per hectare per year in mangrove fringe habitat (Alberts-Hubatsch et al., 2016). Mud crabs dig burrows to rest during the day, reworking mud substrates within mangrove forests and mudbanks. They play a significant trophic role in mangrove ecosystems.

Mud crabs demonstrate a larval life-history strategy (see Robins et al. (2020) for recent comprehensive review): adult crabs mate in the estuary and the females migrate offshore to spawn (September to November; larvae require marine salinity) (Hill, 1994; Hill, 1975; Meynecke et al., 2010; Welch et al., 2014). Their larvae transform to megalopae (the final larval stage) that move by drift inshore where they settle as benthic juveniles in estuarine mangrove and mudflat habitats (Alberts-Hubatsch et al., 2016; Meynecke et al., 2010; Robins et al., 2020). The larval form facilitates ontogenetic migration as crabs grow to the juvenile stage and settle to their inshore habitats and also long-distance dispersal and genetic mixing (Gopurenko and Hughes, 2002; Gopurenko et al., 2003; Robins et al., 2020). Initial recruitment to inshore habitats occurs at the mangrove forest fringe, and as crabs grow their dependence on estuarine mangroves declines (Alberts-Hubatsch et al., 2014). Mud crabs remain in the estuary for several years as sub-adults and adults before the females alone emigrate to spawn (Hill, 1994). Regionally, the annual wet season and subsequent runoff is a significant determinant of their recruitment strength and total catch (possibly lagged by 1 to 2 years) in the estuary and near-shore zone (Meynecke and Lee, 2011; Meynecke et al., 2010). Recent analyses of Gulf of Carpentaria catches support the finding that river flow enhances mud crab catch; however, the research also shows that the Southern Oscillation Index and high air temperature during the wet season can be dominant negative influences on mud crab abundance within estuarine habitats (Blamey et al., 2023).

Mud crabs in the Victoria catchment and marine region

Scylla serrata mud crabs can be found in the Victoria catchment, including the estuary of the Victoria River (Figure 3-49), and they are caught commercially in these waters. Northern Territory Fisheries reporting grid #1429 centres on the eastern Joseph Bonaparte Gulf and includes the littoral habitats in the Victoria and Keep rivers (Anon, 2017). This grid is part of the Arafura-West managed region of the Northern Territory Mud Crab Fishery, which includes both the Arafura coast of the Top End and the Victoria catchment study area. Commercial catches from the Arafura-West mud crab stock have averaged 124 t over the period 2007 to 2016, ranging between 67 and 149 t (Anon, 2017). The proportion of the catch of Arafura-West stock that originates from the Victoria catchment region rather than the Arafura region is not known, but is probably small. Northern Territory Fisheries suggests that 5% of the mud crab catch is harvested by the Indigenous fishing sector (Anon, 2017).

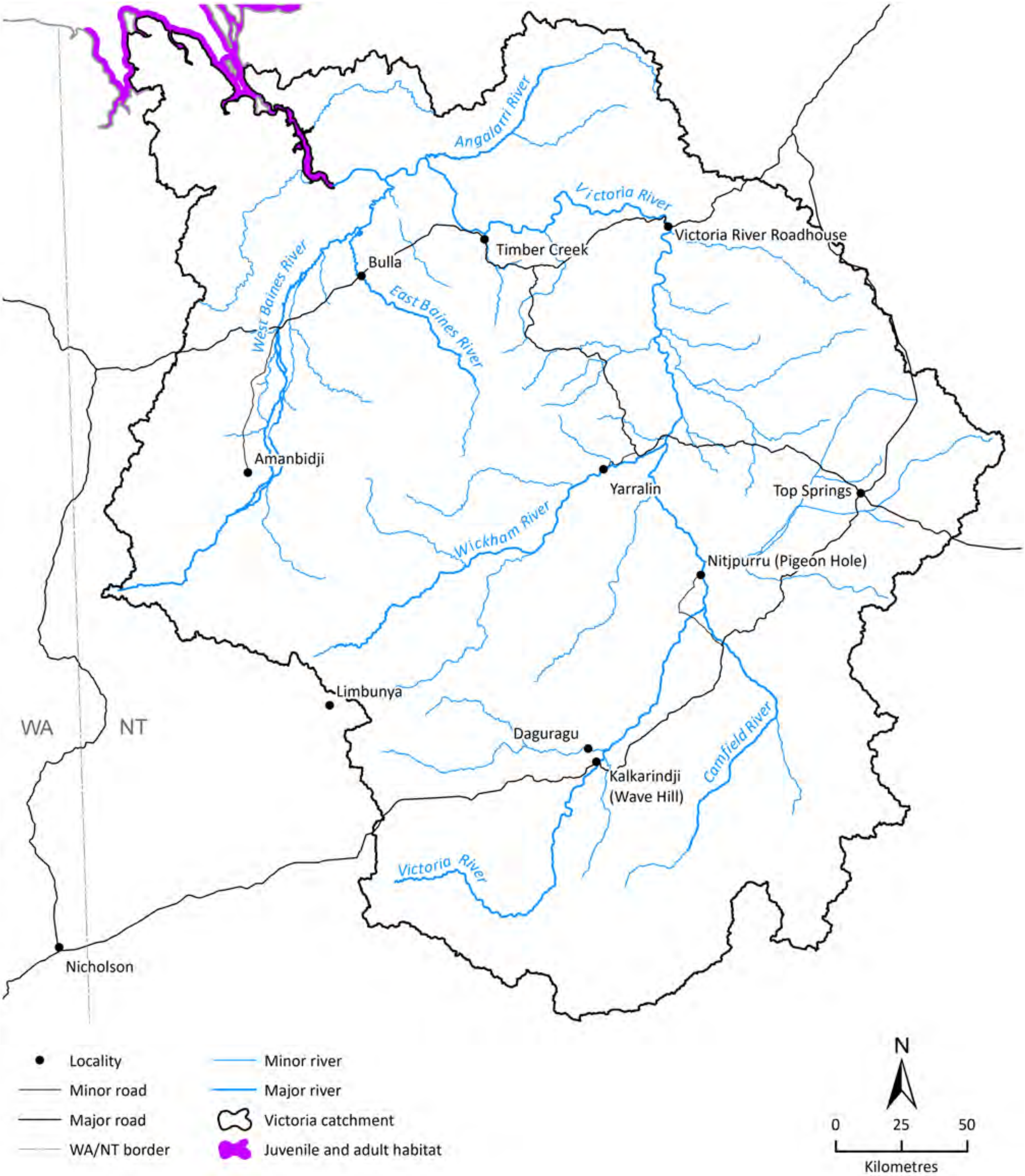

Figure 3-49 Mud crab habitat in the Victoria catchment marine region

Mud crab juveniles use the mangrove forest and mudbank habitats within the estuary, while adult crabs are caught within the estuary and in shallow subtidal habitats in the littoral zone. Female mud crabs migrate offshore to spawn.

Flow–ecology relationships for mud crabs

The mud crab life-history strategy renders the species critically dependent on the natural flow regime in the wet-dry tropics. Juvenile and adults mud crabs are estuarine and littoral-coast residents, and both of these habitats are influenced by freshwater inflows. The effect of freshwater flows on mud crabs is difficult to define compared to species that emigrate as they grow into a new life-history stage. Mud crabs do not use freshwater riverine or palustrine habitats as juveniles, nor do they emigrate from their estuarine habitats to marine adult habitats. Adult female mud crabs emigrate from inshore to marine habitats to spawn, but they reside in estuarine and coastal habitats as adults prior to their reproductive response (Alberts-Hubatsch et al., 2016). Optimal estuarine conditions for mud crab growth and survival are found in a brackish ecotone between marine habitats and the freshwater riverine habitats (Ruscoe et al., 2004). Mud crabs are not subject to emigration cues, though freshwater inflows may cause movement down the estuary as upper reaches become too fresh to tolerate (Robins et al., 2020).

Although positive relationships between flow and mud crab catches across the Gulf of Carpentaria and other northern estuaries have been identified previously (Robins et al., 2005), many other environmental parameters are also correlated with catch (Plagányi et al., 2022; Robins et al., 2020).

Female mud crabs spawn offshore from September to November. Their larvae require marine salinities (25–30 ppt) and warm waters (26–30 °C) for optimal growth (Alberts-Hubatsch et al., 2016; Welch et al., 2014). Megalopae are tolerant of 15 to 45 ppt salinity, facilitating their occupation of diverse inshore habitats where physical parameters can be variable. Though larvae survive best in marine waters, the growth and mortality of juvenile mud crabs is optimal in brackish waters characteristic of the tropics: about 25 to 30 °C with a salinity of 10 to 20 ppt (for growth) and 10 to 30 ppt (for survival) (Meynecke and Lee, 2011; Meynecke et al., 2010; Ruscoe et al., 2004). Mud crabs can tolerate cool conditions (<20 °C) for short periods, but require temperatures higher than 20 °C to grow and function (~25–30 °C is optimal). Juvenile mud crabs resident in estuaries can tolerate a broader salinity range (5–45 ppt); they benefit from perennial baseflows and low flood flows that create brackish conditions in the estuary (Alberts-Hubatsch et al., 2016; Welch et al., 2014). Estuaries in the Australian tropics often become hypersaline in the lead up to the wet season and in years of very low rainfall. Under hypersaline conditions, growth and survival of the crabs may be inhibited until first rains and low-level river flows reduce estuarine salinity to brackish levels (Table 3-19). Adult mud crabs are euryhaline animals, capable of living in freshwater-flooded to hypersaline waters (<5−45 ppt) (Alberts-Hubatsch et al., 2016).

High-level flows benefit the estuarine mud crab population via increased productivity due to nutrient loads delivered to estuarine and near-shore littoral habitats (Burford et al., 2016; Burford et al., 2012; Burford and Faggotter, 2021). Also, mangroves rely on the depositional environment sustained by sediment loads on large floods to maintain their intertidal habitat (Asbridge et al., 2016). However, very large floods cause the loss of marine influence and may negatively affect inshore crab habitats in the year of the flood; though they may be beneficial in subsequent years due to medium-term productivity enhancement (Robins et al., 2020) (Table 3-19). Large floods that create a freshwater estuary cause mortality and movement from estuaries: juvenile crabs in fresh water suffer 100% mortality (Ruscoe et al., 2004) and during a one-in-fifty-year flood in the south-east Gulf of Carpentaria in 2009, adult crabs in freshwater estuaries emigrated elsewhere (Gary Ward (Gulf of Carpentaria fisher), 2010, pers. comm.). In contrast, during lower-level floods survival of juvenile crabs in salinities of 5 to 40 ppt was high (optimally 15–25 ppt), and adult crabs were abundant in brackish estuaries (Robins et al., 2020). The ecological functions that support mud crabs, and their associated flow requirements, are summarised in Table 3-19.

Table 3-19 Ecological functions supporting mud crabs and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintenance of brackish water conditions in estuaries that supports growth and survival of mud crab juveniles during the dry season	Dry-season duration and intensity	Low flows
Persistence of brackish conditions in estuaries in the early dry season that supports growth and survival of mud crab juveniles (from April onwards)	Early dry-season first low-level flow	Low flows
Provision of brackish estuarine conditions in the late dry season that supports growth and survival of mud crab juveniles (before January)	Late dry-season first low-level flow	Low flows
Early wet-season flood flows that supports growth and survival of mud crab juveniles	Early wet-season first flush	Flow timing and magnitude. Seasonality of flows
Wet-season moderate-level flows that maintains mud crab habitat	Moderate flood flows	Flow timing and magnitude. Seasonality of flows
Wet-season high-level flows that produce a freshwater estuary that damages mud crab habitat	Large flood flows that cause freshwater estuarine habitats and scour estuaries	High flows, their frequency and seasonal reoccurrence

Pathways to change for mud crabs

Mud crabs exhibit a life history that would be significantly affected by interruptions to the natural flows of northern Australian rivers. In the western Gulf of Carpentaria, the short wet season (3 months at most) and unreliability of annual rainfall (including consecutive years of low rainfall) render mud crabs highly vulnerable to climate events, especially cumulative heat from November to March (Robins et al., 2020).

While river flow and rainfall have been shown to be positively related to mud crab catch in the eastern, southern and western Gulf of Carpentaria, environmental stressors in the ecosystem, such as evaporation and heat stress, can be extreme and have major negative impacts on mud crab populations (Robins et al., 2020). Particularly in the western and south-western Gulf of Carpentaria, evaporation during the dry season and heat stress during the wet season decreased mud crab catch.

Analysis of environmental factors and commercial catch by Robins et al. (2020) showed that river flow and water stress (rainfall offset by evaporation; less stress if rainfall is high) had a positive effect on mud crab catch in Gulf of Carpentaria catchments, while other stressors such as evaporation during the dry season and heat stress during the wet season had negative effects on catch. Mean sea-level anomaly during the wet season and the Southern Oscillation Index were positive for catches in this region (Blamey et al., 2023; Robins et al., 2020). Moreover, Blamey et al. (2023) showed that both constructing dams and harvesting river flows via pumped water extraction affect aspects of the mud crab life history that limit the resilience of their population. Reduced volume and duration of high-level flows due to water resource development ,and variability in the seasonality and volume of low-level flows, affect estuarine habitat suitability (brackish conditions preferred), growth and survival of mud crabs (Blamey et al., 2023; Robins et al., 2020). The impact of water resource development such as the construction of dams or water harvest at several levels of extraction on coastal mud crab populations has been modelled (Blamey et al., 2023). With the exception of the perennial Mitchell River, an array of water harvest and impoundment scenarios predicted a reduction in both the biomass and commercial catch of mud crab by 36 to 46% on average in the ephemeral and temporally variable Gilbert and Flinders rivers in the eastern and southern Gulf of Carpentaria. The risk to mud crab population was assessed as negligible (one scenario), major (one scenario) and severe (two scenarios) for the four water resource development scenarios. The risk to the commercial fishery for mud crabs was assessed as major for two of the scenarios, severe for one, and negligible for the remaining scenario (Plagányi et al., 2023).

Hence, reduction in river flows due to water resource development would be expected to have detrimental effects on mud crab catches in the Victoria catchment and marine region. In particular, reduced low-level flows − those flows that condition estuaries to brackish habitats after the extended dry season − would reduce the growth and survival of mud crabs in a hypersaline estuary. The ecological outcomes of threatening processes on mud crabs in northern Australia, and their implications for changes to growth and mortality, community composition, habitat and population are illustrated in Figure 3-50.

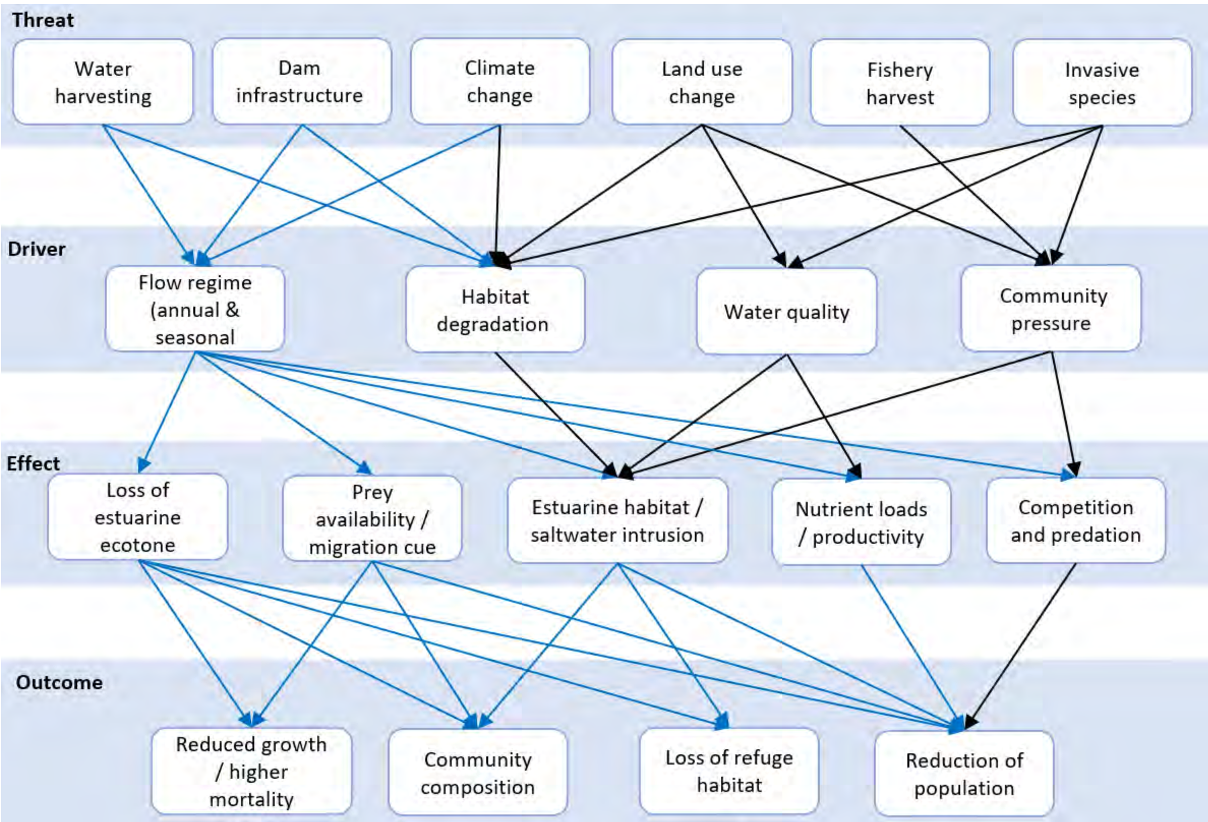

Figure 3-50 Conceptual model showing the relationship between threats, drivers, effects and outcomes for mud crabs in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4 Freshwater-dependent habitats

3.4.1 Floodplain wetlands

Description and background to ecology

Wetlands in the wet-dry tropics of Australia have great conservation value (Finlayson et al., 1999), and are one of the most diverse aquatic ecosystems in Australia (Douglas et al., 2005). Wetlands provide permanent, temporary or refugia habitat for both local and migratory waterbirds (van Dam et al., 2008), spawning grounds and nurseries for floodplain-dependent fish (Ward and Stanford, 1995), and habitat for many other aquatic and riparian species (van Dam et al., 2008). Floodplain wetlands are an important source of nutrients and organic carbon, driving primary and secondary productivity (Junk et al., 1989; Nielsen et al., 2015). Wetlands also provide a range of additional ecosystem services, including water quality improvement, carbon sequestration and flood mitigation (Mitsch et al., 2015).

Hydrological regimes are fundamental to sustaining the ecological characteristics of rivers and their associated floodplains (Pettit et al., 2017). In the wet-dry tropics of northern Australia, the ecology of wetlands is highly dependent on the seasonal rainfall−runoff pattern and the associated high and low flows (Pidgeon and Humphrey, 1999; Warfe et al., 2011). These flows are important drivers of floodplain wetland ecosystem structure and processes (Close et al., 2012; Warfe et al., 2011). Changes to these flow characteristics are likely to have a significant impact on the aquatic biota (Close et al., 2012). The timing, duration, extent and magnitude of wetland inundation has the greatest impact on the ecological values, including species diversity, productivity and habitat structure (Close et al., 2015).

Under the Ramsar convention a wetland is defined as (Ramsar Convention Secretariat, 2004):

> *'areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres.'*

The Northern Territory Government defines wetlands as including coastal saltmarshes, mangrove swamps, freshwater lakes and swamps, floodplains, freshwater ponds, springs and saline lakes, that can be permanent, seasonal or intermittent, and can be natural or artificial (Northern Territory Government, 2020). This Assessment does not consider areas within the river channel to be wetlands (they are considered to be inchannel waterholes (see Section 3.4.2)). Similarly, marine or saline habitats including mangroves and coastal saltmarshes (salt flats) are also treated as separate assets in this project (sections 3.4.4 and 3.4.5, respectively).

Floodplain wetlands in the Victoria catchment

The Victoria catchment has two nationally significant wetlands listed under the Directory of Important Wetlands in Australia (DIWA): Bradshaw Field Training Area and Legune Wetlands (Figure 3-51) (Department of Agriculture, Water and the Environment, 2021). There are no Ramsar-listed wetlands within the catchment.

The Bradshaw Field Training Area is in the north of the Victoria catchment near Timber Creek and is approximately 871,000 ha in size (Figure 3-51) (Department of Agriculture, Water and the Environment, 2021). While it is currently used as a military training area, Bradshaw Field Training

Area is typically inundated each wet season by flooding in the Fitzmaurice and Victoria rivers (Department of Agriculture, Water and the Environment, 2021).

The Legune Wetlands, comprising the Legune Homestead Swamps and the Osman Lake system, is on the western boundary of the Victoria catchment near the coast (Figure 3-51). The Legune Homestead Swamps are approximately 5000 ha of freshwater sedge swamps, wooded swamps and grassy marshes. They provide important habitat for waterbirds. The Osman Lake system is approximately 4000 ha and consists of a lake and a series of 15 claypans. It mostly fills directly from rainfall but does receive inflows from the surrounding area (Department of Agriculture, Water and the Environment, 2021). As it is not connected to the Victoria River network, it is not considered a floodplain wetland asset for the purposes of this analysis.

As well as these two nationally significant wetlands, there are a few floodplain areas within the Victoria catchment that are inundated during the wet season (shown as land subject to inundation in Figure 3-51). This includes floodplains that occur in association with Victoria, Wickham, Angalarri and Bullo rivers, and Alpha, Gipsy, Gorgon and Lalngang creeks (Figure 3-51). A large area in the northwest corner of the catchment is also subject to inundation, but as it is not connected to the river network, it is not considered floodplain wetlands and thus not considered in this analysis.

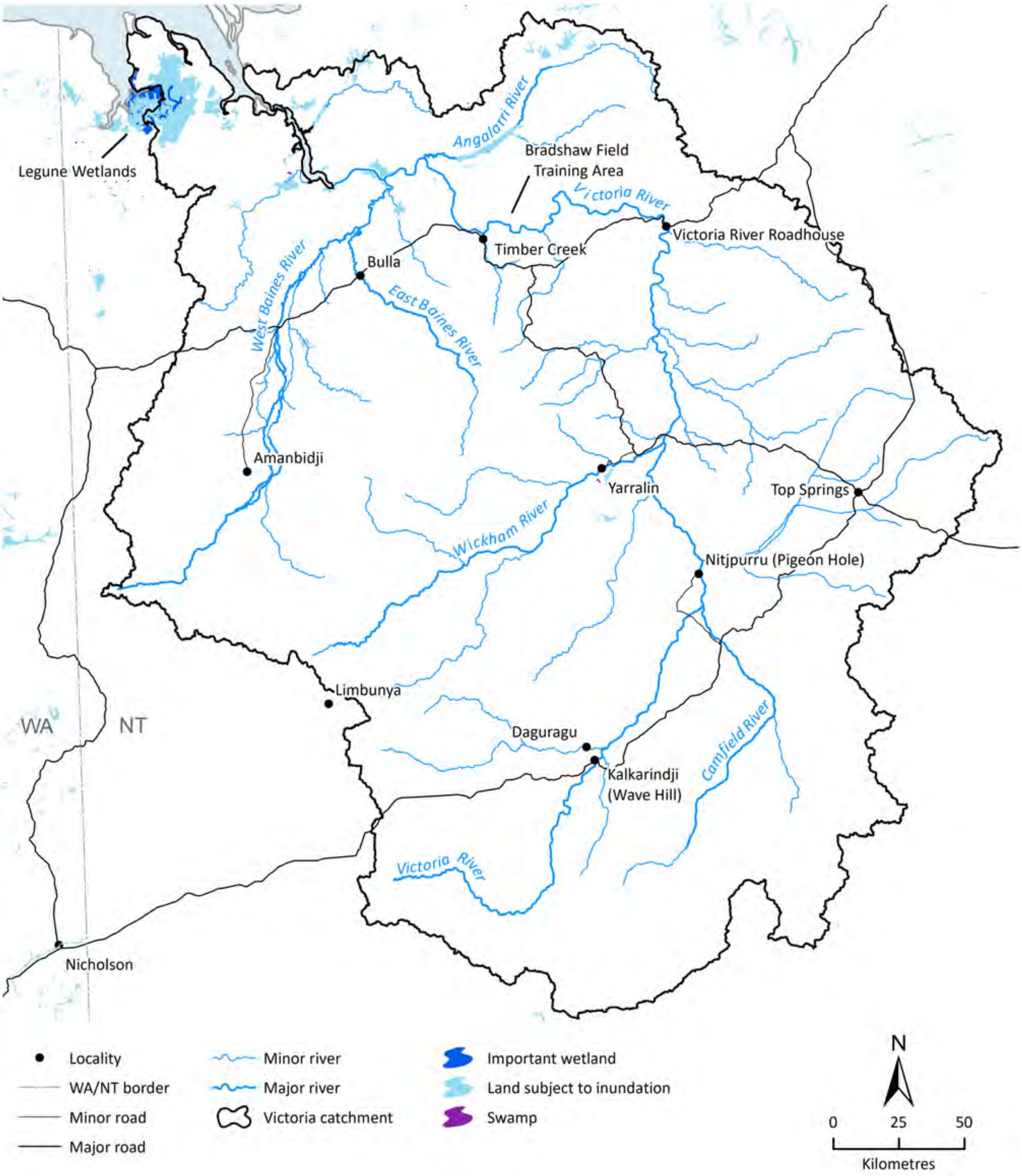

Figure 3-51 Location of land subject to inundation (potential floodplain wetlands) and nationally important wetlands (DIWA) in the Victoria catchment

DIWA = Directory of Important Wetlands in Australia

Data sources: Geoscience Australia (2017); Department of the Environment and Energy (2010)

Flow–ecology relationships for floodplain wetlands

The inundation pattern, including the extent, duration, depth, rate of inundation and timing are important factors for maintaining the ecological function of wetlands (Bunn and Arthington, 2002; Pettit et al., 2017). The pattern of connectivity is important for the movement of nutrients and biota on and off the floodplain (Junk et al., 1989). Changing the pattern of connectivity can change primary production on the floodplain, which is thought to be a major determinant of the level of species diversity, productivity and habitat structure (Close et al., 2015). This, in turn, can affect the productivity of the overall system (Brodie and Mitchell, 2005; Hamilton, 2010). The timing and duration of flooding events can be important factors determining the success of a breeding event (e.g. bird nesting, fish spawning) (Close et al., 2012). The extent of the flood influences the extent to which habitat is provided for biota. A reduction in the flood extent will reduce suitable habitat available to biota and potentially the viability of populations (Bunn and Arthington, 2002). Table 3-20 outlines these important ecological functions and their corresponding flow component or attribute.

Table 3-20 Ecological functions supporting floodplain wetlands and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Movement of nutrients and biota on and off the floodplain	River–floodplain connectivity pattern	Occurrence and frequency of overbank flows
Providing habitat for spawning and breeding events	Event timing and duration	Flow seasonality and event duration
Providing sufficient habitat to biota	Flood extent	Flood magnitude

Pathways to change for floodplain wetlands

Several threatening processes could affect the persistence of wetlands in the Victoria catchment, including river regulation, water extraction, climate change and land use change.

River regulation can affect wetlands when regulating structures capture flows and there is a downstream demand for water. Instream dams can have a significant impact on the immediate upstream and downstream environments, but may have less impact lower down in the catchment if the storage is located in the upper catchment and if other tributaries are unregulated (Petheram et al., 2008).

The ecological impacts of dams on wetlands can be numerous. Dams can prevent water from flowing onto floodplain wetlands by capturing water from moderate to large rainfall events, preventing flood pulses from moving down the channel (Kingsford, 2000). This loss of connectivity to the floodplain can result in the reduction of wetland area, and even loss of wetlands, as they may transition into terrestrial habitats. Disruption to the natural flow regime, including altered magnitude, frequency, duration, timing and rate of change of flows within a system, can affect all aspects of a riverine ecosystem (Poff and Zimmerman, 2010). These aspects include the structure, function and biodiversity of wetland ecosystems (Poff and Zimmerman, 2010; Richter et al., 1996).

Water extraction can be from both groundwater and surface water sources, the latter being rivers or standing water bodies, such as lakes or wetlands. For a range of reasons, extraction of surface water generally has less impact on the environment than instream storages. One reason is that surface water extraction can occur during high-flow events, such as floods, and not during low-flow periods (Petheram et al., 2008). Water extraction can lower the quantity of water in the river, providing less water for the environment. Reduced flooding extent and duration is also likely to reduce local groundwater recharge and thus reduce groundwater flows back into wetlands once floodwaters recede, placing pressure on floodplain wetland ecosystems that depend on groundwater discharge to sustain them during dry periods (Froend and Horwitz (2018); refer to aquatic GDEs section 3.4.2).

Climate change is a major threat to wetlands (Salimi et al., 2021). Future changes in the climate may affect rainfall, runoff and evapotranspiration patterns (Grieger et al., 2020; Salimi et al., 2021), affecting the hydrology of a system, including the baseflow and flood patterns (Erwin, 2009). Changes to the hydrology can also affect the water quality through, for example, increased erosion and changes to water temperature (Erwin, 2009). Changes to the hydrology and water temperature of wetlands can affect their biogeochemistry and function, and therefore the ecosystem services that they provide (Salimi et al., 2021).

Climate change, including changes in precipitation and rates of evaporation, can affect the quantity of inflows to a river. Vulnerability of wetlands to climate change depends on the hydrology conditions experienced and the wetlands' positions within the landscape (Winter, 2000). Hydrological landscapes are defined by their water source and their flow characteristics. Winter (2000) found that wetlands that depended on rainfall were more vulnerable to changes in climate than wetlands that depended on regional groundwater, due to the buffering capacity of groundwater systems. Coastal wetlands in particular may be vulnerable to climate change. Climate change impacts on coastal wetlands may include accelerated sea-level rise, a change in freshwater inputs, and changes to the frequency and intensity of storms and storm surges (Day et al., 2008; Nicholls et al., 1999). Sea-level rise and reduced freshwater inputs can lead to saltwater intrusion of wetlands (Close et al., 2015; Close et al., 2012), which in turn can convert freshwater floodplains to saline habitats (Finlayson et al., 1999).

Land use change can include modification of land management practices, changes to the intensity or type of agricultural production, increased vegetation clearing, or increased mining or urbanisation. These changes can affect water quality by increasing nutrient loads, sediment and turbidity levels (Finlayson et al., 1999). Changes to land use can also increase the likelihood of invasive species, due to the increased level of disturbance (Finlayson et al., 1999).

Taken individually, these threats can each have significant impacts on wetlands and their ability to provide ecosystem services and habitat, and the interactions of these threats can compound these impacts. The ecological outcomes of threatening processes on floodplain wetlands in northern Australia, with their implications for changes to floodplain wetland biodiversity and function, are presented in Figure 3-52.

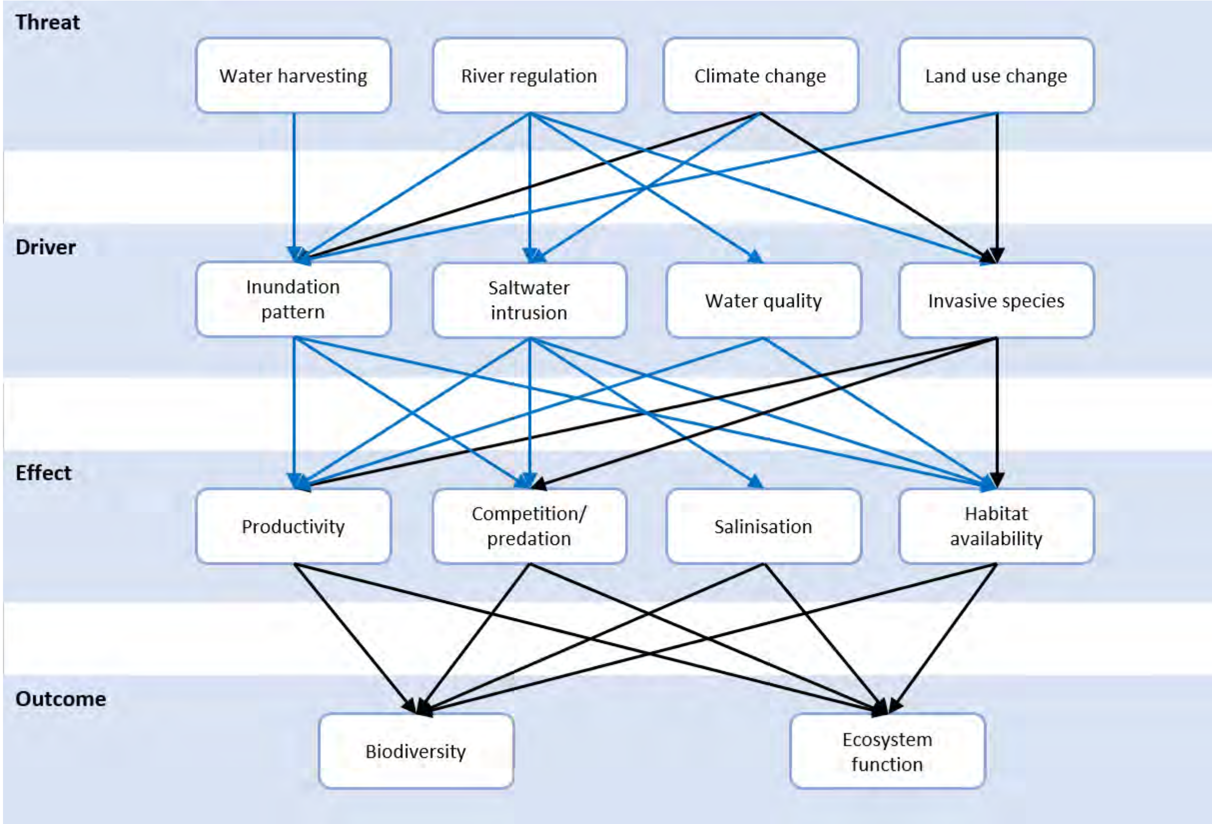

Figure 3-52 Conceptual model showing the relationship between threats, drivers, effects and outcomes for floodplain wetlands in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4.2 Groundwater-dependent ecosystems

Description and background to ecology

GDEs are defined as habitats that require groundwater at critical times (continuously, seasonally or only sporadically) to continue their existence and support the plants and animals that inhabit them and other ecosystem functions and services they provide (modified from Richardson et al. (2011b)). For example, in these habitats, groundwater may support vegetation such as the red cabbage palm (*Livistonia mariae*; Box et al. (2008)) in areas where it would not otherwise persist, fish may persist in groundwater-fed waterholes during dry seasons (e.g. McNeil et al., 2013), and animals that live exclusively in underground water cavities maintained by groundwater (e.g. stygofauna in karst aquifers; Oberprieler et al. (2021)). In the wet-dry tropics typical of northern Australia, groundwater is important, being recharged over wet periods and supporting ecological function of water-dependent habitats and species during dry periods. GDEs are typically categorised into three functional types:

- aquatic groundwater-dependent ecosystems
- terrestrial groundwater-dependent ecosystems
- subterranean groundwater-dependent ecosystems.

These functional types are described in the following sections.

Aquatic groundwater-dependent ecosystems

Aquatic GDEs are surface water habitats that require groundwater discharge to the surface or the presence of near-surface groundwater (e.g. for hyporheic exchange, which is mixing between surface water with shallow subsurface water in the sediment surrounding rivers and wetlands). They include groundwater-fed spring, wetland, river, estuary and coastal (submarine groundwater discharge) ecosystems. The loss of groundwater can have extreme consequences, such as the complete drying out of mound springs and loss of all dependent species (e.g. Fairfax and Fensham, 2002).

Habitats largely supported by surface water flow can still rely on groundwater at specific times or to maintain processes, such as maintaining the quality or temperature of water available (e.g. for fish spawning (Geist et al., 2002)) or nutrients for animal and plant growth (Moore (2010)). The impacts of reduced groundwater can appear over long periods and may lead to lower recruitment, loss of species diversity and abundance, proliferation of invasive species, and changes in the structure and function of the ecosystem (e.g. Nevill et al., 2010).

Terrestrial groundwater-dependent ecosystems

Terrestrial GDEs are vegetated habitats supported by subsurface groundwater, for example, trees that use groundwater and the various plants and animals supported by the habitat the trees provide. Groundwater-dependent terrestrial vegetation requires access to groundwater at critical times for survival (varies depending on species, climate, environment and soil water–holding properties), flowering and successful recruitment (e.g. Horner et al. (2009)). Some terrestrial vegetation species only occur where groundwater is available (obligate GDEs), while other species use groundwater in some habitats (facultative GDEs) but can also exist in habitats where sufficient water within unsaturated soils (driven by climate and plant-available water capacity of soils) removes the need for groundwater (e.g. Pritchard et al., 2010). Regardless of the species, mature vegetation is unlikely to be able to adapt to changes in water availability outside natural variation (e.g. threshold responses; Kath et al. (2014)). Terrestrial GDEs have some inbuilt resilience to changes in water availability and quality, but long-term change in groundwater regime (driven by water resource development or climate change) is likely to result in dieback of groundwater-dependent vegetation (whether obligate or facultative) after some lag period. Dieback of groundwater-dependent vegetation may have broad environmental implications, causing shifts in ecosystem composition and structure (change in the density and diversity of species) and function (e.g. change in the ecosystem’s ability to provide suitable food or habitat for animal species, e.g. Betts et al. (2010), Fleming et al. (2021)).

Obligate versus facultative GDEs – challenging definitions

A common misconception has broadly propagated though GDE literature that the term ‘obligate GDE’ refers to ecosystems that require a permanent source of groundwater, and the term ‘facultative GDE’ refers to ecosystems that only use groundwater opportunistically, implying that groundwater is not critical to the survival of the ecosystem and that facultative GDEs will survive if groundwater availability is permanently removed. This definition is misleading. Facultative GDEs will become degraded if groundwater is not available at critical times. Therefore, within this project, the terms are defined as follows:

Obligate GDE: an ecosystem that will only naturally occur where groundwater is available at critical times (this may be continuous, seasonally or sporadically).

Facultative GDE: an ecosystem that naturally occurs in some environments (under specific climate and site conditions) in which it must receive groundwater at critical times (this may be continuous, seasonally or sporadically), but it can also occur in other environments in which it naturally receives enough water from other sources (e.g. rainfall, surface water flows, unsaturated soil stores) that it never uses groundwater.

In the case of facultative GDEs, groundwater dependence cannot be proven based on species composition alone. Further studies will be required to determine sources of water used.

For example, *Melaleuca leucadendra* uses groundwater in some environments (Canham et al., 2021) but not in others (O'Grady et al., 2006).

Figure 3-53 demonstrates that obligate groundwater-dependent vegetation only occurs in parts of the landscape where there is a reliable source of groundwater. In contrast, facultative GDEs grow and depend on groundwater in some areas but can also establish and thrive in areas where there is sufficient soil water to sustain them without ever having access to groundwater. Obligate GDEs are always vulnerable to unprecedented declines in groundwater availability. Facultative GDEs are vulnerable to groundwater declines in some parts of the landscape, but in other parts they may not require groundwater. Further site assessment is required to establish water dependence of facultative GDE species.

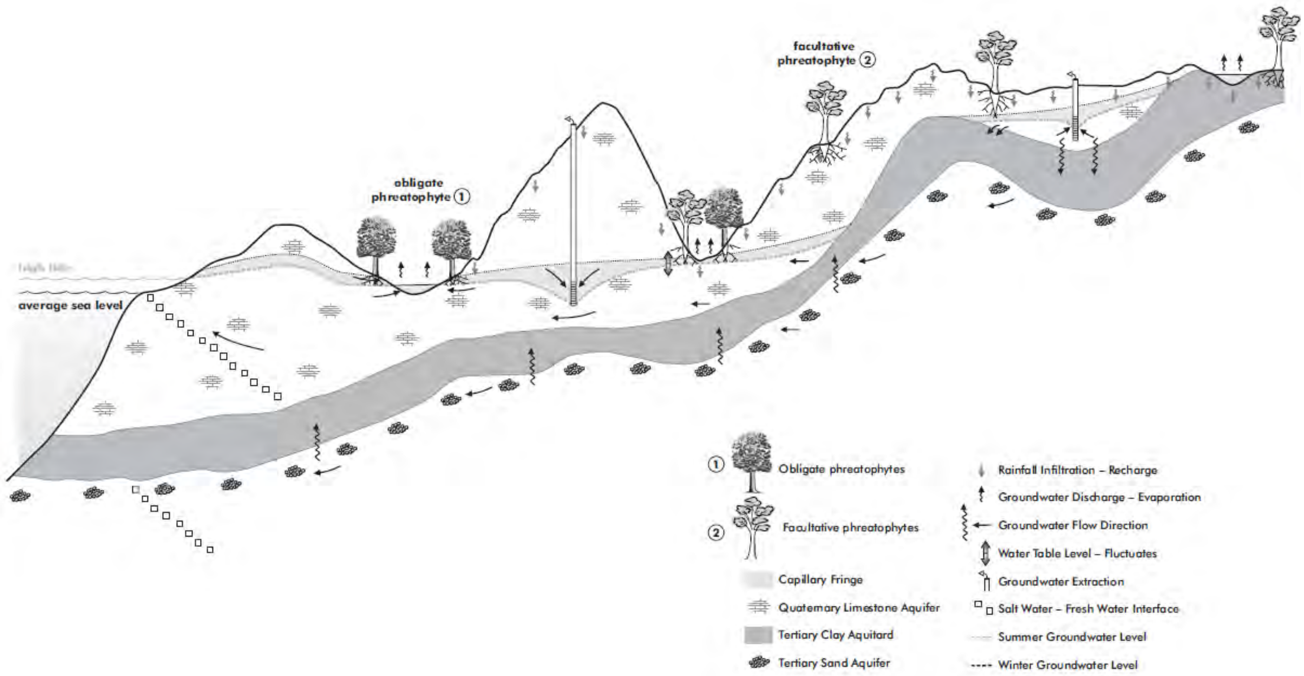

Figure 3-53 Conceptualisation of obligate and facultative groundwater-dependent vegetation

Phreatophytes are vegetation that draw their water from near the water table.

Source: Pritchard et al. (2010)

Figure 3-54 depicts natural terrestrial vegetation vigour associated with changes in water availability. During the wet season (I), vegetation has access to sufficient water and productivity and diversity are high. During the dry season (II), there is reduced water availability and productivity. As soils dry, annual vegetation species die back while deeper-rooted species stay green through access to deeper soil water or groundwater. If water availability is reduced beyond natural dry-season variation (III), deeper-rooted species also die back once deeper soil water and groundwater resources become inaccessible. This is likely to result in a shift in ecosystem type (e.g. forest to savanna) and makes the ecosystem more susceptible to invasive plants.

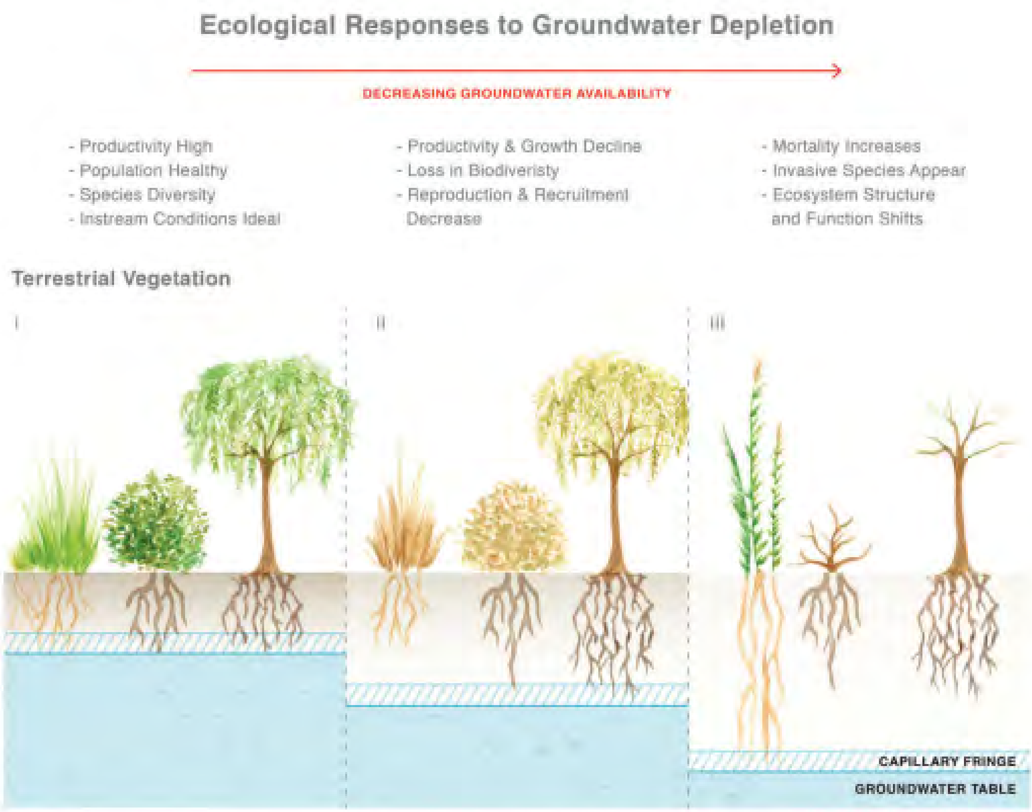

Figure 3-54 Conceptualisation of terrestrial GDEs: (I) vigorous ecosystems with seasonally high water availability, (II) ecosystem condition with seasonally low water availability, and (III) seasonal low after groundwater development

Source: Rohde et al. (2017)

Subterranean groundwater-dependent ecosystems

Subterranean GDEs are cave and aquifer systems that provide habitat for subterranean fauna that depend on the presence of groundwater (e.g. troglofauna (cave dwelling) and stygofauna (aquifer dwelling); Richardson et al. (2011a)). Subterranean fauna have limited mobility, and changes in groundwater beyond natural fluctuation in watertable elevation or groundwater quality risks loss of the local communities (Hose et al., 2015). Some subterranean fauna are only known to exist in discrete localities (e.g. Hancock and Boulton, 2008), so loss of local communities can result in species extinction. Apart from their intrinsic biodiversity value, subterranean ecosystems are indicators of groundwater health, and they potentially provide ecosystem services such as nutrient cycling and water purification (Glanville et al., 2016; Smith et al., 2016).

GDEs in the Victoria catchment

The National GDE Atlas (Bureau of Meteorology, 2017) contains maps of the distribution of known and potential groundwater-dependent inland aquatic and terrestrial ecosystems. Mapping of potential GDEs within the GDE Atlas was based on the location of known GDEs and their extrapolation to regional scales using a process that integrated expert opinion, remote sensing data (2000 to 2010) and geographic information systems (Doody et al., 2017). Little is known about coastal or submarine groundwater discharge along the northern coast of Australia.

In Australia, the biodiversity and distribution of subterranean ecosystems remain largely unknown. Three types of aquifers are known to provide subterranean ecosystems that can support stygofauna: karstic, fractured rock and alluvial. Typically these occur where the depth to groundwater is less than 30 m (Doody et al., 2019), but some have recently been found to depths of 70 m (Oberprieler et al. (2021)). All these aquifer types occur in the Victoria catchment (see section on subterranean GDEs below).

Aquatic GDEs in the Victoria catchment

Most rivers in the Victoria catchment cease flowing and become a series of disconnected pools during the dry season; only sections of Wickham River (upstream of the Humbert River junction) and Angalarri River maintain flow year round (Midgley, 1981) above tidal influence. Studies have shown that the Victoria catchment contains groundwater-fed springs that persist during most dry seasons (Bureau of Meteorology, 2017), and these habitats support aquatic life and fringing vegetation, particularly during the dry season. These aquatic ecosystems are mapped as ‘known GDEs’ in the GDE Atlas (Figure 3-56) and tend to occur within or adjacent to perennial rivers in the Victoria catchment. There are also hundreds of river sections, lakes and wetlands that are believed to be supported by groundwater discharge based on remote sensing work and expert opinion, and these are mapped as ‘potential GDEs’ (Bureau of Meteorology (2017); Figure 3-56). These ecosystems may not always contain surface water throughout the dry season but are thought to be supported by groundwater discharge for some of the year.

Figure 3-55 Bullshead Spring, Victoria River Downs Station

Photo attribution: CSIRO

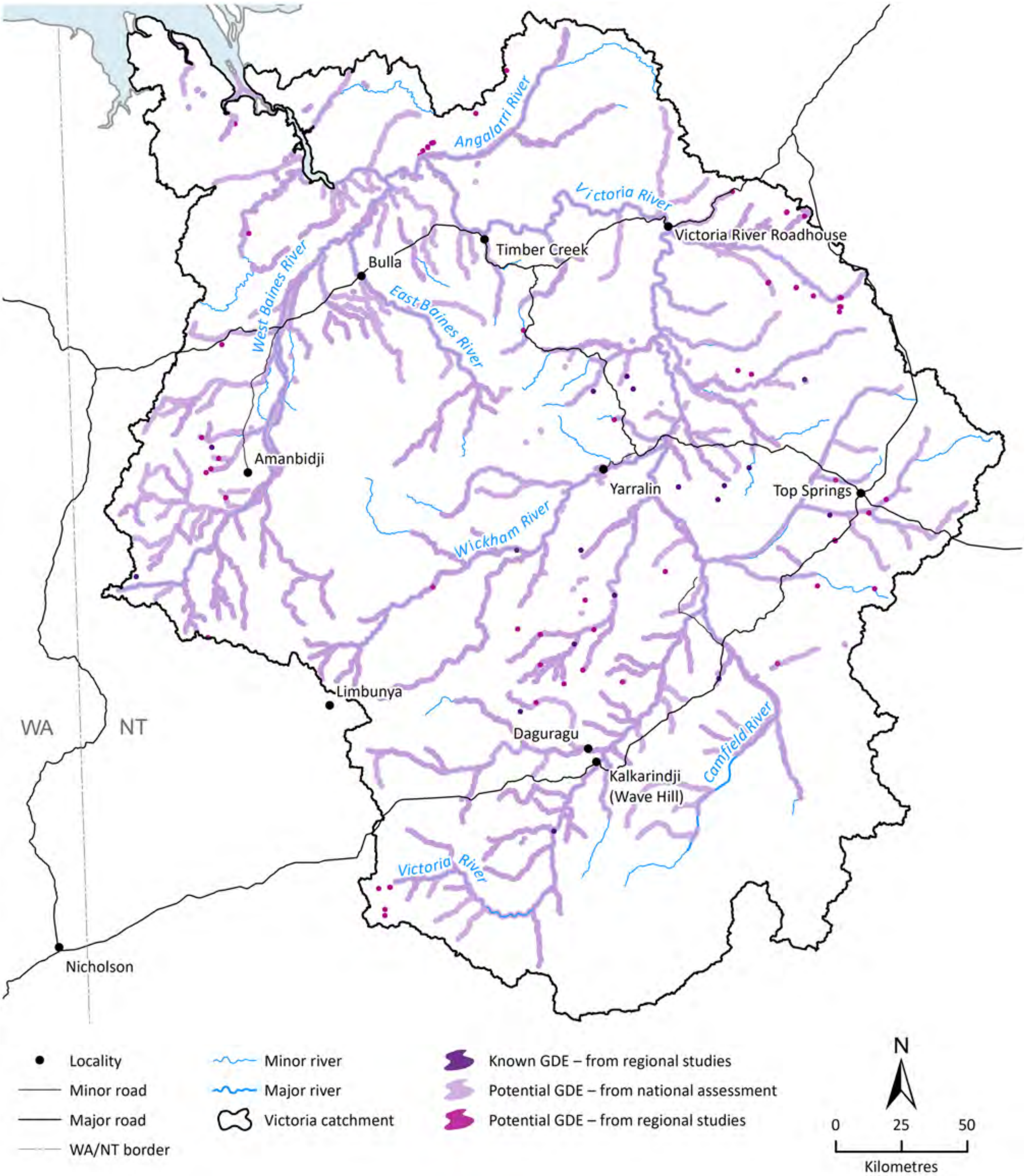

Figure 3-56 Distribution of known and potential groundwater-dependent aquatic ecosystems in the Victoria catchment

Note: A buffer of 1 km has been applied to GDE mapping so that they are visible on the map scale.

Dataset: Bureau of Meteorology (2017)

Most springs mapped in the Victoria catchment (Figure 3-57) are also included as aquatic GDEs in the GDE Atlas dataset (Figure 3-56). It is assumed that because these surface water features are labelled springs, they are groundwater discharge features and should be considered aquatic GDEs. Known sinkholes associated with the Montejinni Limestone are mapped along the south-eastern edge of the Victoria catchment (Figure 3-57). Sinkholes may contain groundwater and support aquatic ecosystems throughout the dry season, but their connection to groundwater is currently unknown.

Little is known about coastal or submarine groundwater discharge in the Victoria catchment. Global-scale modelling suggests there is potential for submarine groundwater discharge off the coast of the Victoria River (Luijendijk et al., 2020), but there have been no local-scale studies to substantiate this.

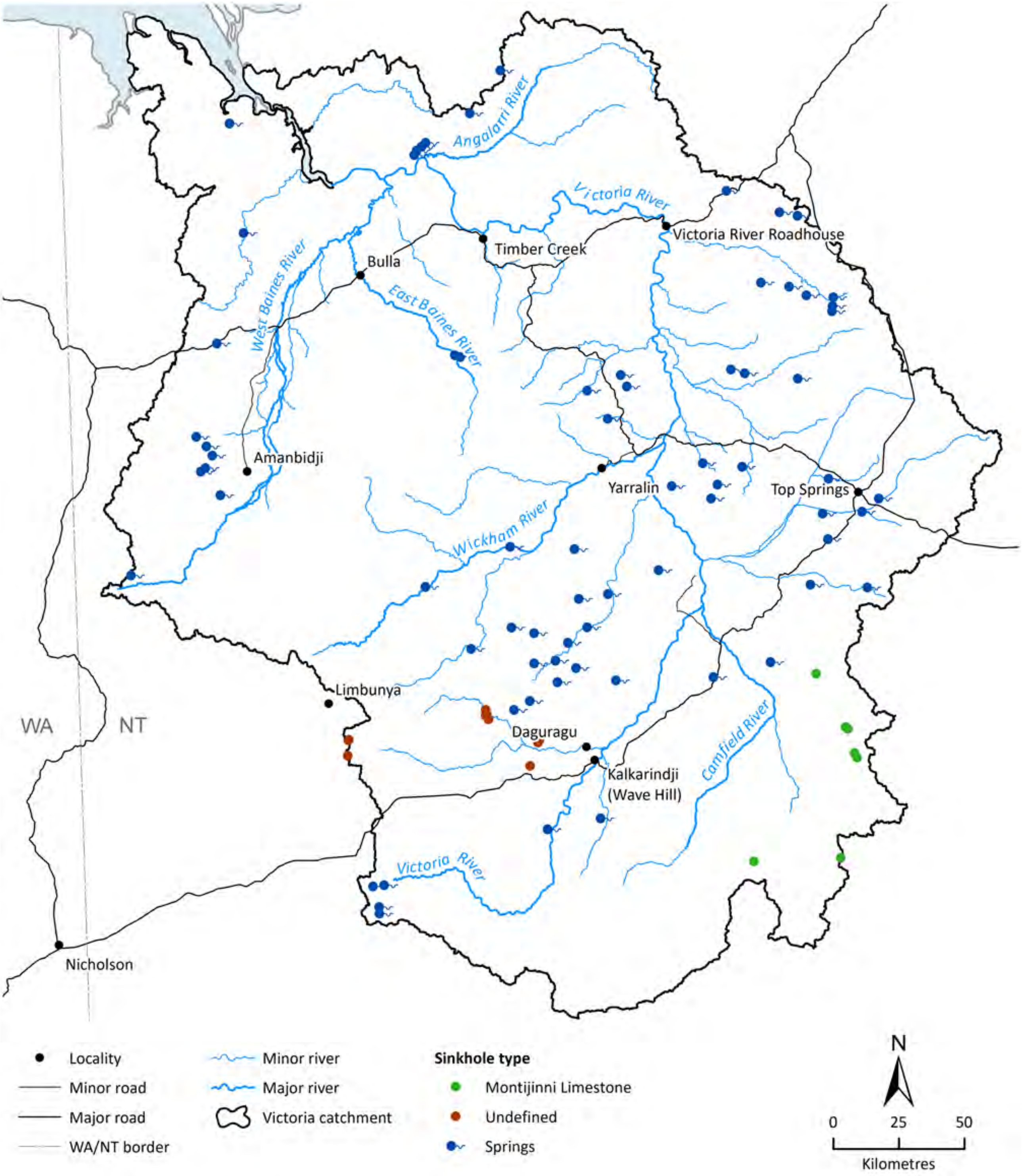

Figure 3-57 Locations of springs and sinkholes in the Victoria catchment

Dataset: Department of Environment, Parks and Water Security (NT) (2013; 2014)

Terrestrial GDEs in the Victoria catchment

GDE Atlas analysis (based on remote sensing work and expert opinion) suggests that groundwater-dependent vegetation potentially occurs along sections of the Victoria River, Wickham River, West and East Baines rivers and many of their major tributaries; these areas are mapped as 'potential GDEs' (Bureau of Meteorology (2017); Figure 3-58). However, no on-ground studies were available to verify groundwater dependence of vegetation within the Victoria catchment.

Terrestrial GDEs mapped in the GDE Atlas within the Victoria catchment include monsoon vine forests and *Melaleuca* forests and woodlands (including *Melaleuca cajuputi*, *M. viridiflora*, *M. argentea*, *M. leucadendra*, *M. sericea*). A number of other terrestrial vegetation species occurring in the Victoria catchment are also likely to use groundwater (e.g. river red gum, *Eucalyptus camaldulensis*). A preliminary indication of where other potential terrestrial GDE species exist within the Victoria catchment is shown in Figure 3-59 and Figure 3-60. Figure 3-59 maps observed occurrence of three tree species (*E. camaldulensis*, *M. argentea*, *Barringtonia acutangular*) that are thought to only occur naturally where they have access to groundwater at critical times (i.e. obligate GDE species; Lamontagne et al. (2005); Mensforth et al. (1994)). Most of the observed occurrences are riparian, along the major rivers, but comprehensive species distribution mapping does not exist. *B. acutangular* is known to occur in freshwater mangroves and *M. argentea* is known to occur in floodplain and swamp habitats. Therefore, the distribution of obligate GDE vegetation species is expected to be more extensive than mapped in Figure 3-59. Figure 3-60 shows the observed occurrence of many other known GDE species grouped by vegetation types ('GDEs') that are known to use groundwater in some locations, but under some climate and/or site conditions may not be groundwater dependent (i.e. facultative GDE species). It also shows the occurrence of vegetation that are potential GDE species grouped by vegetation type ('potential GDEs'). These are species that are suspected to use groundwater, but this remains unconfirmed. A complete list of species included in the GDEs mapped in Figure 3-60 is provided in Appendix C.

Recent work by Castellazzi et al. (2023) (Figure 3-61) proposes a more detailed, finer-resolution map of the distribution of drought-resilient vegetation across the Victoria catchment than previously available. The map is formed by combining radar and optical satellite imagery products, ALA data of known locations of obligate groundwater-dependent vegetation types (Figure 3-59, Appendix C) and limited ground-truthing. It is more detailed and up to date than the groundwater-dependent vegetation mapping included in the GDE Atlas (Figure 3-58). Figure 3-61 shows that terrestrial GDEs potentially extend beyond the banks of major rivers and occupy broader areas of the landscape in Victoria catchment. The source of groundwater used by vegetation and the timing and frequency of groundwater use remain challenges for further investigation. Regional-scale groundwater flow systems exist in the Wiso Basin, which occurs in the south-west of Victoria catchment (groundwater tech reporting). The remainder of the aquifers in Victoria catchment are local- to intermediate-scale groundwater flow systems.

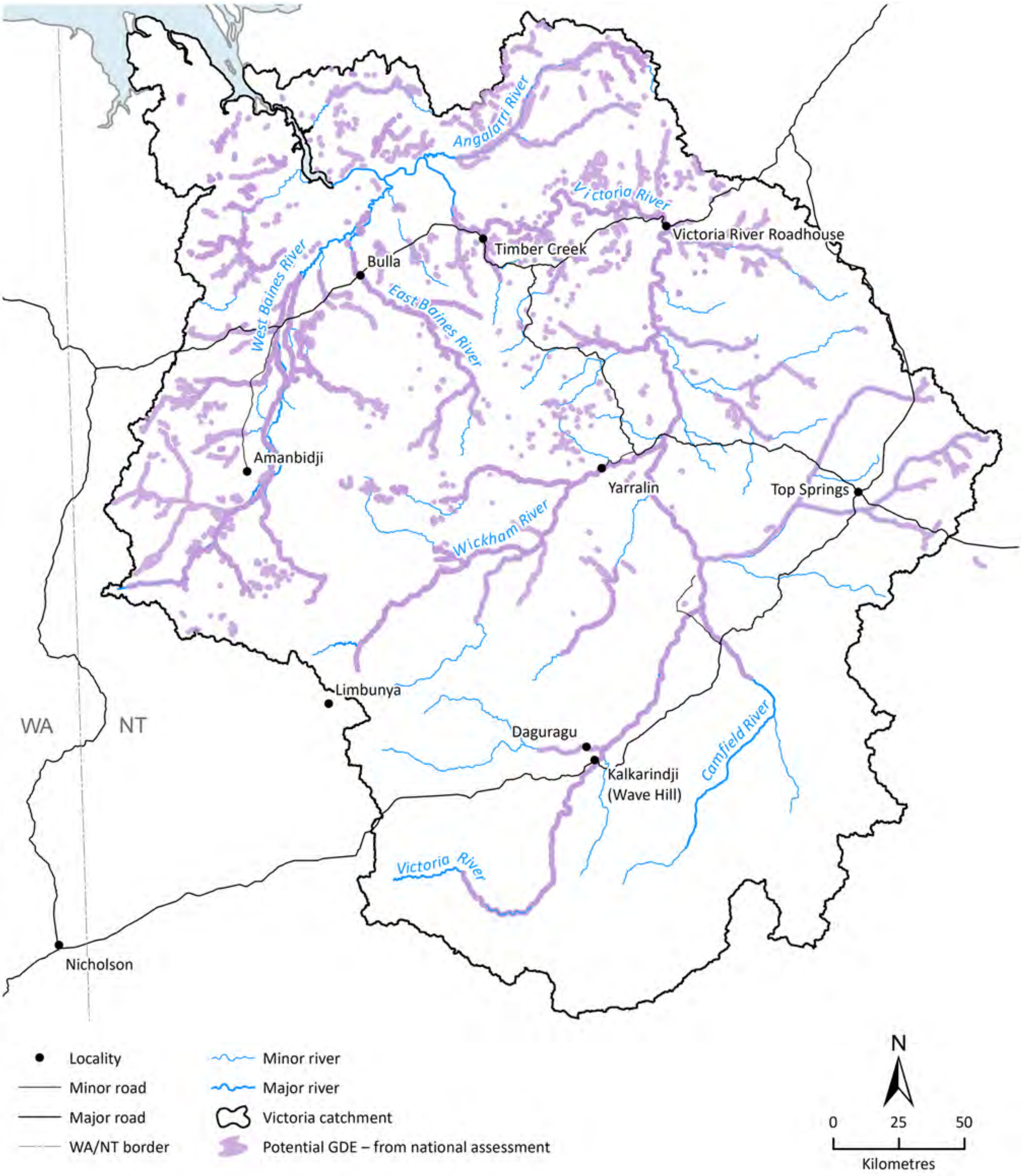

Figure 3-58 Distribution of known and potential groundwater-dependent terrestrial ecosystems in the Victoria catchment

Note: A buffer of 1 km has been applied to GDE mapping so that they are visible on the map scale.

Dataset: Bureau of Meteorology (2017)

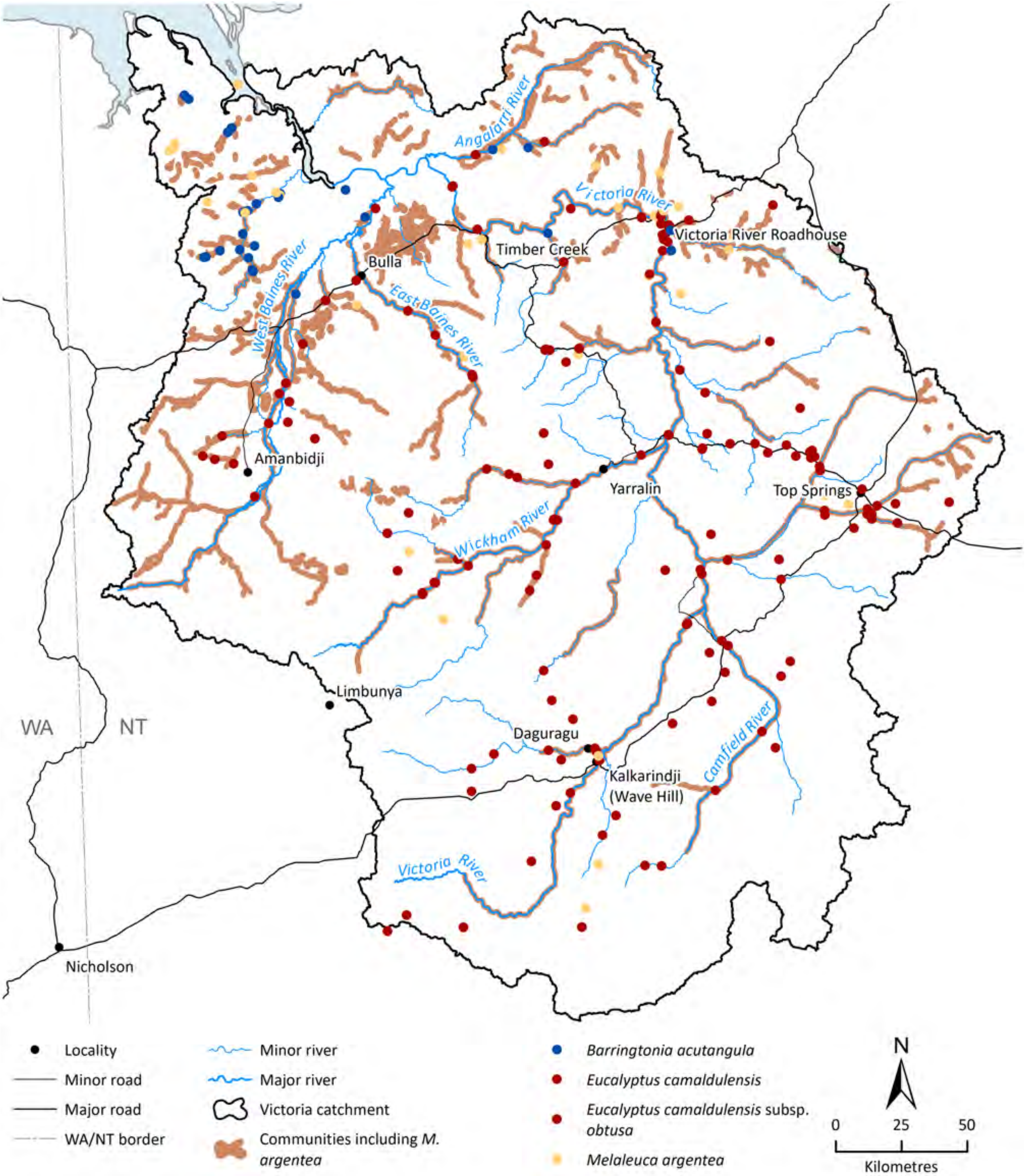

Figure 3-59 Observed locations of obligate terrestrial GDEs in the Victoria catchment

Note: A buffer of 1 km has been applied to NT *Melaleuca* mapping so that they are visible on the map scale.

Datasets: Atlas of Living Australia (2023); Department of Environment Parks and Water Security (2000a)

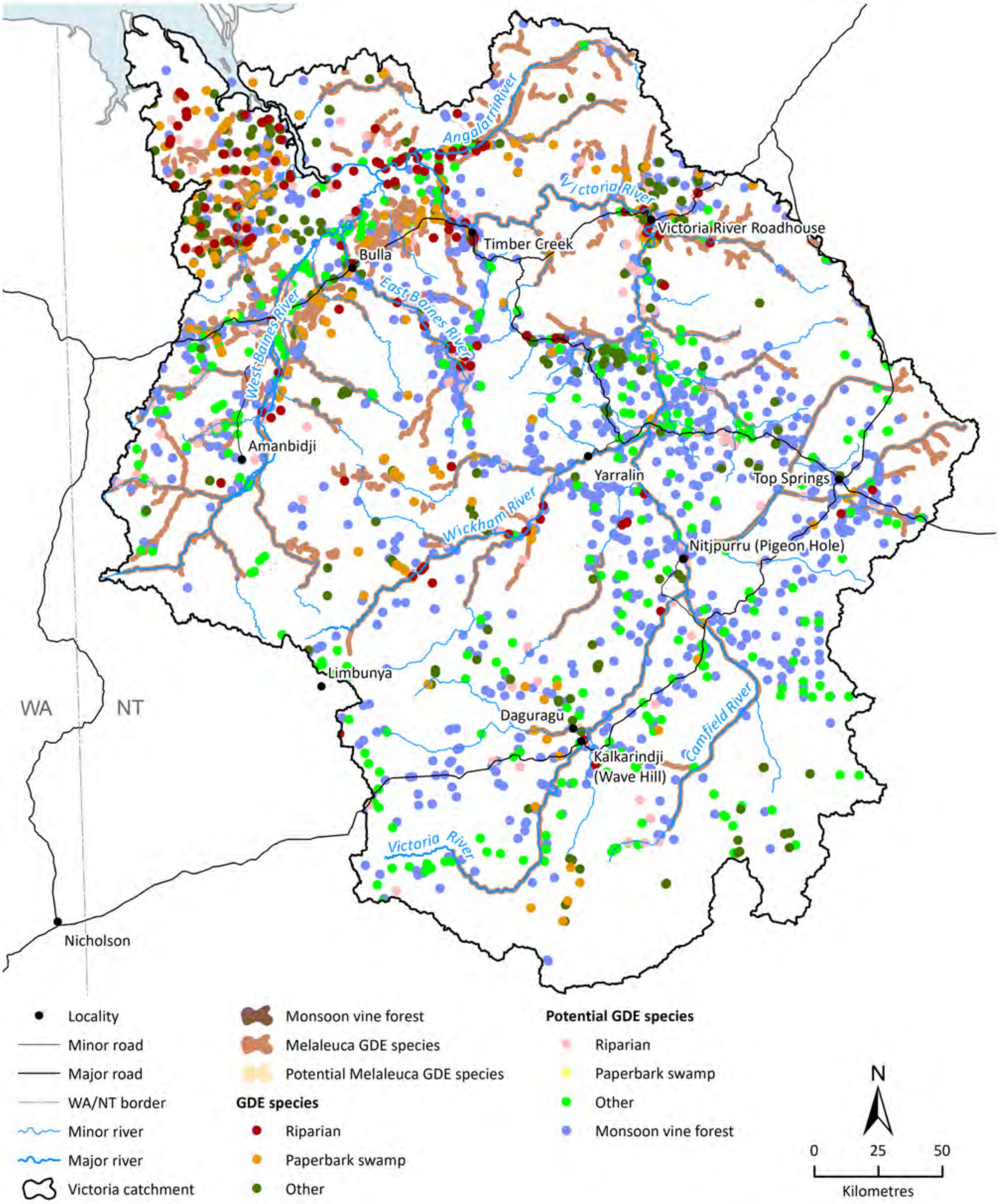

Figure 3-60 Locations of facultative and potential GDE vegetation types in the Victoria catchment grouped by relevant vegetation type

Note: A buffer of 1 km has been applied to NT *Melaleuca* and Monsoon vine forest mapping so that they are visible on the map scale.

Datasets: Atlas of Living Australia (2023); Department of Environment Parks and Water Security (2000a; 2000b)

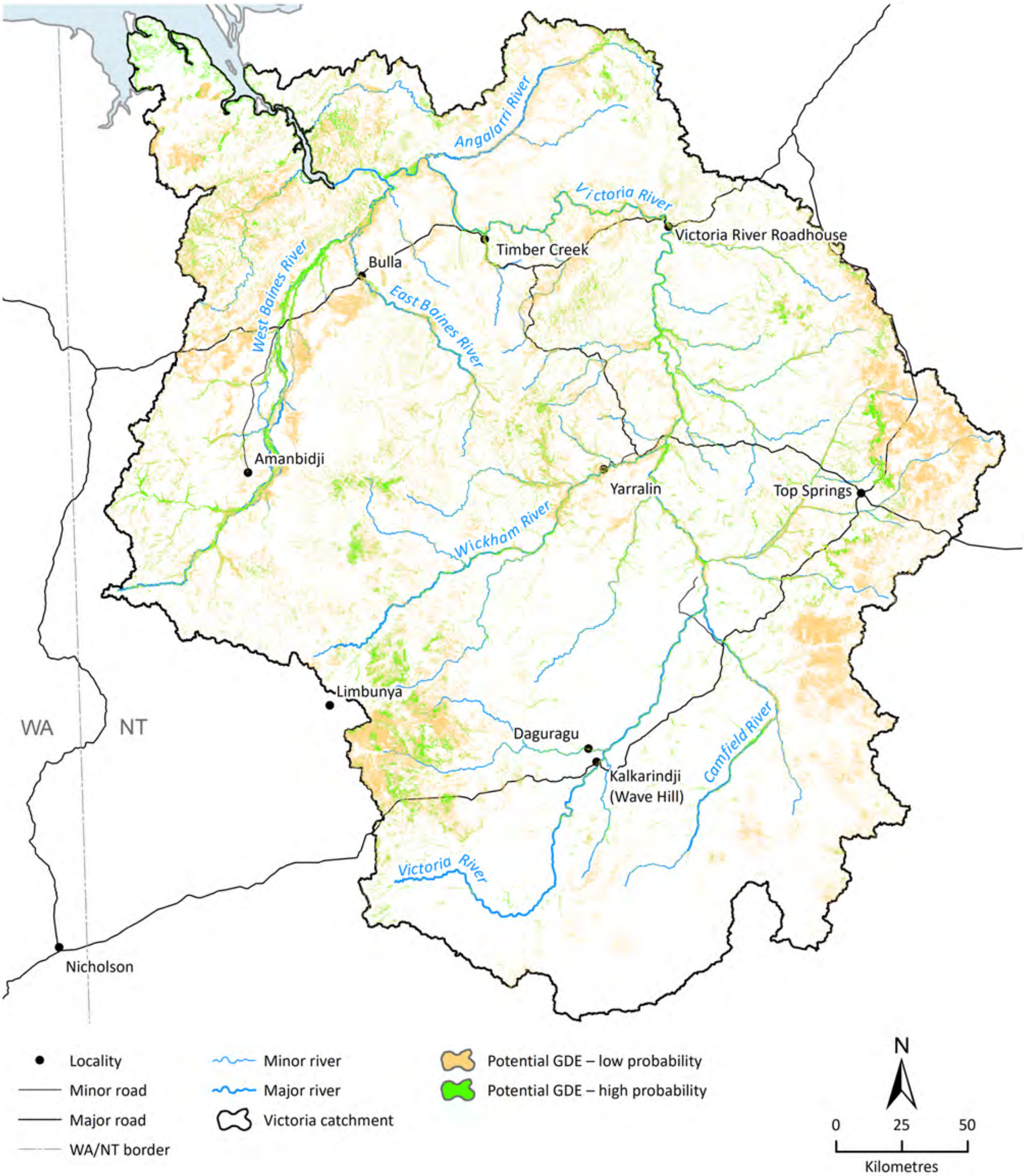

Figure 3-61 Distribution of potential groundwater-dependent vegetation in Victoria catchment

Dataset: Castellazzi et al. (2023)

Subterranean GDEs in the Victoria catchment

Subterranean aquatic ecosystem sampling is limited to Bullita Caves in the Victoria catchment, where the presence of suspected troglofauna and stygofauna species was discovered (Moulds and Bannink (2012); Figure 3-62). Figure 3-62 shows where additional subterranean GDEs may be found based on the presence of favourable habitats for subterranean GDE species based on available data (caves and alluvial and karstic aquifers).

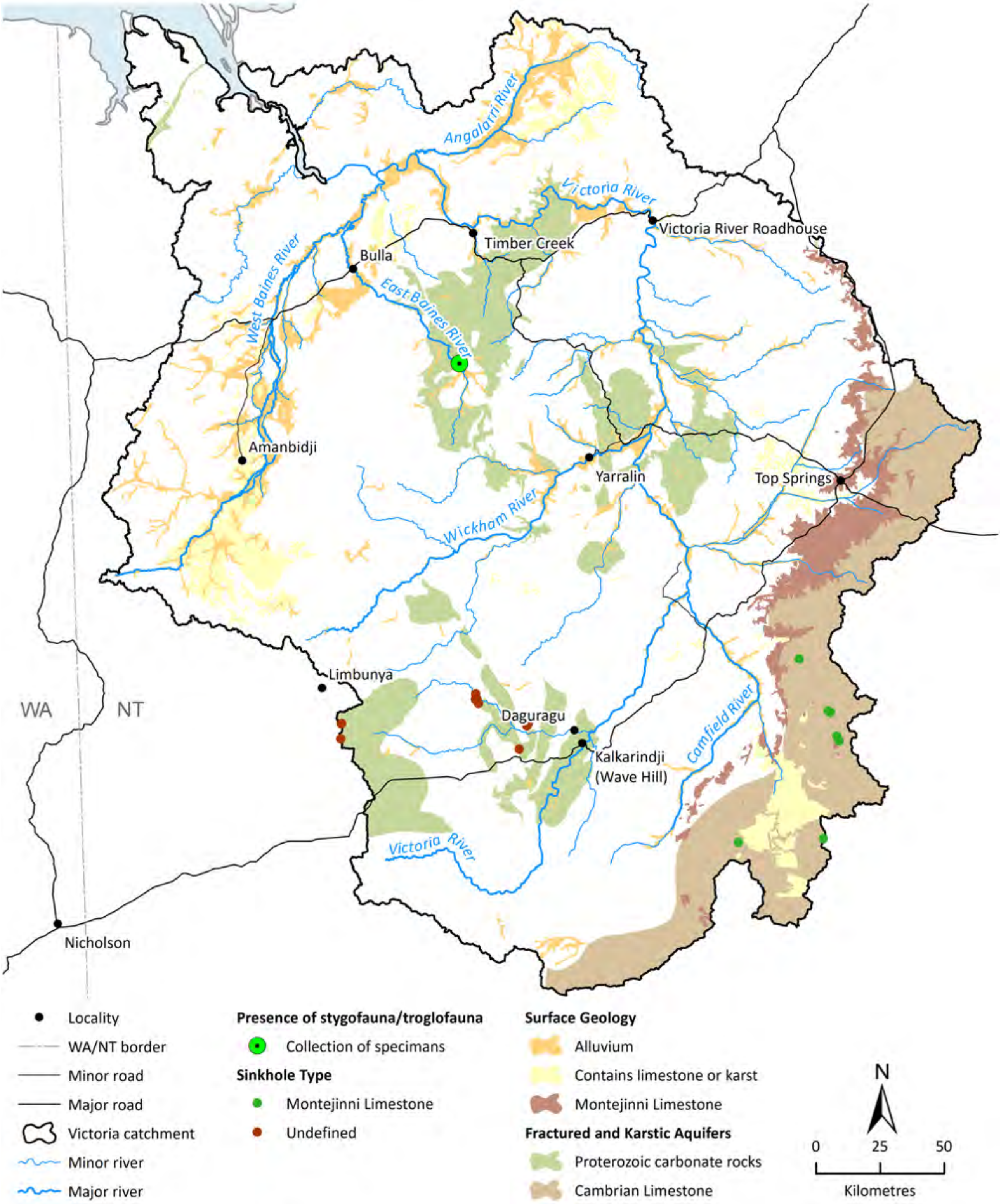

Figure 3-62 Locations tested for the presence of subterranean GDEs in the Victoria catchment and locations of caves and alluvial and karstic aquifers that may provide habitat for subterranean GDEs

Datasets: Rees et al. (2020); Department of Environment, Parks and Water Security (NT) (2014).

Flow–ecology relationships and water requirements for GDEs

GDEs are sensitive to changes in water quality and availability. Aquatic GDEs may be sustained by surface flows for much of the year, but when surface flows become low, they are often sustained by groundwater discharge. For some aquatic GDEs, there may be recruitment and/or breeding events that are exclusively triggered by groundwater discharges (this could be caused by timing and/or quality of groundwater inputs). Relationships between groundwater discharge and aquatic GDEs in northern Australia remain unknown. Floodplain wetland and inchannel waterhole flow–ecology relationships are reported in Sections 3.4.1 and 3.4.3, respectively.

Terrestrial GDEs are often sustained by a mixture of soil water and groundwater; however, some may also require periodic flooding to induce flowering and seed fall (e.g. river red gum; George (2004)) and recruitment. Groundwater requirements of terrestrial GDEs are highly variable depending on the species present and soil and climate conditions. Surface water inundation requirements for maintaining terrestrial GDE function and services are largely unknown. However, there is some crossover between groundwater-dependent and surface-water-dependent terrestrial vegetation (Section 3.4.6), for which flow–ecology relationships are reported in Table 3-25.

Most subterranean fauna have limited mobility and become stranded and die in unsaturated soils when groundwater levels drop rapidly (Hose et al., 2015). Conversely, when groundwater levels rise, subterranean fauna may not be able to rise with groundwater and become stranded in waters with insufficient oxygen to sustain them (Hose et al., 2015). The water level and quality changes that subterranean GDEs can withstand probably varies broadly with species and aquifer type, but is largely unknown. Table 3-21 specifies broad flow–ecology relationships that need to be considered when assessing the impact of changes in flow on subterranean GDEs.

Table 3-21 Ecological functions supporting GDEs and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Inundation to recharge aquifers that support subterranean GDEs	Maximum, minimum and seasonal amplitude of variation in groundwater levels	Extent and frequency of flooding recharging groundwater

Pathways of change for GDEs

Changes to GDEs from water resource development can occur due to a range of different processes, depending on the type of water resource development and how it is managed. This section discusses the impacts of water harvesting (from groundwater and directly from rivers), dam infrastructure and regulation of river flows, climate change and land use change on aquatic (Figure 3-63), terrestrial (Figure 3-64) and subterranean (Figure 3-65) GDEs.

GDEs are inherently sensitive to changes in the availability and quality of groundwater at critical times, but most are highly dependent on surface water as well. Groundwater drawdown near GDEs may result in reduced discharge to aquatic GDEs (e.g. wetlands, rivers), reduced connection between groundwater and dependent vegetation (terrestrial GDEs) or loss of subterranean GDEs altogether.

Surface water harvesting, river regulation, dam infrastructure, climate change and land use change can all disturb the natural groundwater recharge regime, altering the depth to water, the seasonal cyclicity of groundwater levels, and groundwater quality. In areas where groundwater recharge is reduced, the impacts on GDEs over the long term are similar to those of groundwater drawdown. In areas where groundwater recharge is enhanced, there could be:

- local increases in groundwater discharge to aquatic GDEs. In some areas this can be a source of high salt loads to surface water systems (e.g. Jolly et al. (1993)) that potentially increase the longitudinal connectivity along rivers during the dry season, putting pressure on some aquatic GDE species and potentially favouring non-native aquatic species (e.g. Yarnell et al. (2015))
- shallower groundwater levels, potentially leading to soil salinisation due to evapotranspiration from shallow watertables (e.g. Smith and Price, 2009) and/or a shift in the type of terrestrial vegetation supported (e.g. from *Melaleuca* swamp to grassland (Department of Environment and Science Queensland, 2013))
- potential mortality of subterranean GDEs in anoxic waters if stygofauna lose connection with relatively aerated water at the top of the watertable (Hose et al., 2015).

Most aquatic and terrestrial GDEs require surface water in addition to groundwater to sustain their water requirements. Activities that affect the volume, timing, frequency and quality of surface water flows or inundation are likely to affect aquatic GDEs and fringing vegetation. The ecological outcomes of threatening processes on aquatic, terrestrial and subterranean GDEs in northern Australia, and their implications for changes to biodiversity and ecosystem function, are illustrated in Figure 3-63, Figure 3-64 and Figure 3-65, respectively.

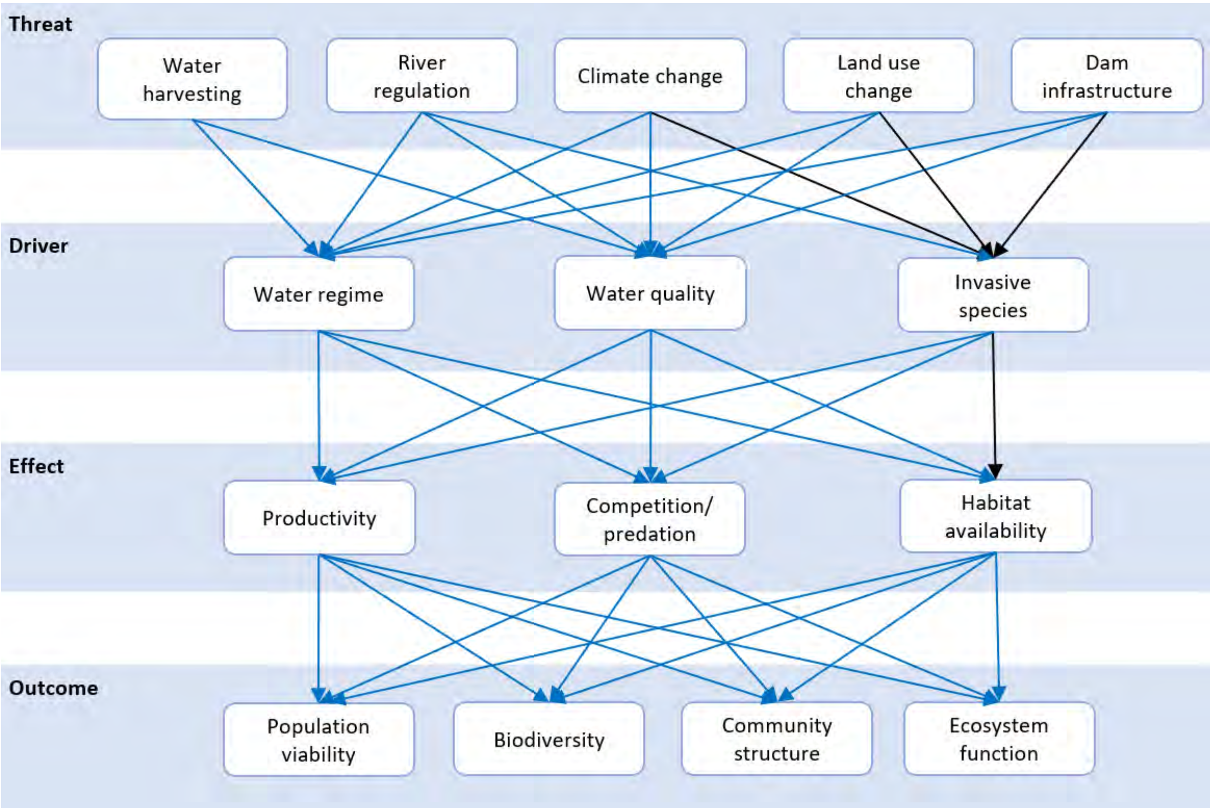

Figure 3-63 Conceptual model showing the relationship between threats, drivers, effects and outcomes for aquatic GDEs in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

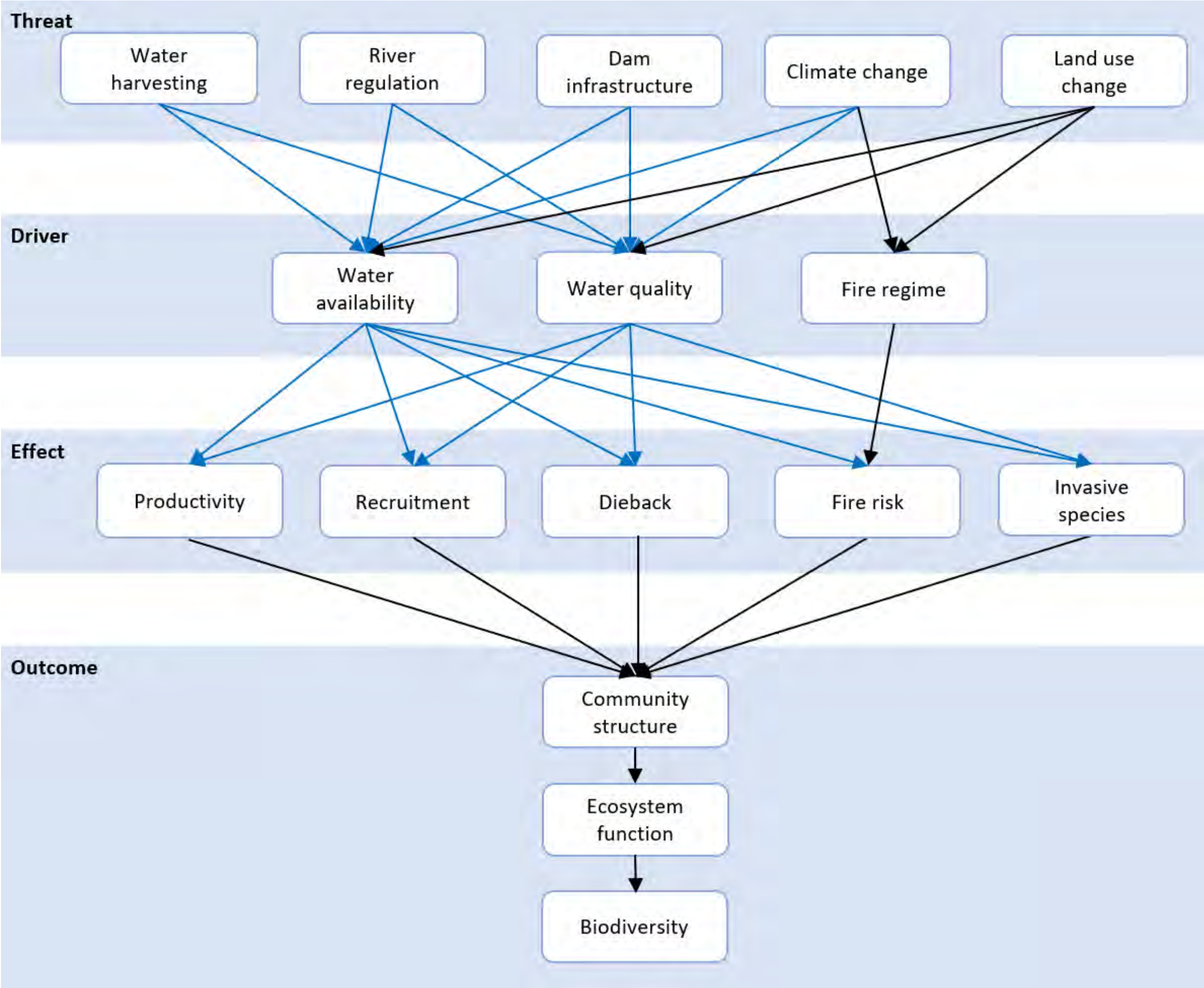

Figure 3-64 Conceptual model showing the relationship between threats, drivers, effects and outcomes for terrestrial GDEs in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

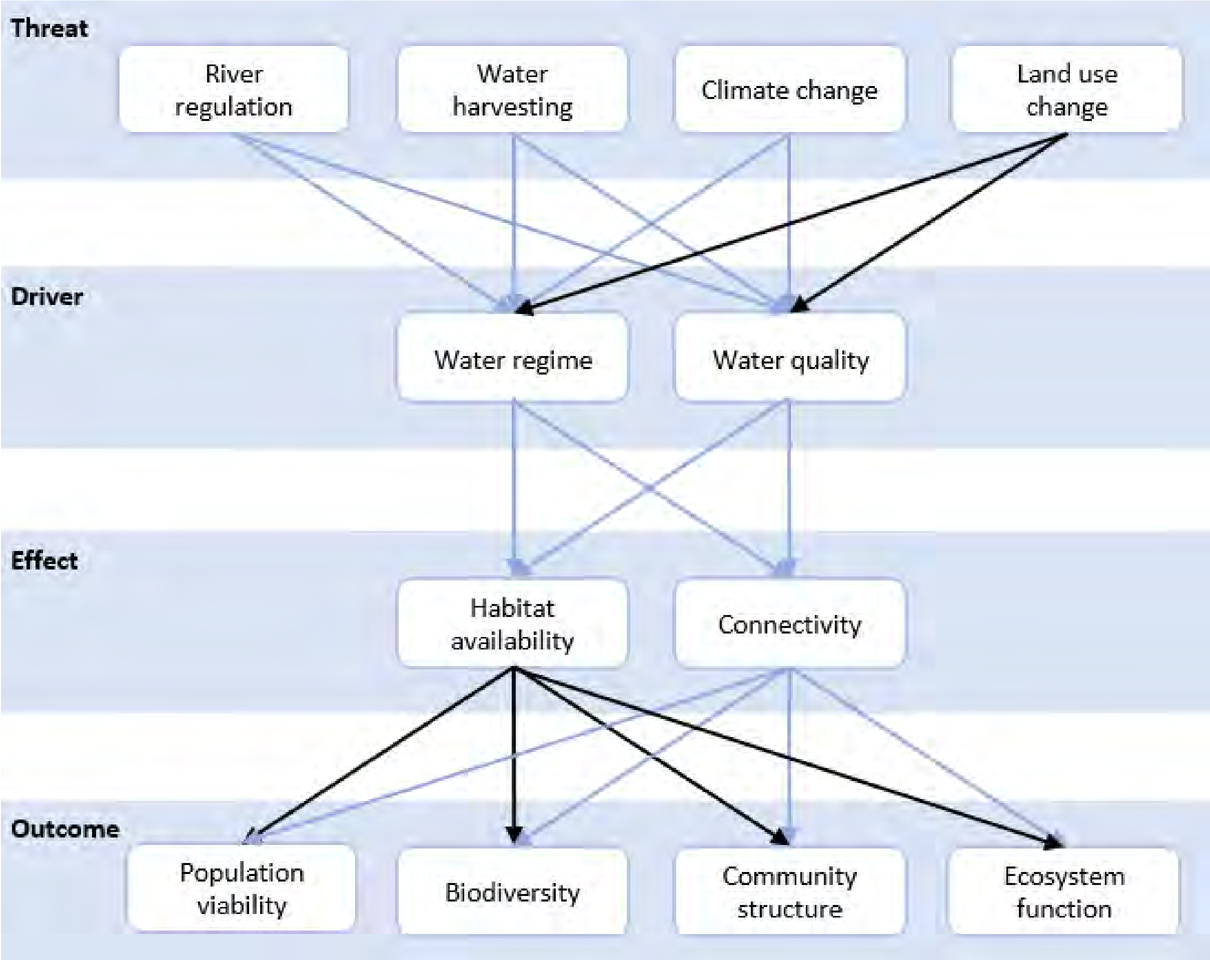

Figure 3-65 Conceptual model showing the relationship between threats, drivers, effects and outcomes for subterranean GDEs in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4.3 Inchannel waterholes

Description and background to ecology

Rivers located in northern Australia's wet-dry tropics are subject to highly seasonal rainfall, often resulting in high wet-season flows, and low dry-season flows (Close et al., 2012; Petheram et al., 2008). During the dry season, many rivers cease to flow and can retract to a series of discrete and disconnected waterholes (McJannet et al., 2014; Waltham et al., 2013). In ephemeral rivers and their tributaries, the waterholes that retain water for periods sufficient to outlast dry spells provide vital refuge habitat and resources for both flora and fauna (Sheldon, 2017). Waterholes are also an important social resource, particularly during the dry season, by providing places for recreation as well as providing cultural functions (Centre of Excellence in Natural Resource Management, 2010; McJannet et al., 2014).

Waterholes provide direct habitat for water-dependent species, including fish, sawfish and turtles, and a source of water for other species more broadly within the landscape (McJannet et al., 2014; Waltham et al., 2013). Larger more-stable waterholes that retain water during extended dry periods also often support a vibrant riparian vegetation community (Figure 3-66), and they can be assisted through having more-reliable groundwater (see aquatic GDEs, Section 3.4.2). Riparian vegetation that grows in association with the banks of waterholes can further enhance the habitat value for many species that use the waterhole.

Once river flows recommence and reconnect aquatic habitats in the early wet season, waterholes act as habitat sources for recolonisation of other parts of the catchment (Garcia et al., 2015; Lymburner and Burrows, 2008). Areas with a higher number of persistent waterholes, and often those with a range of different habitat characteristics, are recognised as enhancing biodiversity at regional scales (Arthington et al., 2010; DERM, 2011). Despite their comparatively small contribution to the total area of the catchment, waterholes often provide high habitat value with often disproportionately high biodiversity values.

For the purpose of this Assessment, waterholes are defined as locations within river channels or watercourses that retain water during periods of low or no flow. This definition excludes large lakes and storages, and it pertains to areas of retained water occurring within often-disconnected locations within the river channel, rather than on the floodplain or in the estuary (see Floodplain wetlands, Section 3.4.1). Waterholes can include bodies of water occurring in main channels, braided channels or oxbows, with persistence maintained due to the size or position of the waterhole or in some locations through connection to contributions such as groundwater inflows (also see GDEs in Section 3.4.2).

Figure 3-66 A waterhole with fringing vegetation in Jasper Creek

Photo attribution: CSIRO

Inchannel waterholes in the Victoria catchment

Larger and more persistent inchannel waterholes occur throughout many parts of the Victoria catchment (Figure 3-66). In many locations within the Victoria catchment, groundwater discharge maintains an often significant level of baseflow during periods that would otherwise result in highly reduced flow or cease-to-flow conditions. In areas other than these, however, many tributaries demonstrate the ephemeral flows that are seasonally characteristic of northern

Australian rivers more broadly (Petheram et al., 2008). In these ephemeral reaches, waterholes that persist provide important habitat values. In the Victoria catchment, these biodiversity values are highlighted by providing habitat for species listed under the EPBC Act, including the freshwater sawfish (Vulnerable; Section 3.1.5).

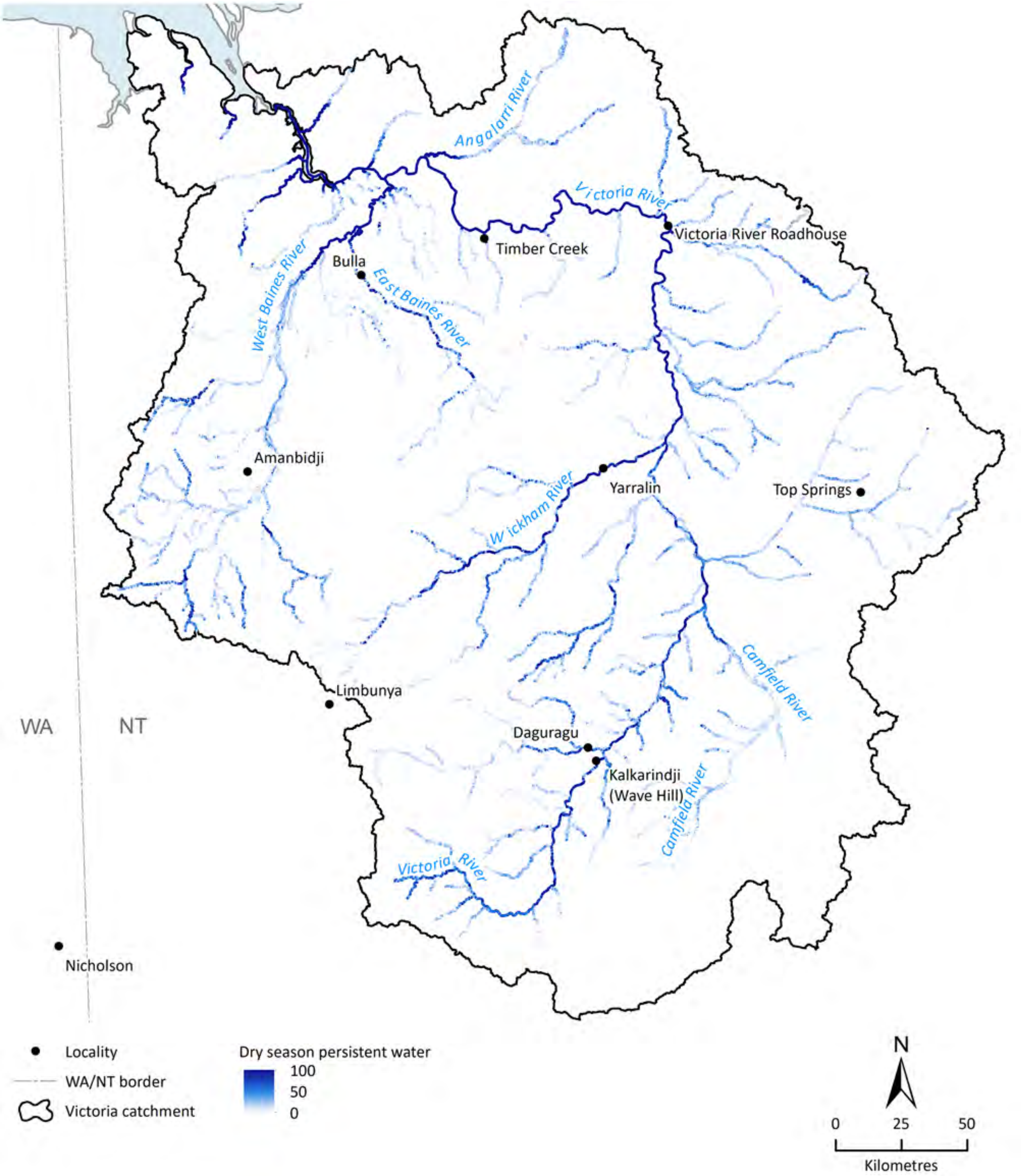

Figure 3-67 Location of persistent inchannel waterholes in the Victoria catchment

Waterholes are mapped at the end of each dry season using Landsat imagery (as described in Sims et al. (2016)). For this, an inchannel mask containing a 500 m buffer from the watercourse is divided into 200 m segments along each watercourse. The percentage of dry seasons containing at least one pixel of water within each 200 m segment is calculated to allow for the fact that a waterhole can vary in shape and location through time. A buffer of 1 km is applied to the derived water persistence product to make it visible at the map scale.

Data source: see Sims et al. (2016)

Flow–ecology relationships for inchannel waterholes

Waterholes are sensitive to changes in low-flow magnitudes, low-flow duration, periods of cease to flow, and timing of first wet-season inflows (Table 3-22). The habitat conditions within waterholes and the persistence of waterholes within the landscape decline where the duration of low-flow periods is extended, where water is removed from the river during low flows, or where water is extracted directly from waterholes. Where dry season flow is reduced, waterholes are increasingly prone to drying out, resulting in a loss of habitat quality and extent, reduced water quality, and changes in competition and food web structure for biota. The timing of a first-flow pulse is important for breaking the dry period, improving water quality and reconnecting habitats. Similarly, conversion of ephemeral systems to perennial systems due to dam or barrier construction will alter the cycle of ephemeral systems and change the natural habitat conditions as low flows and cease-to-flow conditions are important for maintaining ecosystem function, including habitat partitioning and limiting habitat suitability and persistence of non-native species (Yarnell et al., 2015). Infrequent large flows are likely important for maintaining structure within the waterhole. The ecological functions that support inchannel waterholes, and their associated flow requirements, are summarised in Table 3-22.

Table 3-22 Ecological functions supporting inchannel waterholes and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintain waterhole persistence during the dry season	Sufficient inflows to occur during the dry season or shorter duration of low-flow periods for waterhole persistence	Dry-season duration and intensity
Maintain natural ephemerality vs conversion to perennial systems	Occurrence of natural flow regimes including periods of dry conditions	Dry-season duration and intensity
Maintain water quality in waterholes during the dry season	Baseflows into waterholes	Low flows
Waterhole flush and reconnection in the early wet season	Early wet-season first flush. Duration of low-flow periods	Flow timing and magnitude. Seasonality of flows
Waterhole fluvial geomorphology and bed maintenance	Large flood flows to scour and maintain bed structure to occur at suitable frequency	High flows and frequency or reoccurrence of large flood events

Pathways of change for inchannel waterholes

Changes to waterholes resulting from water resource development can occur due to a range of different processes, depending upon the type of water resource development and how it is managed. This assessment considers upstream water capture and storage, water harvesting direct from flows within the channel and climate change (Figure 3-68).

Changes in the flow regime associated with upstream water capture and storage, surface and groundwater extraction, and rainfall and higher evaporation due to climate change have the potential to reduce inflows and influence the natural filling and drying cycles of waterholes (Arthington et al., 2010; McJannet et al., 2014; Waltham et al., 2013). Waterholes are likely to be particularly sensitive to changes in the duration and severity of dry periods and changes in the timing of first flushes and inflows. Other drivers to waterhole persistence and quality can include use of groundwater that results in reduced inflows or faster drawdown of waterholes.

Maintaining the quality of waterhole habitat during periods of low flow is crucial for the local persistence of many of aquatic species (Department of Environment and Resource Management, 2010). Lower dry-season flows resulting in longer periods of low flows due to water resource development threaten to reduce the habitat value of waterholes. This can occur due to loss of waterholes within the landscape or decreases in the condition of the waterholes that remain (Department of Environment and Resource Management, 2010).

Capturing or harvesting water upstream, or extracting water directly from the waterhole, can lead to drawdown or early loss of the waterhole from within the landscape (McJannet et al., 2013). This may result in a localised loss of dependent biota (both aquatic and terrestrial) and the loss or degradation of habitat (McJannet et al., 2014). Where loss of waterholes occurs more frequently within the landscape, it has the potential to result in biodiversity impacts from local to more regional scales across the catchment (James et al., 2013). The number, size and heterogeneity of waterholes is considered important for sustaining biodiversity at larger spatial scales.

Modification of the current duration or timing of low-flow or cease-to-flow periods threatens to change the ecological character of waterholes. During cease-to-flow events, when no surface water enters waterholes, species lose pathways for movement, including longitudinal connectivity along the river channel important to biotic movement. In addition, water quality often deteriorates due to lower exchange along the watercourse. During periods of low inflows, waterhole area is reduced resulting in the loss of important 'slide' and riffle habitat, or potential loss of entire waterholes. The location of individual waterholes within the catchment is an important contributing factor to the duration of the cease-to-flow period, with waterholes in upper catchments more likely to undergo prolonged periods of disconnection under current conditions (Pollino et al., 2018a).

Waterholes persist owing to the hydrological balance within the system that results from the timing and duration of both filling events and drawdown (Close et al., 2012). While loss of waterholes can cause a range of impacts, the alternative of an increase in the persistence of waterholes may also have environmental impacts. Increased water persistence could occur due to the construction of instream barriers, and as a result of persistent or unseasonal releases from upstream storages. Increases in waterhole persistence can alter the natural system to which the flora and fauna are adapted, with possible impacts on habitat structure, water quality, productivity and food web complexity. For example, shifts in the characteristics of waterholes may change predator–prey balances, reduce predator-free habitat for communities of smaller fish species due to loss of many smaller waterholes, or cause conditions to change to those favouring non-native species (McJannet et al., 2013; Yarnell et al., 2015).

The species in each catchment have adapted to the range of conditions that result from the climate and geomorphology of the system. Changes in the range of conditions experienced during the dry or wet seasons, or the transitions between seasons, can alter the species composition of a region. Decreases in flow during the wet season result in loss of connectivity and decreases in flow during the dry season result in loss of critical refuge habitat. Also, homogenisation and loss of the extent of seasonal variation changes the environment to which species have adapted.

Waterholes are typically surrounded by riparian vegetation, which offers shade and structural diversity and acts as an interface between aquatic and terrestrial ecosystems. Changes in waterhole permanence could affect the plants providing this habitat at local and regional scales.

Pest species such as buffalo and pigs, and unrestricted cattle access to waterholes, can damage riparian vegetation and increase sedimentation, turbidity and nutrients within waterholes. Changes in the condition or persistence of waterholes could also provide a competitive advantage to non-native fish species. Invasive species are recognised to often be at an advantage in modified habitats (Bunn and Arthington, 2002). Modified landscapes such as lakes or newly created perennial streams can create habitat for the establishment of pest plant and fish species or be a source for their introduction, whether incidental, accidental or deliberate (Close et al., 2012; Ebner et al., 2020). The ecological outcomes of threatening processes on inchannel waterholes in northern Australia, and their implications for changes to biodiversity and ecosystem function are illustrated in Figure 3-68.

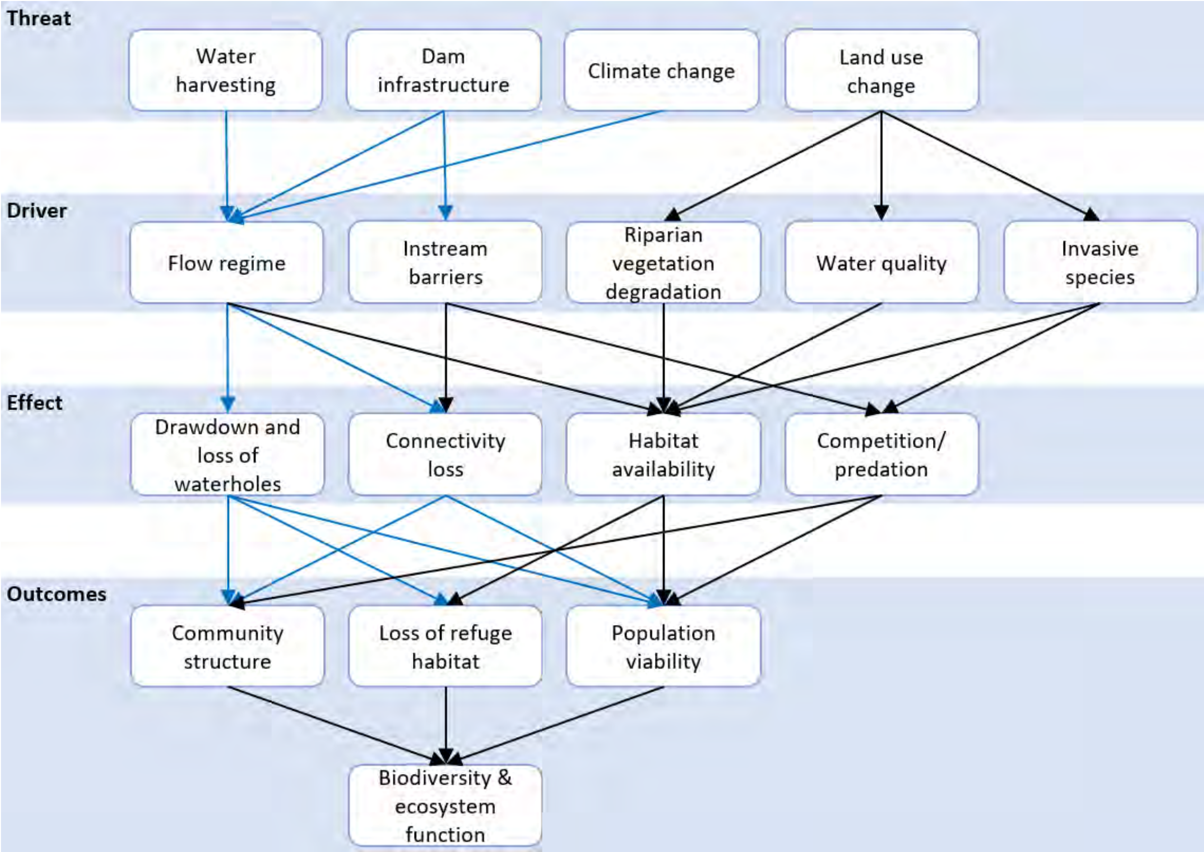

Figure 3-68 Conceptual model showing the relationship between threats, drivers, effects and outcomes for inchannel waterholes in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4.4 Mangroves

Description and background to ecology

Mangroves are a group of woody plant species, ranging from shrub to large tree to forest, that are highly specialised to deal with daily variation in their niche within the intertidal and near-supra-littoral zones along tidal creeks, estuaries and coastlines (Duke et al., 2019; Friess et al., 2020; Layman, 2007). Their occurrence is a result of changes across temporal scales from twice-daily tides to seasonal and annual cycles; mangroves have acclimatised to variable inundation, changing salinity, anoxic sediments, drought and floods, and sea-level change. Mangrove forests provide a complex habitat that offers a home to many marine species ranging including molluscs (McClenachan et al., 2021), crustaceans (Guest et al., 2006; Thimdee et al., 2001), birds (Mohd-Azlan et al., 2012), reptiles (Fukuda and Cuff, 2013) and numerous fish species. During periods of inundation at high tide, fish and crustaceans access mangrove forests for shelter against predation. Fish and crustaceans use mangroves as refugia during larval phases and settle there as benthic juveniles (Meynecke et al., 2010) or access them for food (Layman, 2007; Skilleter et al., 2005). Mangrove forests support many of the species and groups reported as biota assets in this report, particularly commercial species such as banana prawns (Section 3.3.1), barramundi (Section 3.1.1), mud crabs (Section 3.3.3), threadfin (Section 3.1.7) and mullet (Section 3.1.4) (Blaber et al., 1995; Brewer et al., 1995).

Mangrove forests provide a diverse array of ecosystem services, including stabilising shoreline areas from erosion and severe weather events (Zhang et al., 2012), and they play an important role in greenhouse gas emission and carbon sequestration (Lovelock and Reef, 2020; Owers et al., 2022; Rogers et al., 2019). Mangroves continually shed leaves, branches and roots, contributing from approximately 44 to 1022 g carbon per square metre per year from leaves and 912 to 6870 g carbon per square metre per year from roots, though these rates continue to be explored (Robertson, 1986; Robertson and Alongi, 2016). Intertidal crabs living in mangrove forests play an important role in processing and storing mangrove carbon, either through burial in their burrows or uptake directly into production. The decomposition and processing of mangrove material is important also in the cycling of nutrients. If consumed and released, these nutrients support a local food web (Abrantes et al., 2015; Guest et al., 2004), and some of the organic carbon can be transported offshore where it supports fisheries production more broadly (Connolly and Waltham, 2015; Dittmar and Lara, 2001; Lee, 1995).

Mangroves in the Victoria catchment and marine region

Lymburner et al. (2020) mapped the extent of mangroves in Australia using 25 m spatial resolution Landsat 5 (TM, ETM, OLI) sensor data, finding an area of 11,142 ± 57 km^2 (95% confidence interval (CI)) in 2017, which is down slightly from the 2011 extent of 11,388 ± 38 km^2 (95% CI). Most of the change was found to have occurred along the northern Australian coastline and be concentrated in major gulfs and sounds. While coastal urban and industrial development can result in direct loss of coastal wetland ecosystems, including mangroves (Firth et al., 2020; Murray et al., 2022), climate change has also notably caused mangrove loss in northern Australia. The most significant and obvious example was the dieback event between late 2015 and early 2016 along more than 1000 km of coastline in the Gulf of Carpentaria (Duke et al., 2017).

Mangroves in the Victoria catchment are restricted to a narrow fringe immediately along both sides of connecting tidal channels and main estuaries as shown in Figure 3-69. It is estimated that 200 km^2 of mangroves that 533 km^2 of 'supratidal salt flats' (Short, 2020). Mangroves provide habitat and refuge for estuarine fish and crustacean communities in the study area (Kenyon et al., 2004); many fish species that are found in mangroves (Blaber et al., 1995; Brewer et al., 1995). During periods when tidal connection permits access, several fish species will access the mangroves for shelter and food – which is similar to mangrove forests on the east coast of Queensland (Sheaves and Johnston, 2009; Sheaves et al., 2016). While the extent of mangrove forests in this catchment area is relatively small, particularly when compared to the extent of intertidal saltpans, they are still important for coastal fisheries production and provide habitat for local wildlife. In addition, mangroves provide erosion protection, sediment accumulation and carbon sequestration services.

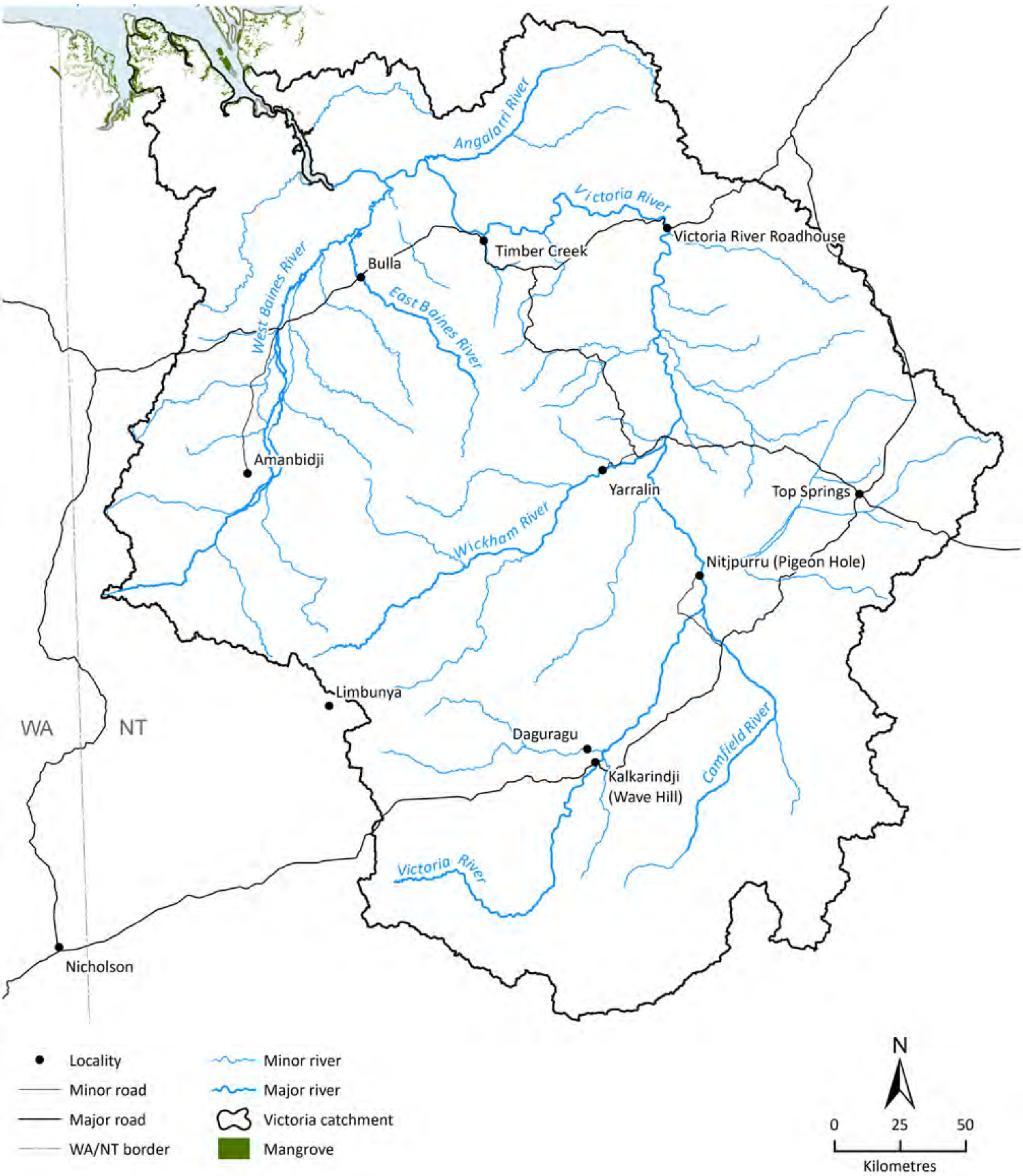

Figure 3-69 Location of mangroves in the Victoria catchment marine region

Data source: Geoscience Australia (2017); NVIS Technical Working Group (2017)

Flow–ecology relationships for mangroves

Large flood events from rivers mobilise catchment sediments and deliver them to the coastal zone. This can be detrimental to some coastal habitats (e.g. seagrass beds can be smothered when sediments inhibit sunlight penetration through the water column). In mangrove forests, while sediment delivered to the coast can also smother mangrove root systems, sediment accumulation is generally considered beneficial as it assists with habitat substrate stability and the accumulation of carbon in sediments (Owers et al., 2022). Asbridge et al. (2016) suggest that Gulf of Carpentaria mangroves have expanded seaward in recent years and that without sediment replenishment these mangrove forests would erode.

The hydrology of mangroves is complex; it is influenced by tidal inundation, rainfall, soil water moisture content, groundwater seepage and evaporation, all of which influence soil salinity that can have profound effects on mangrove growth and survival. Mangroves require access to fresh water, though many species are found at the upper salinity threshold (Robertson and Duke, 1990). A challenge for mangroves is when soil moisture content changes, and they can be greatly affected if soils dry out and the moisture content reduces. The large mangrove dieback in the Gulf of Carpentaria is an example of when soil moisture content was low because of lower sea levels and mangroves were not able to access water (Duke et al., 2017). Mangroves are connected to the sea and estuaries via tidal inundation, which rehydrates soils; the only other time soils become waterlogged is during rainfall or wet-season flow, which recharge soil moisture and groundwater in mangrove forests (Duke et al., 2019). Altered freshwater flow in catchments that previously caused rivers to overtop their banks and spread across coastal floodplains could therefore contribute to mangrove stress and potentially die back. The ecological functions that support mangroves, and their associated flow requirements, are summarised in Table 3-23.

Table 3-23 Ecological functions supporting mangroves and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Maintain natural tidal connection and wet-season freshwater flows (frequency and duration of inundation in mangrove forests) vs conversion to less connection	Tidal flushing patterns and wet-season rainfall inundation keeps soils hydrated	Timing of tidal connection (particularly during the dry season with infrequent freshwater inundation) and freshwater flood connection during the wet season

Pathways of change for mangroves

Several threatening processes can affect mangroves in northern Australia, including river regulation, climate change and land use change. The ecological impacts of dams and river regulation can be numerous; most obviously they can prevent water from flowing onto floodplains by capturing large rainfall events, preventing flood pulses from moving down catchment and reaching dynamic estuaries and near-shore coastal areas. This loss of connectivity to the coastal floodplain areas, including mangrove forests, can result in the reduction or loss of coastal wetland vegetation areas. This was the case in the Gulf of Carpentaria mangrove dieback − while not driven by river regulation, it was a consequence of an unusually lengthy period of severe drought conditions, unprecedented high temperatures and a temporary drop in sea level (Duke et al., 2017). In this extreme event, high temperatures resulted in mangrove dehydration and death – they could not access freshwater sources during critical periods of high summer temperature (Duke et al., 2017).

River regulation can disrupt the natural flow regime. The alteration of the magnitude, frequency, duration, timing and rate of change of flows within a system can affect all aspects of a riverine and floodplain ecosystem (Abrial et al., 2019; Chemagin, 2019; Poff and Zimmerman, 2010). For mangroves these changes can include impacts on the structure, function, sedimentation and biodiversity of mangrove communities. Building dams and other hydrological barriers also affects mangrove forests by choking off sediment loading while increasing nutrient pollution (Godoy et al., 2018). Sedimentation, for example, is critical for the protection of mangrove forests; without sediment supply from river catchments, they would erode (Asbridge et al., 2016).

Coastal wetlands, including mangroves, are particularly vulnerable to climate change (Feller et al., 2017). Climate change impacts may include accelerated sea-level rise, changes in freshwater inputs, and changes to the frequency and intensity of storms and storm surges (Day et al., 2008; Nicholls et al., 1999). Sea-level rise and a decrease of freshwater inputs can lead to the saltwater intrusion of wetlands (Close et al., 2015; White and Kaplan, 2017), which in turn can result in the loss or retreat of mangroves and the conversion of freshwater floodplains to estuarine ecosystems (Duke et al., 2019; Finlayson et al., 1999).

Changes in rainfall, runoff and evapotranspiration patterns (Grieger et al., 2020; Salimi et al., 2021) affecting the hydrology of a system can alter the baseflow and flood patterns (Erwin, 2009). These hydrology changes can also affect the water quality through, for example, increased erosion and changes to temperature (Erwin, 2009). Drought and a lower sea level have been shown to be the cause of mangrove loss in the Gulf of Carpentaria in 2015, and the same event has been reported elsewhere in northern Australia (Duke et al., 2017; Lovelock et al., 2017). Changes to hydrology and temperature can affect the biodiversity ecosystem services that mangroves provide (Dudgeon et al., 2006; Finlayson et al., 2006; Mitsch et al., 2015).

Land use change is a major threat to the extent and fragmentation of mangroves, and there are many examples of mangrove loss in developing areas (Xu et al., 2019). Land use change has contributed to loss of mangroves directly or because of changes in hydrology and flow, causing increased erosion. Changes includes modifying land management practices; changing the intensity or type of agricultural production; increasing vegetation clearing; and increasing mining, urbanisation and industrial development. Evidence of landward expansion of mangroves has been documented (Armitage et al., 2015), but this expansion can only occur where there is sufficient space, and it will be restricted by hard engineering structures or urbanisation that prevents this expansion (Doody, 2004; Leo et al., 2019). The loss of extent or fragmentation of mangroves as a direct result of land use changes or deforestation can reduce carbon sequestration stock (Atwood et al., 2017). In addition, mangroves, when inundated with tidal water, provide critical nursery habitat for local species, including commercial fishery species that would also be affected by the loss of mangroves (Sheaves et al., 2016).

Both the construction of dams and the harvest of river flows via pumped water extraction affect mangrove community stability and replenishment. Reduction in the volume and duration of high-level flows, as well as variability in the seasonality and volume of low-level flows, affects freshwater delivery to the mangrove community and its survival, particularly during the latter dry season when water stress is high (Duke et al., 2017; Plagányi et al., 2023). The impact of water resource development such as dam construction or several levels of pumped water extraction on mangrove communities has been modelled using predicted streamflow data generated under water resource development scenarios (Plagányi et al., 2023). Mangrove biomass was predicted to decline by up to approximately 40% in some river estuaries (predicted average declines of 26, 44 and 28% for the Mitchell, Gilbert and Flinders river systems, respectively, under a ‘high extraction’ scenario). The risk to the mangrove community was assessed as negligible to severe across the four water resource development scenarios (Plagányi et al., 2023).

The ecological outcomes of threatening processes on mangroves in northern Australia, and their implications for changes to biodiversity and ecosystem function, are illustrated in Figure 3-70.

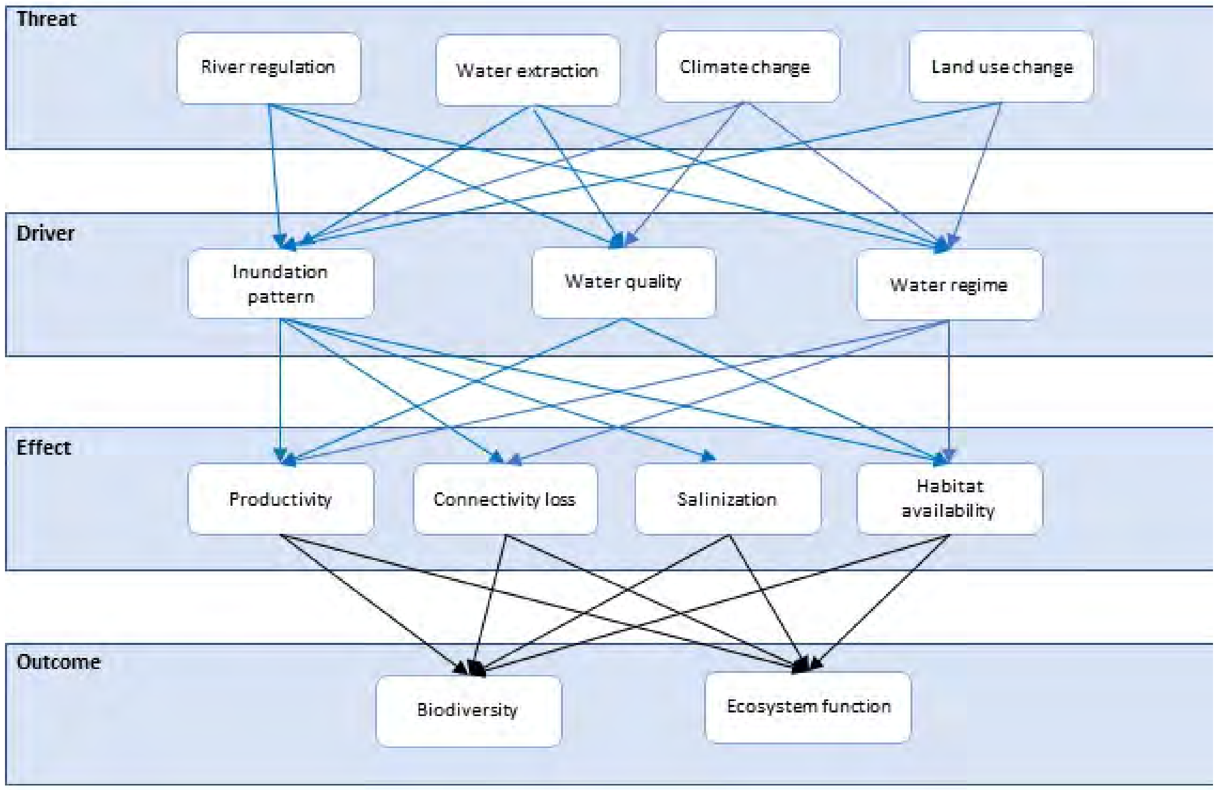

Figure 3-70 Conceptual model showing the relationship between threats, drivers, effects and outcomes for mangroves in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4.5 Saltpans and salt flats

Description and background to ecology

Saltpans and salt flats are intertidal areas that are devoid of marine plants and are located between mangroves and saltmarsh meadows. Saltmarshes (Figure 3-71) occur in the supra-littoral zones that is inundated infrequently by the tide and where subsequent water evaporation leaves behind expanses of minerals and salts (Cotin et al., 2011). Despite their infrequent inundation, saltpans provide habitat for some estuarine fish, such as barramundi (Russell and Garrett, 1983), and shrimps of the genus *Metapenaeus* (Bayliss et al., 2014) during periods when the tide covers these habitats.

Inundation of saltpans mostly occurs during the annual wet season when large tides and rainfall surface runoff ponds as shallow wetted areas within the saltpans and shallow tidal-cut gutters that intersect them. In northern Australia, saltpan sediments are infused with dormant algae that remain inactive in a desiccated state during the dry season (most of the year). However, during overbank inundation from flooded rivers or extensive rainfall, the saltpan soil algae become active and photosynthesise and increase nutrient contribution to the ecosystem (Burford et al., 2016). After several days, active algal growth occurs and carbon, nitrogen and phosphorous compounds are produced. Estimates suggest that saltpans can contribute an extra 0 to 13% of ecosystem primary productivity depending on the extend of saltpan inundation during the wet season (Burford et al., 2016). Saltpans would be most productive during high-level overbank flood flows.

Figure 3-71 Saltpan area in northern Australia, which are generally located between mangrove and saltmarsh areas
Photo attribution: Nathan Waltham

The inundation of saltpans expands the available habitat to hitherto estuarine benthic fish and crustaceans, provided they can tolerate brackish conditions. In northern Australia, coastal saltpans can extend tens to hundreds of square kilometres. They provide habitat for a range of benthic infauna (Dias et al., 2014), which are an important food source for high-order consumers including shorebird species that use saltpans as resting and/or feeding areas during their migration, which can include long flights to Asia (Cotin et al., 2011; Lei et al., 2018; Rocha et al., 2017). The extent of saltpans in Australia is unknown, though they are common and extensive in more arid coastal areas, most notably in northern Australia (Duke et al., 2019). The northern Australian coastline extends for thousands of kilometres and is relatively pristine; low beach profiles backed by extensive saltpans, possibly 5 to 10 km inland, are characteristic of hundreds of kilometres of coastline (Short, 2022). Despite limited tidal exchange, saltpans provide important habitat resources for migratory birds (see Section 3.2.3) that access these areas for feeding and shelter (Lei et al., 2018). In addition, these habitat features also provide erosion and sediment accumulation opportunities in estuaries as well as carbon sequestration services.

Saltpans in the Victoria catchment and marine region

Saltpans in the Victoria River estuary are restricted to an area of tidal inundation on the landward side of mangrove habitats that line the main river channel (Figure 3-72). The spatial data presented illustrates the extent of saltpans in this catchment − for the Victoria River, estimated to 533 km^2 of 'supratidal salt flats' fringed by 200 km^2 of mangroves (Short, 2020). There has been no targeted scientific survey of fish and crustacean communities over the saltpans of the Victoria River floodplain, presumably because they are located so high in the intertidal zone and are only infrequently covered with tidal water. Also, the catchment is remote, which leads to difficulties with access and sampling. Similar to saltpans elsewhere in northern Australia, these saltpans provide important habitat opportunities for many species, including fish and crustaceans during inundation from tides or floods, and migratory birds also use many of them for resting, feeding and shelter.

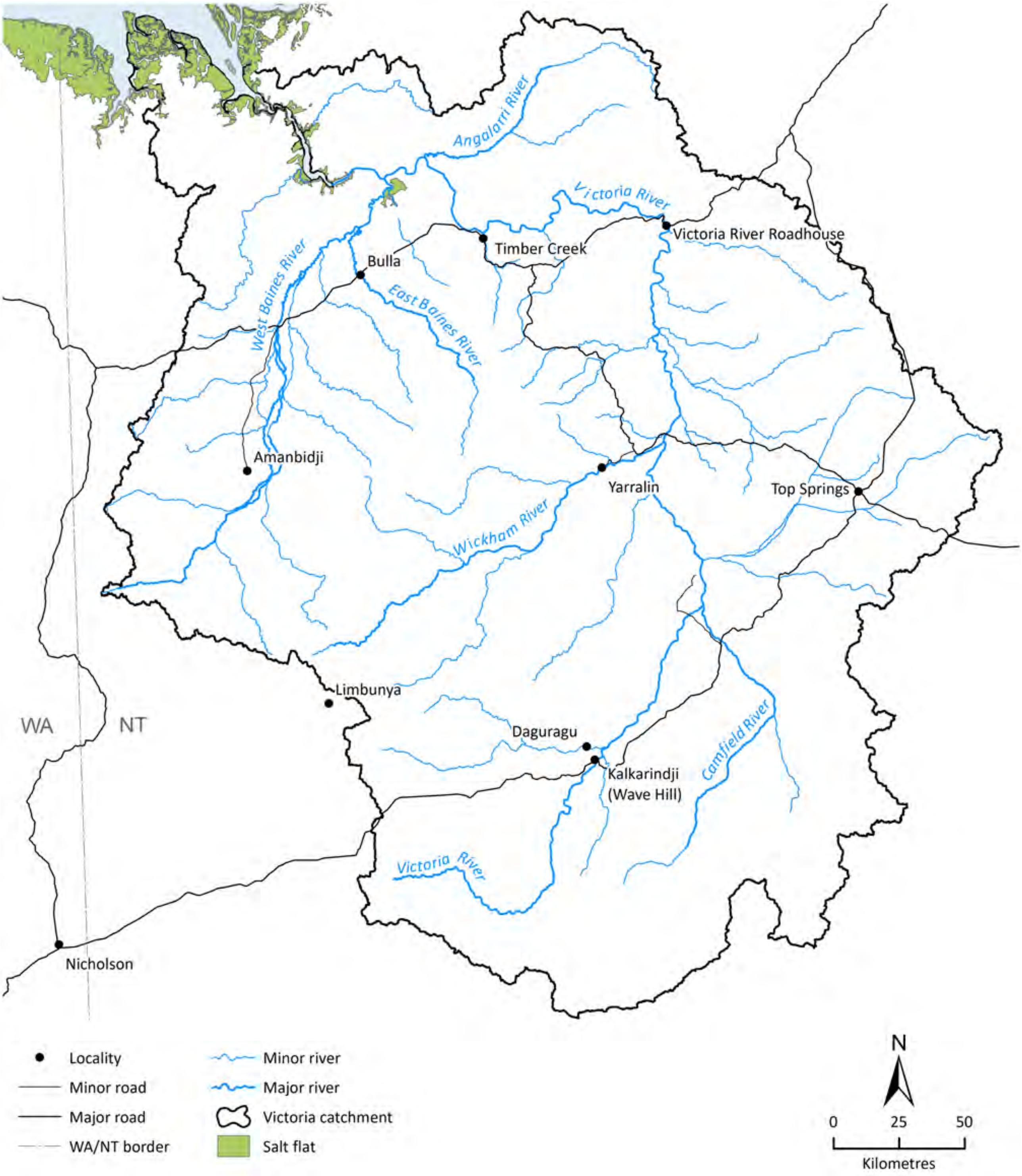

Figure 3-72 Location of salt flats in the Victoria catchment marine region

Data source: Geoscience Australia (2017)

Flow–ecology relationships for saltpans in northern Australia

The hydrology of saltpans is complex. Tidal inundation, rainfall, soil water, groundwater seepage and evaporation all influence soil salinity, which can have profound effects on the services saltpans provide in the seascape. A great challenge to the flora and fauna found on saltpans is change in soil moisture content, particularly if soils dry out and the moisture content reduces, which causes these areas to become hypersaline in the surface soils. Saltpans are connected to sea and estuaries via infrequent tidal inundation, which rehydrates soils. The only other time soils become waterlogged is during rainfall or wet-season flow, which recharges soil moisture and groundwater. Altered freshwater flow in catchments that would otherwise have caused rivers to overtop their banks and spread across coastal floodplains could contribute to wide-scale impacts on the services provided by these habitat resources. The ecological functions that support saltpans, and their associated flow requirements, are summarised in Table 3-24.

Table 3-24 Ecological functions supporting saltpans and their associated flow requirements

ECOLOGICAL FUNCTION	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE
Tidal flushing patterns and wet-season rainfall to keep soils hydrated, maintaining benthic algae production and invertebrate communities, and providing habitat provision and foraging areas for migratory bird and fish species	Maintain natural tidal connection and wet-season freshwater flows (frequency and duration of inundation in saltpans) vs conversion to less connection	Timing of tidal connection (particularly during the dry season with infrequent freshwater inundation) and freshwater flood connection during the wet season

Pathways of change for saltpans

Several threatening processes can affect the saltpans in northern Australia, including river regulation, water extraction, climate change and land use change.

The ecological impacts of dams and river regulation can be numerous; most obviously they can prevent water from flowing onto floodplains by capturing large rainfall events, preventing flood pulses from moving down catchment and reaching dynamic estuaries and near-shore coastal areas. This loss of connectivity to coastal floodplain areas, including saltpans, can result in the reduction or loss of coastal wetland areas (Lei et al., 2018; Velasquez, 1992).

Extraction of surface water generally has less impact on the environment than instream storages, as surface water extraction tends to occur during high-flow events such as floods rather than during low-flow periods (Petheram et al., 2008). As a result, water extraction can lower the peak of a flood, allowing less water for the environment. The reduction in peak flow can decrease the duration and extent of a flood event, and can also prevent overbank flooding altogether (Kingsford, 2000).

Coastal wetlands are particularly vulnerable to climate change (Feller et al., 2017). Climate change impacts include accelerated sea-level rise, a change in freshwater inputs, and changes to the frequency and intensity of storms and storm surges (Day et al., 2008; Nicholls et al., 1999). Sea-level rise and a decrease of freshwater inputs can lead to the saltwater intrusion of wetlands (Close et al., 2015; White and Kaplan, 2017), which in turn can result in the conversion of freshwater floodplains to salt flats (Duke et al., 2019; Finlayson et al., 1999). In the Gulf of Carpentaria, a dieback of mangroves occurred along a large stretch of the coast. This dieback was a response to low rainfall and freshwater runoff from catchments, warmer temperature conditions and a lower sea level than typical during the summer wet season in the Gulf of Carpentaria (Duke et al., 2017). Asbridge et al. (2016) described the replenishment of mangrove habitats due to natural flows in the southern Gulf of Carpentaria, supporting the idea that reduction in flow may reduce sediment loads and set up conditions for erosion of mangrove foreshores and possibly the saltpan habitats behind them.

The loss of saltpan extent or fragmentation as a direct result of land use changes or sea-level rise can reduce carbon sequestration stocks (Atwood et al., 2017). In addition, saltpans when inundated with tidal water provide critical nursery habitat for local species, including commercial fishery species that would also be affected (Sheaves et al., 2016). Invasive species, such as feral pigs, and vehicles driving across saltpans can also change the habitat quality directly through trampling or digging and tyre tracks left behind, which has the potential to alter hydrological connectivity of saltpans with river channels (Trave and Sheaves, 2014; Vulliet et al., 2023; Waltham et al., 2020). Changes in this connectivity could alter soil moisture and leave saltpans degraded and of low-quality habitat for migratory birds (Duke et al., 2019). The ecological outcomes of threatening processes on saltpans in northern Australia, with their implications for changes to biodiversity and ecosystem function, are illustrated in Figure 3-73.

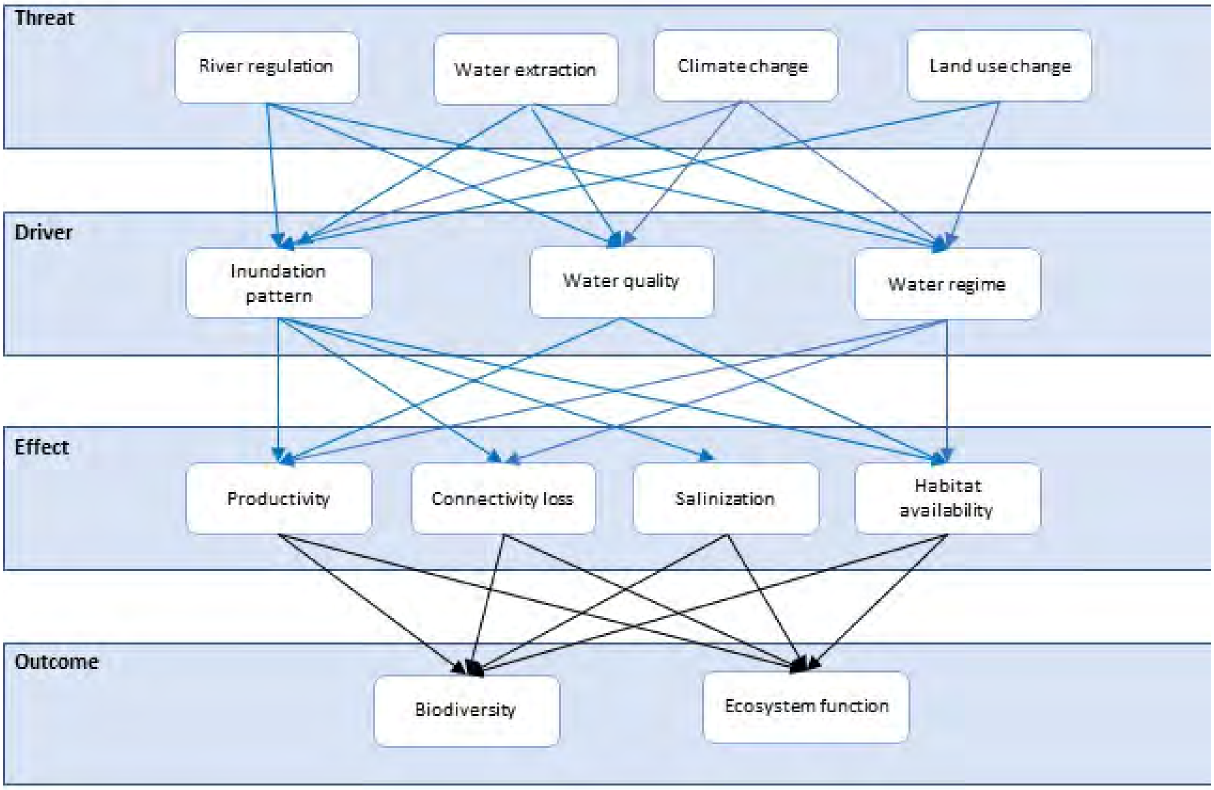

Figure 3-73 Conceptual model showing the relationship between threats, drivers, effects and outcomes for saltpans in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

3.4.6 Surface-water-dependent vegetation

Description and background to ecology

Across much of the northern Australia, terrestrial vegetation survives on water derived from local rainfall that recharges soils during the wet season and can be accessed by the root systems within unsaturated soils throughout the year. Terrestrial vegetation that receives extra water (i.e. in addition to local rainfall), for example, recharge from flood waters (or by accessing shallow groundwater; see Section 3.4.2), often provide a lush green and productive forest ecosystem (high diversity, dense tree cover) within an otherwise drier or more sparsely vegetated savanna environment (e.g. Pettit et al., 2016). This is referred to as surface-water-dependent vegetation. While water availability influences the distribution of savanna versus forest ecosystems across the northern Australia landscape, their distributions are also linked to fire regime, nutrient availability, soil type and herbivory (Murphy and Bowman, 2012). Terrestrial vegetation that receives extra water may contain unique species (e.g. the Carpentarian Rock-rat (*Zyzomys palatalis*; Endangered, EPBC Act; Critically Endangered, IUCN) which is unique to monsoon forest, Crowley (2010)) and provide critical habitat for fauna (e.g. *Melaleuca* forests in the NT support many nationally significant rookeries for waterbirds (Woinarski, 2004)). Such habitats often occur along rivers and floodplains, fringing wetlands and springs or where the depth to groundwater is within reach of the roots.

Vegetation naturally inhabits and thrives in niches in the environment that provide the right combination of water conditions, including surface water depths (during high and low flows), groundwater depth, timing and flood frequency (return interval), and flood duration.

The optimal water regime will vary for different climate conditions (rainfall regime), site conditions (soil type and water availability) and vegetation types. The water regime supports vegetation survival, growth, flowering and fruiting, germination and successful establishment of new saplings for the diversity of ecosystem species, and maintains their functions and services.

Vegetation is unlikely to be able to adapt to changes in water availability outside natural variation. Vegetation has some inbuilt resilience to natural changes in water availability, but prolonged change is likely to result in dieback after some lag period and a shift in ecosystem structure and function (e.g. Mitchell et al., 2016).

Terrestrial vegetation that requires surface water inundation and/or access to groundwater is at risk from water resource development if the natural surface water and groundwater regimes are modified beyond some limit. To anticipate potential impacts of any future water resource development in northern Australia, this section reviews the water regimes that support three terrestrial vegetation types:

- paperbark swamps
- river red gum
- monsoon vine forest.

In northern Australia, these ecosystems provide food and habitat for many species (e.g. for migratory waterbirds, flying foxes, crocodiles and honeyeaters) and play a role in nutrient cycling and providing buffering against erosion.

Paperbark swamps

Paperbark is a term commonly used to describe a range of *Melaleuca* species that have a distinctive papery bark texture. Some paperbark species occur in low-lying areas that are seasonally inundated with fresh water (Department of Environment and Science Queensland, 2013). Many paperbark species co-occur with eucalypt species in riparian and floodplain tree swamps (Department of Environment and Science Queensland, 2013), but here a ‘paperbark swamp’ means the non-tidal coastal and sub-coastal swamp (tree swamp) occurring in the equatorial tropical and subtropical areas of the NT and Queensland (Department of Environment and Science Queensland, 2013) that are dominated by *Melaleuca* species with papery textured bark.

The dominant paperbark swamp species of northern Australia include broad-leaved paperbark (*Melaleuca viridiflora*), weeping paperbark (*M. leucadendra*), silver paperbark (*M. argentea*), blue paperbark (*M. dealbata*) and yellow-barked paperbark (*M. nervosa*) (Department of Environment and Energy, 2017), but may also include *M. acacioides, M. cajuputi, M. citrolens, M. minutifolia* and *M. stenostachya* in the NT and *M. arcana, M. citrolens, M. clarksonii, M. fluviatilis,* *M. foliolosa, M. saligna, M. stenostachya* and *M. tamariscina* in Queensland (based on Department of Environment and Energy, 2017).

The combination of the dominant paperbark swamp species (*M. viridiflora*, *M. leucadendra, M. argentea, M. nervosa* and *M. dealbata*) can flower all year round (Brock, 2022), providing an almost constant source of nectar and pollen for insects, birds and bats (Department of Environment and Science Queensland, 2013). Paperbark swamps provide nesting sites for native birds and flying foxes and are a critical food source for migratory birds (Williams, 2011) and honeyeaters, especially when part of an ecotone (a transition between two ecological communities (Franklin and Noske, 1998)). Fukuda and Cuff (2013) found that about 10% of crocodile nests in the northern coastal and sub-coastal regions of the NT occurred in *Melaleuca* forests and woodlands. It is unknown whether crocodile nesting would continue in *Melaleuca* forests if surface water inundation regimes were altered due to water resource development. Coastal paperbark swamps are hypothesised to provide spawning habitat for gudgeon that move between rivers and floodplains during floods (Department of Environment and Science Queensland, 2013).

Paperbark swamps can be inundated for 3 to 6 months of the year. If they are inundated for longer periods, they may shift towards more grass, sedge and herb-type wetlands (Department of Environment and Science Queensland, 2013). Some species are more tolerant of extended flooding than others, with *M. leucadendra* and *M. cajuputi* occurring in the most flood-prone areas of swamps in northern Australia (Franklin et al., 2007).

Investigations at Howard Springs, NT, showed that paperbark swamps were generally inundated between December and June, and water levels fluctuated between 1 m above ground during the wet season and down to 2.5 m below ground level during the dry season (Cook et al., 1998). There appeared to be sufficient water available to *M. viridiflora* without the need to access shallow groundwater during the monitoring period (based on a water balance study incorporating investigations of evapotranspiration using eddy correlation and sap flow, groundwater dating, soil moisture properties, runoff; Cook et al. (1998)). However, *Melaleuca* species were shown to use groundwater in other parts of northern Australia (e.g. *M. dealbata*; Department of Water and Environmental Regulation (2017), based on water potentials and depth to groundwater data; *M. leucadendra*; Canham et al. (2021), based on stable isotopes of water analyses), indicating that some paperbark swamps are GDEs.

Not much is known about the conditions required for regeneration of paperbark swamps. Major *Melaleuca* germination may be triggered by the timing and extent of wet-season rains (Woinarski, 2004). In general, Franklin et al. (2007) observed very few paperbark seedlings but occasional areas, most often recently burnt, with abundant saplings.

River red gum

River red gum (*Eucalyptus camaldulensis*) commonly line permanent or seasonal rivers and sometimes form forests over floodplains (Costermans, 1981) that are subject to frequent or periodic flooding.

The water requirements of *E. camaldulensis* have not been investigated in northern Australia. However, in the Murray–Darling Basin (MDB), *E. camaldulensis* experiences episodic flooding and drought and it uses more water than is available from rainfall alone (Doody et al., 2015). It can use groundwater with salinities up to a maximum of approximately 30 mS/cm (Overton and Jolly, 2004). Falling groundwater levels have resulted in *E. camaldulensis* dieback when groundwater levels dropped below critical levels or thresholds (12 to 22.6 m below ground surface; Horner et al. (2009); Kath et al. (2014); Reardon-Smith et al. (2011)). The threshold groundwater levels are variable and depend on climate conditions and soil characteristics.

Flooding requirements for maintaining healthy river red gum have been estimated for various floodplain forests and riparian woodlands in the MDB; they range from a flood duration of 2 to 8 months every 1 to 3 years (Rogers and Ralph, 2010) to 2 months duration every 3 to 5 years (Wen et al., 2009). *E. camaldulensis* may require flood to induce seed fall (George, 2004), but excessive flooding can destroy seeds (Rogers and Ralph, 2010). Note that these flooding relationships exist for trees found in the MDB where extensive research has focussed on maintaining this ecosystem type. However, these relationships cannot be directly extrapolated to the different hydrology, soil and climate conditions of northern Australia. Specific water requirements for *E. camaldulensis* and subspecies found in northern Australia are unknown.

Monsoon vine forest

Monsoon vine forest can be found in tropical and subtropical regions of northern Australia, with patches spanning the NT, Queensland and WA. While generally falling under the umbrella term 'rainforest', with its closed canopy and high leaf cover exceeding 70% (Stork et al., 2008), it can be further characterised by canopy height, leaf size, proximity to permanent moist soils and species composition. This forest type is typically found in areas of 600 to 2000 mm mean annual rainfall (Bowman, 2000).

Most monsoon vine forests seem limited to areas with permanent soil moisture, such as creek lines, springs and seeps. They are thought to be remnants of a wetter period during Australia's geological history, when changes in climate, fire regime and water availability restricted their distribution to small pockets (of less than several hectares) across northern Australia (Bowman, 2000). However, the hydrological and geomorphic environments of these ecosystem communities are poorly understood, and while monsoon forest can typically be found in areas that offer fire protection, such as boulder outcrops and areas of high soil moisture, a change in water availability may make monsoon vine forests more prone to fire (Larsen et al., 2016; Russell-Smith, 1991).

While a set definition of what constitutes a monsoon vine forest, vine thicket or rainforest is not wholly agreed upon, the definitions provided by (Webb, 1968; Webb, 1959; 1978) and Russell-Smith (1991) seem to be widely used and are therefore used throughout this report. Furthermore, Russell-Smith (1991) categorised monsoon vine forests into 16 different floristic assemblages or rainforest types; he defined these by where they grew (coastal vs inland), water regime (wet vs dry), rainforest type (forest vs vine thicket) and canopy type and height. This report uses water regime as a focus for selecting monsoon forest types. It focuses on forests that require annual inundation, regular watering through streamflow or are groundwater dependent. These are roughly defined as 'wet', having near constant waterlogging of soils with very little soil drying out, or 'dry', occurring on floodplains or being seasonally flooded and experiencing regular drying out of soils. See Appendix D for a further breakdown of monsoon forest types. Under the EPBC Act, the semi-deciduous vine thickets of WA are considered a Threatened Ecological Community and Endangered (Fisher et al., 2014).

Surface-water-dependent vegetation in the Victoria catchment

The distribution of red gum is not comprehensively mapped, but available data indicate that it occurs along the banks of major rivers in the Victoria catchment (based on data from Atlas of Living Australia (2021); Figure 3-74), even though most of the rivers dry up and cease to flow during the dry season. In the Victoria catchment, paperbark predominantly occur along the West Bains River, Angalarri River and Bullo Creek valleys in the northern, more humid area of the catchment. Paperbark are also common, albeit more restricted, alongside major rivers in the southern, more arid part of the catchment. While Figure 3-74 shows known observed occurrences of paperbark species in the Victoria catchment, it is unconfirmed whether all these occur within swamp habitats. 'Wet' monsoon forest species diagnostic of springs, seasonal flooding and groundwater use (obligate GDEs) occur adjacent to river channels predominantly in the northern reaches of the Victoria catchment, with those species diagnostic of seasonal flooding extending further south (based on ALA data, Atlas of Living Australia (2021)). Significant areas of monsoon forest diagnostic of springs and groundwater use occur in Gregory National Park adjacent to the Victoria River and East Baines River and in the more coastal areas adjacent to Bullo Creek and Napp Springs Creek.

Flow–ecology relationships for surface-water-dependent vegetation

Red gum, paperbark and 'wet' monsoon forest vegetation are sensitive to changes in water availability because they need more water than is available from local rainfall alone to sustain them. Some require periodic inundation by floodwaters and/or access to groundwater to survive, flower, fruit and/or reproduce, as summarised in Table 3-25. The amount, source, timing and frequency of extra water needed by vegetation will vary depending on climate, local soils and vegetation type. The water needs for all vegetation types are not well defined, particularly in northern Australia.

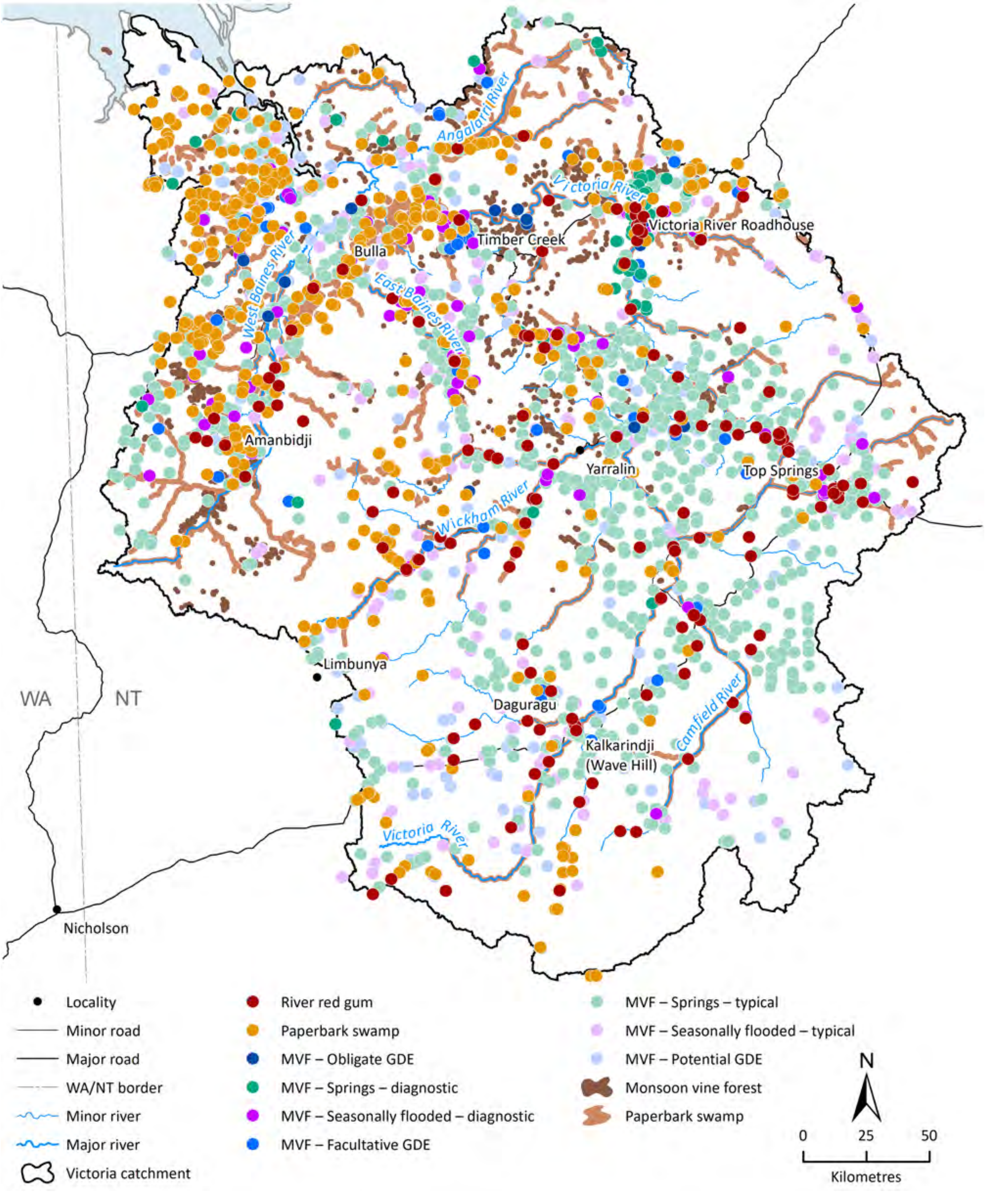

Figure 3-74 Locations of observed selected surface-water-dependent vegetation types in the Victoria catchment

Species within each vegetation type are listed in Appendix D

Datasets: Atlas of Living Australia (2023) Department of Environment Parks and Water Security (2000a)

Table 3-25 Ecological functions supporting surface-water-dependent vegetation and their associated flow requirements

ECOLOGICAL FUNCTION	ECOSYSTEM TYPE	REQUIREMENT OR DEPENDENCY	FLOW COMPONENT OR ATTRIBUTE	SUPPORTING LITERATURE NOTES
Inundation pattern to increase soil moisture	Paperbark swamps *Melaleuca* spp.	Paperbark swamps require annual inundation to maintain health, possibly access to groundwater during the dry season	High-flow conditions <u>(Wetland)</u>	• *M. leucadendra* median period of 56 days of inundation per year, 37 consecutive days, can remain underwater for 6 months[1, 6] • Can handle inundation periods – species dependent[1,6] • *M.nervosa* & *M.viridiflora* can be flooded with a depth of 15–30 cm, length of time not known[8]
	River red gum *E. camaldulensis*	In the MDB, river red gums require inundation every couple of years, this would require overbank flow of rivers – water requirements specific for northern Australia not well studied	High-flow conditions <u>(Riparian Veg)</u>	• Indicative requirements, not specific to northern Australia: • Every 1–3 years for forests, 2–4 years for woodlands, any less than this tree will lose condition[2] • Optimum inundation period lasting 2–8 months[2]
	Monsoon forests wet	Require near-constant water supply, either through surface water inundation through flooding or groundwater supply, depends on location[4,7]		• Relies heavily on soil moisture, probably a groundwater source[4,5]
	Monsoon forests dry	Usually occurs in well-draining soils that dry out annually depending on location, seasonally wet habitat[7]		• Requires perennial water supply, or regular inundation, may or may not have a groundwater source[4]
Water quality & disturbance	Paperbark swamps	Has a high preference for inundation through flooding, some species are associated with tidal floodplains and is fire tolerant	Daily flows <u>(Wetlands)</u> Frequency & duration_ low & high pulses <u>(Wetlands)</u>	• *Melaleuca leucadendra* can tolerate 400 mM NaCl – can also occur in tidal floodplains[3,8]
	River red gum	Has a preference for inundation through flooding every couple of years and is fire tolerant	Daily flows (Riparian Veg)	• Can handle moderately saline groundwater[2], can tolerate 300 mM NaCl[3]
	Monsoon forests wet	Water quality requirements not well understood, may be species dependent within this group	Potential groundwater flows, regular annual low flows	• Very sensitive to disturbances such as erosion from flooding, fire, grazing pressure[6] • Potentially sensitive to water quality changes
	Monsoon forests dry	Water quality requirements not well understood, may be species dependent within this group	Potential groundwater flows, regular annual low flows with a drying out period	• Very sensitive to disturbances such as erosion from flooding, fire, erosion, grazing pressure[6] • Potentially sensitive to water quality changes

References: 1. Franklin and Bowman (2003) includes secondary references, 2. Casanova (2015), 3. Bell (1999), 4. Russell-Smith (1991), 5. Larsen et al. (2016), 6. (Franklin et al., 2007), 7. (Wilson et al., 1996), 8. (Finlayson and Woodroffe, 1996)

Pathways of change for surface-water-dependent vegetation

Changes in the water available to each of these vegetation types (both groundwater and surface water dependent) could affect ecosystem function and persistence of the vegetation type into the future. Some paperbark swamps and most wet monsoon forests require near-constant waterlogging or high levels of inundation to maintain health; these may also use groundwater (Franklin et al., 2007; Larsen et al., 2016). These wet environments create conditions that are essentially fireproof in the near annually burnt fire regime practice of northern Australia (Fisher et al., 2014). Indeed, reductions in water availability can adversely affect these systems through effectively 'drying them out', thus making them more fire prone; this in turn could affect recruitment, community structure and the overall biodiversity of the area. In Litchfield National Park, NT, a study was conducted where a naturally occurring, but retreating alluvial knickpoint (see box below) affected the surface water and groundwater availability for a wet monsoon forest. This retreat dried out the histosol soils (peat-like soils) and caused the wet monsoon forest to retreat, becoming more fire prone and suffering fire damage (Larsen et al., 2016). Wet monsoon forests seem particularly sensitive to disturbances such as erosion, flooding, changes to water regimes and fire. If these disturbances increase in frequency in the wet monsoon forest areas, there is the potential of an ecosystem shift from wet monsoon forest to possibly a paperbark (*Melaleuca*) forest over time, as paperbarks are more resilient to these pressures and have a similar watering requirement (Franklin et al., 2007).

Other threats include grazing pressure through introduced species such as cattle, damage from wallowing species such as the water buffalo and feral pigs, and weeds. All can cause degradation to the environment and can affect community structure, loss of biodiversity and ecosystem function (Russell-Smith and Bowman, 1992).

Alluvial knickpoint explained

An alluvial knickpoint is a geomorphological feature of a river or stream where there is a sudden change in elevation or a sudden step or drop in the river or longitudinal profile, like a waterfall (Fryirs and Brierley, 2012). This can be caused by volcanic uplift, an earthquake, landslide, or in the case of Litchfield National Park, bedrock that is resistant to erosional pressures. A retreating alluvial knickpoint occurs when erosion of the bedrock has sped up and the river is retreating or migrating upstream; this in turn changes the topography of the river, and may influence how groundwater interacts with the vegetation downstream of the knickpoint, as is seen in Larsen et al. (2016).

Water levels, inundation time and the velocities of waterways seem to influence what ecosystem types are present in northern Australia (Figure 3-75). If a location is waterlogged or spring fed, and has little disturbance from fire or floods, then the conditions may better support the wet monsoon vine forest type (Franklin et al., 2007; Larsen et al., 2016). However, in the same environmental conditions but with high levels of disturbances (such as those mentioned above), the location may support certain types of paperbark swamp, or the ecosystem may change as the result of this disturbance (Franklin et al., 2007). Indeed, a paperbark swamp with a high frequency of continual inundation and disturbances could shift to a grassland ecosystem (Department of Environment and Science Queensland, 2013). However, Bren (1992) showed that flooded grasslands are at risk of encroaching river red gum (*E. camaldulensis*) forests if inundation patterns change from yearly, to every couple of years with periods of drying out. Therefore, these ecosystem types are very sensitive to changes in water availability, and a change in watering patterns through dam infrastructure, climate change or water harvesting has the potential to change the current ecosystem and generate an ecosystem shift (Figure 3-75). This conclusion is by no means comprehensive, and a change may not follow the exact pattern described.

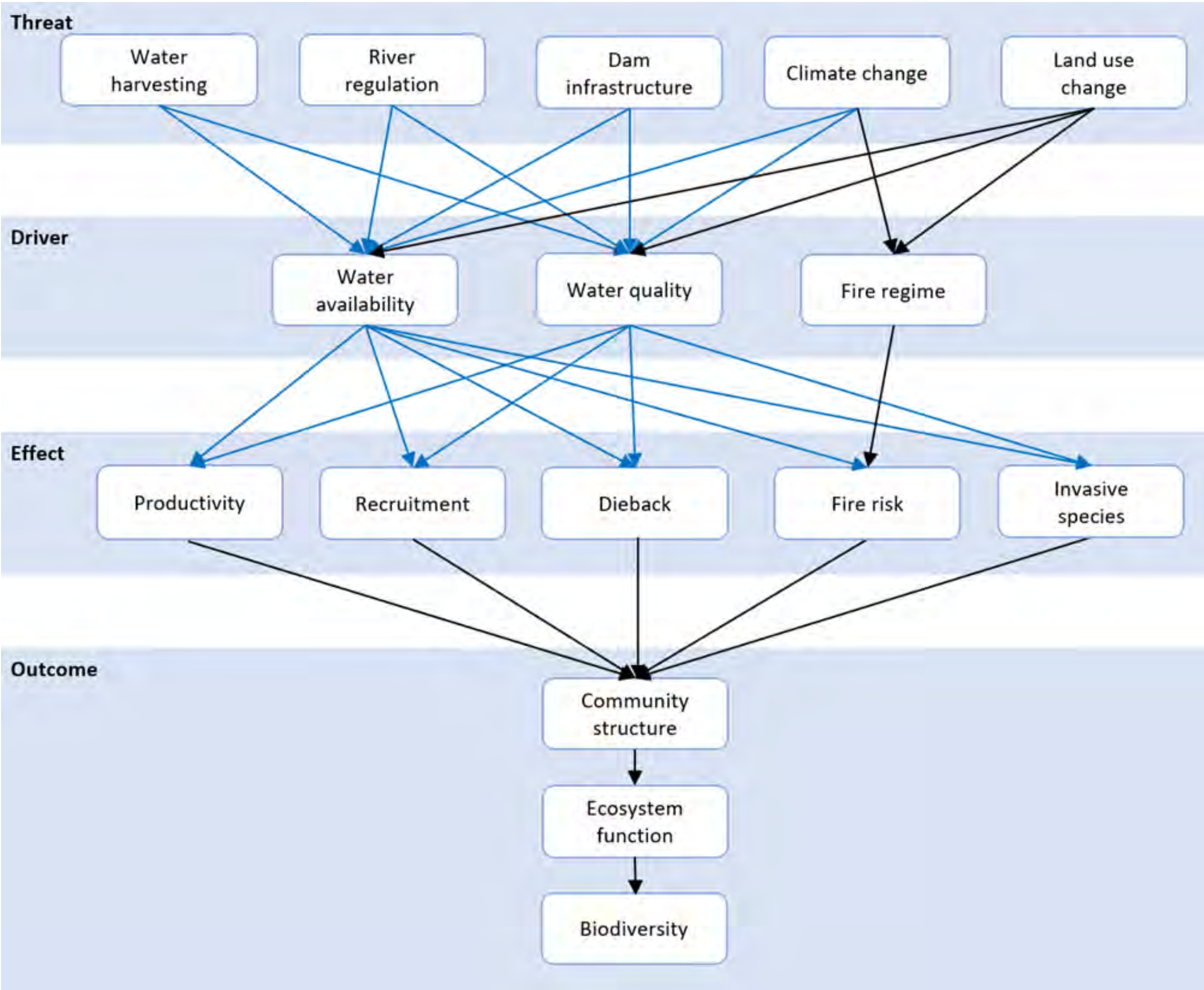

Figure 3-75 Conceptual model showing the relationship between threats, drivers, effects and outcomes for surface-water-dependent vegetation in northern Australia

Blue arrows represent hydrological changes and black arrows represent non-hydrological changes.

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Part II Appendices

Appendix A Species Distribution Model parameters

To estimate modelled distributions for asset species, we first attributed five major classes of predictors to the 954457 of the Australian Hydrological Geofabric (AHGF) 3.2 polygons across Northern Australia that intersect with the AHGF streamlines.

Predictor variables

Predictor variables were attributed to the polygons using the package terra in R

The predictor classes were:

Landuse:
The six highest level classes from the Catchment Scale Land Use map (CLUM)

Conservation and natural environments

Production from Relatively Natural Environments

Production from Dryland Agriculture and Plantations

Production from Irrigated Agriculture and Plantations

Intensive uses

Water

Soils:
14 classes from the Australian Soil Classification Map

Vertosol

Sodosol

Dermosol

Chromosol

Ferrosol

Kurosol

Tenosol

Kandosol

Hydrosol

Podosol

Rudosol

Calcarasol

Organosol

Anthroposol

Geology:
Surface Geology of Australia, summarised into classes that mirror the AHGF NCB 2.1:

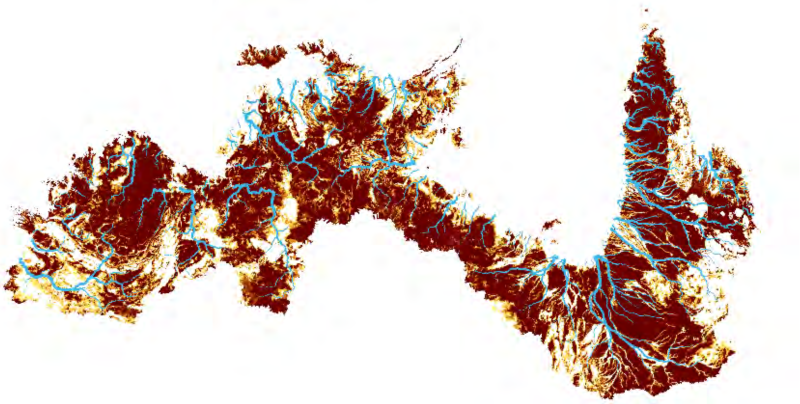
Igneous

Metamorphic

Sedimentary

Mixed

Other



Percent sedimentary rocks in subcatchments

Vegetation
NVIS 6.0, summarised into classes that mirror the AHGF NCB 2.1:

Bare

Forest

Grasslands

Woodlands

Other

Percent woodland in subcatchment

Climate

Variables from the CSIRO 9-second climate surfaces for Australia

Aridity (x3): Mean, min, max

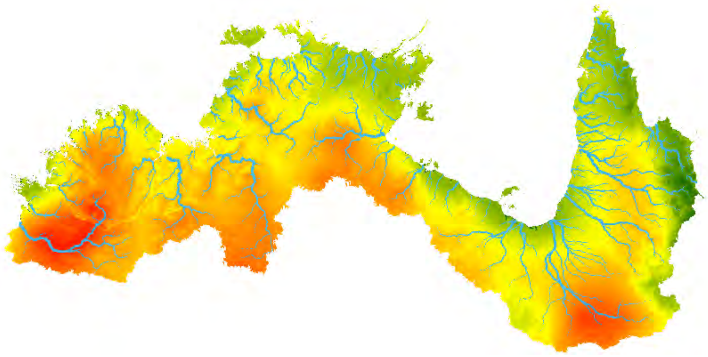
Evaporation (x5): Mean, min, max, as well as actual scaled by MODIS and modelled using water-holding capacity

Precipitation (x5): Annual, min, max, as well as seasonality (equinox and solstice factor ratios)

Minimum Temperature (x3): Annual mean, min, max

Maximum Temperature (x3): Annual mean, min, max

Temperature range (x3): Annual range, diurnal min and max



Average annual evapotranspiration

Ecological data preparation and modelling

Using the package 'galah' in R, we built a pipeline to automatically download ALA datapoints for the entire area of Northern Australia and (see above) and attributed them to the AHGF3.2 polygons using terra.

To reduce autocorrelation and tone down artificial weighting inflation, we reduced the 64 predictor variables to 10 principal components which we then used in the modelling (see Linke et al., 2008)

We then ran three different modelling algorithms:

Random Forests (RF) –an ensemble learning method used for classification and regression. It operates by constructing a multitude of decision trees at training time and outputting the class that is the mode of the classes (classification) or mean prediction (regression) of the individual trees. Random decision forests correct for decision trees' habit of overfitting to their training set. The algorithm combines the simplicity of decision trees with flexibility, making it a robust and accurate model. It handles large data sets efficiently and can manage thousands of input variables without variable deletion.

Generalized Linear Models (GLM) extend linear regression by allowing the linear model to be related to the response variable via a link function and by allowing the magnitude of the variance of each measurement to be a function of its predicted value. GLMs are flexible in handling different types of response variables, like binary, count, or continuous outcomes. They usually avoid overfitting, but result in slightly lower evaluation metrics.

Maxent, short for Maximum Entropy Modeling, is a widely used algorithm in ecology for species distribution modeling. It estimates the probability distribution of a species' occurrence based on environmental constraints, using the principle of maximum entropy. This approach assumes that without additional information, the best distribution is the one that maximizes entropy (i.e., is most uniform) while remaining consistent with the given constraints. Maxent is especially popular for its effectiveness with incomplete data sets and its ability to handle presence-only data, making it ideal for predicting species distributions under various environmental conditions.

We thinned occurrences that were present in the same subcatchment to reduce observation bias and created 1000 pseudo-absences (or if >1000 observations matched the presences with pseudo-absences).

While Maxent and RF have a built in variable weighing algorithm, we ran a best subsets selection procedure for GLMs.

Appendix B Environmental DNA (eDNA) sampling in the Victoria catchment

Collecting environmental DNA (eDNA) is a relatively new scientific approach that can be used to detect and identify species non-invasively, in any given region (Taberlet et al., 2012). Organisms shed genetic material in the form of cells, scales and faeces into their surrounding environment (such as water, soil or air). The process of eDNA analysis involves taking water, soil or air samples from an environment of interest, preserving or processing the samples to avoid the degradation of the DNA in the samples and then, in a laboratory, analysing the DNA present in these samples using techniques such as polymerase chain reaction (PCR) and DNA sequencing (Goldberg, 2016). One of its key advantages is its ability to detect the presence of species even when it is elusive, rare or difficult to observe directly. This makes eDNA particularly valuable for tracking endangered species, monitoring the spread of invasive species, and assessing the overall health of ecosystems. In Australia, eDNA methods are being used to monitor aquatic animals including fish, amphibians and mammals across waterways, estuaries and wetlands. This study uses a vertebrate metabarcoding assay to assess the presence of EPBC-listed species at 13 sites in the Northern Territory.

Methods

Field sampling

Water samples were collected from 13 sites in the Victoria catchment (Apx Figure B-1) between 19 and 21 May 2023. At each site, up to two replicate samples were obtained by passing up to 400 ml of water (mean = 383 ml) through a 1.2 µm filter. Samples were filtrated on-site to reduce DNA degradation during the transport of water samples. The samples were sent to Enviro DNA Pty Ltd for metabarcoding analyses.

Metabarcoding analysis

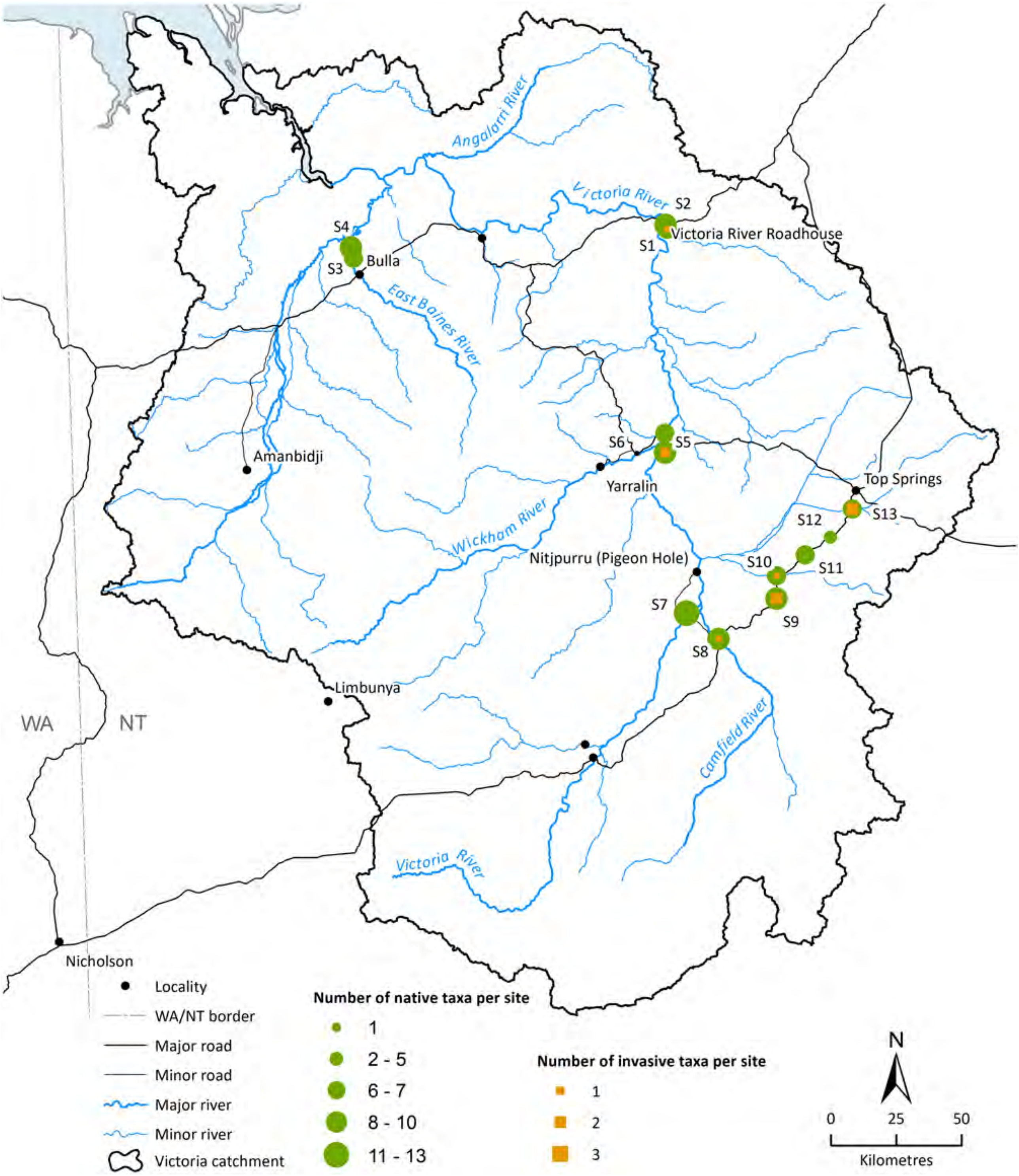
Scientists at Enviro DNA Pty Ltd analysed the samples as follows. DNA was extracted from the filters using a Qiagen PowerSoil Kit that minimises compounds that can inhibit PCR reactions in environmental samples. The species library construction involved two rounds of PCR. On the first-round, gene-specific primers (Vertebrate 12S) were employed to amplify the target region. The second round incorporated sequencing adapters and unique barcodes for each sample−amplicon combination included in the library. During the library construction, negative controls were included. Negative controls consisted of nuclease-free water used in place of DNA during both rounds of PCR. Sequencing was carried out on an Illumina sequencing platform.

Primer sequences truncated and low-frequency reads were removed as part of the quality control process. DNA sequences were then clustered into operational taxonomic units (OTUs) on the basis of sequence similarity. Taxonomic assignment was performed with VSEARCH software (Rognes, 2016). This means that each OTU cluster was assigned a species identity by comparison against a reference sequence database using a 95% threshold. Where a species could not be assigned (i.e. the reference database was deficient and/or taxa were poorly characterised), taxonomic assignments were manually vetted by first obtaining a list of possible species through BLASTN searches against the public repository Genbank (www.ncbi.nlm.nih.gov), followed by elimination of species on the basis of their geographic distributions using information from the Atlas of Living Australia. In cases where an OTU could not be adequately resolved to a single species (e.g. due to shared haplotypes), either a list of multiple species is included, or the OTU is assigned to the lowest taxonomic rank possible without further classification.

Results

A total of 24 samples were analysed by Enviro DNA Pty Ltd from 13 sites in the Victoria catchment using a vertebrate metabarcoding analysis. They presented the results in a spreadsheet showing the taxa detected in each sample and the number of sequence reads for each taxon. The number of sequence reads is not directly interpreted as taxa abundance. Reads may be used to help assign a level of confidence in species detection along with the number of replicates in which the species was detected.

Overall, 37 vertebrate taxa were detected, including four introduced species, three of which − cane toad (*Rhinella marina*), cattle (*Bos taurus*) and wild boar (*Sus scrofa*) − are considered invasive species (Apx Table B-1). The identified taxa included 1 frog, 22 fish, 1 reptile, 7 birds, and 6 mammals. Twelve of these species (*B. taurus*, *Liza ordensis*, *Ambassis agassizii*, *Neosilurus hyrtlii*, *Canis familiaris dingo*, *Ctenotus robustus*, *R. marina*, *Glossogobius munroi*, *Glossamia aprion*, *Selenotoca multifasciata*, *Lates calcarifer* and *Neosilurus ater*) were considered new records for the region, as they had not been previously recorded within 1 km buffers surrounding each sample location, based on ALA data (www.ala.org.au; data extracted from 2010 to the present). Across all replicate samples, the number of taxa at each site ranged from 1 to 15; the number of native taxa per site varied from 1 to 9 (Apx Table B-2). No taxa are reported for Site 6 due to sample contamination in the field.



Apx Figure B-1 Native vertebrate species richness at the 13 sampled sites

Marker size is proportional to detected species richness. Note that mapped native richness only includes taxa resolved at the species level. The map does not show the replicates.

Enviro DNA Pty Ltd were able to resolve about 59% of taxa at the species level (Apx Table B 1). Some taxa were not resolved at the species level. This could be due to a variety of issues, such as: lack of data availability in the reference library for the region or limitations with the gene target region (e.g. 12S, 16S) and/or metabarcoding assays in general. Identifying species using metabarcoding interrogates only a very small subset of the entire genome; therefore, the short marker sequences do not always contain enough genetic variation to definitively assign it to a species.

Apx Table B 1 List of taxa identified in each site and replicate. Invasive species are marked with an asterisk (*) in the common name column

SITES (WITH REPLICATES)	SCIENTIFIC NAME	COMMON NAME
S1_1	*Nematalosa erebi*	Bony bream
S1_1	*Liza ordensis*	River diamond mullet
S1_1	*Toxotes chatareus*	Archer fish
S1_1	*Bubulcus ibis*	Western cattle egret
S1_1	*Bos taurus*	Cattle*
S1_1	*Notamacropus*	Wallaby
S1_2	Terapontidae	Grunters
S1_2	*Hephaestus*	Sooty grunters
S1_2	Terapontidae	Grunters
S1_2	*Neoarius graeffei*	Blue Catfish–forktail
S10_1	*Melanotaenia*	Rainbow fish
S10_1	Terapontidae	Grunters
S10_1	*Nematalosa erebi*	Bony bream
S10_1	*Ambassis agassizii*	Glassfish or perchlet
S10_1	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail
S10_1	*Vanellus*	Lapwings
S10_1	*Bos taurus*	Cattle
S10_2	*Melanotaenia*	Rainbow fish
S10_2	Terapontidae	Grunters
S10_2	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail
S10_2	*Bos taurus*	Cattle*
S11_1	*Melanotaenia*	Rainbow fish
S11_1	Terapontidae	Grunters
S11_1	*Nematalosa erebi*	Bony bream
S11_1	*Toxotes chatareus*	Archer fish
S11_1	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail
S11_1	*Canis familiaris dingo*	Dingo
S12_1	Terapontidae	Grunters
S12_2	Terapontidae	Grunters
S13_1	*Melanotaenia*	Rainbow fish
S13_1	Terapontidae	Grunters
S13_1	*Ctenotus robustus*	Eastern striped skink
S13_2	*Melanotaenia*	Rainbow fish
S13_2	Terapontidae	Grunters
S13_2	*Rhinella marina*	Cane toad*
S13_2	*Vanellus*	Lapwings
S13_2	*Bos taurus*	Cattle*
S13_2	*Sus scrofa*	Wild boar*
S13_2	*Ctenotus robustus*	Eastern striped skink
S2_1	Terapontidae	Grunters
S2_1	*Hephaestus*	Sooty Grunter genus

SITES (WITH REPLICATES)	SCIENTIFIC NAME	COMMON NAME
S2_1	Terapontidae	Grunters
S2_1	*Nematalosa erebi*	Bony bream
S2_1	*Glossogobius munroi*	Monroe’s goby
S2_2	Terapontidae	Grunters
S2_2	*Hephaestus*	Sooty Grunter genus
S2_2	Terapontidae	Grunters
S2_2	*Nematalosa erebi*	Bony bream
S2_2	*Oxyeleotris*	Gudgeon
S2_2	*Glossogobius*	Goby
S2_2	*Glossogobius munroi*	Monroe’s goby
S2_2	*Liza ordensis*	River diamond mullet
S2_2	*Canis familiaris dingo*	Dingo
S3_1	*Hephaestus*	Sooty Grunter genus
S3_1	*Nematalosa erebi*	Bony bream
S3_1	*Glossogobius*	Goby
S3_1	*Liza ordensis*	River diamond mullet
S3_1	*Toxotes chatareus*	Archer fish
S3_1	*Neosilurus hyrtlii*	Hyrtl’s catfish – eeltail
S3_2	*Melanotaenia*	Rainbow fish
S3_2	*Hephaestus*	Sooty grunters
S3_2	*Nematalosa erebi*	Bony bream
S3_2	*Glossamia aprion*	Cardinal fish
S3_2	*Toxotes chatareus*	Archer fish
S4_1	*Strongylura*	Longtom or Needlefish
S4_1	*Nematalosa erebi*	Bony bream
S4_1	*Glossogobius munroi*	Monroe’s goby
S4_1	*Liza ordensis*	River diamond mullet
S4_1	*Toxotes chatareus*	Archer fish
S4_1	*Neoarius graeffei*	Blue Catfish – forktail
S4_1	*Bubulcus ibis*	Western cattle egret
S4_1	*Pteropus*	Fruit bat
S4_1	*Notamacropus agilis*	Agile wallaby
S4_2	*Glossogobius munroi*	Monroe’s goby
S4_2	*Selenotoca multifasciata*	Striped Butterfish or Scat
S4_2	*Neosilurus hyrtlii*	Hyrtl’s catfish – eeltail
S4_2	*Aves*	-
S4_2	*Vanellus*	Lapwings
S4_2	*Notamacropus agilis*	Agile wallaby
S5_1	*Melanotaenia*	Rainbow fish
S5_1	*Strongylura*	Longtom or Needlefish
S5_1	*Nematalosa erebi*	Bony bream
S5_1	*Oxyeleotris*	Gudgeon
S5_1	*Glossogobius*	Goby

SITES (WITH REPLICATES)	SCIENTIFIC NAME	COMMON NAME
S5_1	*Aves*	-
S5_2	*Melanotaenia*	Rainbow fish
S5_2	*Hephaestus*	Sooty Grunter genus
S5_2	Terapontidae	Grunters
S5_2	*Nematalosa erebi*	Bony bream
S5_2	*Liza ordensis*	River diamond mullet
S5_2	*Lates calcarifer*	Barramundi
S5_2	*Toxotes chatareus*	Archer fish
S5_2	*Neoarius graeffei*	Blue Catfish – forktail
S5_2	*Rhinella marina*	Cane toad*
S5_2	*Bos taurus*	Cattle*
S7_1	*Hephaestus*	Sooty Grunter genus
S7_1	Terapontidae	Grunters
S7_1	*Nematalosa erebi*	Bony bream
S7_1	*Oxyeleotris selheimi*	Giant gudgeon
S7_1	*Liza ordensis*	River diamond mullet
S7_1	*Lates calcarifer*	Barramundi
S7_1	*Neoarius graeffei*	Blue Catfish – forktail
S7_1	Plotosidae	-
S7_1	*Neosilurus ater*	Black catfish – eeltail
S7_1	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail
S7_1	Accipitridae	Birds of Prey
S7_1	*Geopelia cuneata*	Diamond dove
S7_1	*Notamacropus*	Wallaby
S7_2	*Melanotaenia*	Rainbow fish
S7_2	Terapontidae	Grunters
S7_2	*Nematalosa erebi*	Bony bream
S7_2	*Oxyeleotris*	Gudgeon
S7_2	*Liza ordensis*	River diamond mullet
S7_2	*Neoarius graeffei*	Blue Catfish – forktail
S7_2	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail
S8_1	*Melanotaenia*	Rainbow fish
S8_1	Terapontidae	Grunters
S8_1	*Toxotes chatareus*	Archer fish
S8_1	*Bos taurus*	Cattle*
S8_1	*Pteropus*	Fruit bat
S8_2	Terapontidae	Grunters
S8_2	*Nematalosa erebi*	Bony bream
S8_2	*Oxyeleotris*	Gudgeon
S8_2	Perciformes	-
S8_2	*Toxotes chatareus*	Archer fish
S8_2	*Neoarius graeffei*	Blue Catfish – forktail
S8_2	*Neosilurus hyrtlii*	Hyrtl's catfish – eeltail

SITES (WITH REPLICATES)	SCIENTIFIC NAME	COMMON NAME
S8_2	*Melopsittacus undulatus*	Budgerigar
S8_2	*Bos taurus*	Cattle*
S9_1	*Melanotaenia*	Rainbow fish
S9_1	Terapontidae	Grunters
S9_1	Plotosidae	-
S9_1	*Rhinella marina*	Cane toad*
S9_1	*Sus scrofa*	Wild boar*
S9_1	*Canis familiaris dingo*	Dingo
S9_2	*Melanotaenia*	Rainbow fish
S9_2	Terapontidae	Grunters
S9_2	Siluriformes	-
S9_2	Plotosidae	-
S9_2	*Rhinella marina*	Cane toad*
S9_2	*Eolophus roseicapilla*	Galah
S9_2	*Bos taurus*	Cattle*
S9_2	*Sus scrofa*	Wild boar*

Apx Table B-2 Number of taxa and number of invasive species detected in each site and replicate

Site 6 does not present any taxa due to sample contamination in the field

SITES (_REPLICATE)	NUMBER OF TAXA	NUMBER OF INVASIVE SPECIES
S1_1	6	1
S1_2	4	-
S2_1	5	-
S2_2	9	-
S3_1	6	-
S3_2	5	-
S4_1	9	-
S4_2	6	-
S5_1	6	-
S5_2	10	2
S6_1	-	-
S7_1	13	-
S7_2	7	-
S8_1	5	1
S8_2	9	1
S9_1	6	2
S9_2	8	3
S10_1	7	1
S10_2	4	1
S11_1	6	-
S12_1	1	-
S12_2	1	-
S13_1	3	-
S13_2	7	3

Appendix C Terrestrial GDE observations in the Victoria catchment

Victoria catchment terrestrial GDE observations in ALA

Apx Table C-1 List of groundwater dependent vegetation species observed in the Victoria catchment

Notes: This is based on a search of the literature and species mapped in ALA (Atlas of Living Australia, 2023) and is not a fully comprehensive list. Any subspecies of these varieties present in the ALA database is included in the mapping (Figure 3-60).

	OBLIGATE GDE	FACULTATIVE GDE OR TYPE OF DEPENDENCY UNCONFIRMED	POTENTIAL GDE
Riparian			
Eucalyptus camaldulensis	✓		
Melaleuca argentea	✓		
Acacia auriculiformis		✓	
Cathormion umbellatum		✓	
Eucalyptus coolabah		✓	
Lophostemon grandiflorus		✓	
Lophostemon lactifluus		✓	
Nauclea orientalis		✓	
Pandanus spiralis		✓	
Syzygium armstrongii		✓	
Carallia brachiata			✓
Corymbia bella			✓
Pandanus aquaticus			✓
Paperbark swamp			
Melaleuca alsophila		✓	
Melaleuca dealbata		✓	
Melaleuca leucadendra		✓	
Melaleuca viridiflora		✓	
Melaleuca bracteata			✓
Melaleuca cajuputi			✓
Melaleuca citrolens			✓
Melaleuca stenostachya			?
Monsoon vine forest			
Tylophora cinerascens		✓	
Abrus precatorius			✓
Atalaya variifolia			✓

	OBLIGATE GDE	FACULTATIVE GDE OR TYPE OF DEPENDENCY UNCONFIRMED	POTENTIAL GDE
Bauhinia cunninghamii or Lysiphyllum cunninghamii			✓
Capparis lasiantha			✓
Carpentaria acuminata			✓
Celtis philippensis			✓
Clerodendrum floribundum var. ovatum			✓
Croton habrophyllus			✓
Diospyros humilis			✓
Dodonaea platyptera			✓
Exocarpos latifolius			✓
Flagellaria indica			✓
Flueggea virosa subsp. *melanthesoides*			✓
Grewia breviflora			✓
Gyrocarpus americanus subsp. *pachyphyllus*			✓
Hypoestes floribunda var. *varia*			✓
Jasminum didymum			✓
Operculina aequisepala			✓
Opilia amentacea			✓
Planchonia careya			✓
Sersalisia sericea			✓
Syzygium nervosum			✓
Terminalia ferdinandiana			✓
Tinospora smilacina			✓
Vincetoxicum cinerascens			✓
Acacia aulacocarpa			✓
Antidesma parvifolium			✓
Calophyllum soulattri			✓
Canarium australianum			✓
Denhamia obscura			✓
Ficus coronulata			✓
Ficus racemosa			✓
Ficus virens			✓
Homalanthus novo-guineensis			✓
Ilex arnhemensis			✓
Lindsaea ensifolia			✓

	OBLIGATE GDE	FACULTATIVE GDE OR TYPE OF DEPENDENCY UNCONFIRMED	POTENTIAL GDE
Litsea glutinosa			✓
Lygodium flexuosum			✓
Lygodium microphyllum			✓
Melastoma affine			✓
Planchonella DNA 47005			✓
Sterculia holtzei			✓
Syzygium angophoroides			✓
Terminalia microcarpa			✓
Vitex glabrata			✓
Xanthostemon eucalyptoides			✓
Other habitats			
Barringtonia acutangular	✓		
Eucalyptus miniata		✓	
Eucalyptus tetrodonta		✓	
Melaleuca nervosa		✓	
Atalaya hemiglauca			✓
Cyperus conicus			✓

Appendix D Surface-water-dependent vegetation observations in the Victoria catchment

Apx Table D-1 Red gum species (including subspecies) observed in northern Australia based on ALA (Atlas of Living Australia, 2021) data within Victoria catchment (tick)

The groundwater-dependent ecosystem (GDE) column denotes whether species are known (tick), or assumed (potential but not specifically investigated, P) to use groundwater.

RED GUM SPECIES	GDE	VICTORIA
Eucalyptus camaldulensis	✓	✓
Eucalyptus camaldulensis subsp. *acuta*	P	
Eucalyptus camaldulensis subsp. *camaldulensis*	P	
Eucalyptus camaldulensis subsp. *obtusa*	P	✓

Apx Table D-2 Paperbark species of northern Australia that occur in seasonally waterlogged habitats based on *Melaleuca* swamp species and *Melaleuca* species habitats (Atlas of Living Australia, 2021) and bark texture

The groundwater-dependent ecosystem (GDE, See Section 3.4.2) column denotes whether species are known (tick), or assumed (potential but not specifically investigated, P), not considered (blank) to use groundwater. * Denotes species for which subspecies exist in the ALA datasets and are included in mapping (Figure 3-74).

PAPERBARK SWAMP SPECIES	GDE	VICTORIA
Melaleuca acacioides		✓
**Melaleuca alsophila*	✓	✓
Melaleuca argentea	✓	✓
Melaleuca cajuputi	P	✓
Melaleuca clarksonii		
Melaleuca citrolens	P	✓
Melaleuca dealbata	✓	✓
**Melaleuca ferruginea*		✓
Melaleuca foliolosa		
Melaleuca fluviatilis	✓	
**Melaleuca lanceolata*	✓	
Melaleuca leucadendra	✓	✓
Melaleuca minutifolia		✓
Melaleuca nervosa	✓	✓
Melaleuca saligna		
Melaleuca stenostachya		✓
Melaleuca tamariscina	✓	

PAPERBARK SWAMP SPECIES	GDE	VICTORIA
**Melaleuca trichostachya*	P	
Melaleuca viridiflora	✓	✓

Apx Table D-3 Monsoon forest species that occur where extra water (in addition to rainfall) is available, for example surface water flows or shallow groundwater

Some species typically occur in wet habitats (drainage lines, seasonally flooded areas or around springs) but may also occur in drier areas and these are termed 'typical'. Some species only occur in wet habitats, and these are termed 'diagnostic'. Lists are based on interpretation of data from Russell-Smith (1991), identification of presence in the Victoria catchment based on ALA data (Atlas of Living Australia, 2021). The groundwater-dependent ecosystem (GDE, See Section 3.4.2) column denotes whether species are known (tick), or assumed (potential but not specifically investigated, P), not considered (blank) to use groundwater. Note: Subspecies of monsoon vine forest species present in ALA datasets are included in mapping.

MONSOON FOREST SPECIES	DRAINAGE LINE	SEASONALLY FLOODED	SPRING	GDE	VICTORIA
Abrus precatorius		typical		P	✓
Abutilon andrewsianum		typical			
Acacia aulacocarpa			typical		✓
Acacia auriculiformis			typical	✓	✓
Acmena hemilampra	typical		typical		
Acmenosperma claviflorum	diagnostic		diagnostic		
glaia sapindina			diagnostic		
Allosyncarpia ternata			diagnostic		
Antidesma parvifolium			typical		✓
Atalaya variifolia				P	✓
Barringtonia acutangula		typical		✓	✓
Bauhinia cunninghamii		typical	typical	P	✓
Blechnum indicum	typical		typical		
Caesalpinia major				P	
Calophyllum sil			diagnostic		
Calophyllum soulattri			typical		✓
Canarium australianum		typical	typical		✓
Capparis lasiantha		typical		P	✓
Capparis sepiaria		typical			✓
Carpentaria acuminata			typical	P	✓
Cayratia maritima		typical			✓
Celtis philippensis		typical		P	✓
Celtis strychnoides					✓
Clerodendrum floribundum				P	✓

MONSOON FOREST SPECIES	DRAINAGE LINE	SEASONALLY FLOODED	SPRING	GDE	VICTORIA
Cordyline terminalis			typical		
Croton habrophyllus				P	✓
Cupaniopsis anacardioides		typical			
Denhamia obscura			typical		✓
Diospyros cordifolia		typical			
Diospyros humilis				P	✓
Dodonaea platyptera				P	✓
Drypetes lasiogyna		typical			
Dysoxylum acutangulum			typical		
Dysoxylum latifolium	typical		typical		
Ehretia saligna		typical			✓
Elaeocarpus culminicola	diagnostic		diagnostic		
Erycibe coccinea			typical		
Euodia elleryana			typical		
Exocarpos latifolius				P	✓
Fagraea racemosa			typical		
Ficus apodogynum			typical		
Ficus benjamina			typical		
Ficus coronulata			typical		✓
Ficus leucotricha			typical		
Ficus opposita		typical			✓
Ficus racemosa		typical	typical		✓
Ficus virens		typical	typical		✓
Flagellaria indica		typical		P	✓
Flueggea virosa subsp. *melanthesoides*				P	✓
Glochidion perakense			typical		
Glycosmis trifoliata		typical			
Gmelina schlechteri			typical		
Grewia breviflora				P	✓
Gymnanthera nitida		typical			
Gyrocarpus americanus subsp. *pachyphyllus*				P	✓
Helicia australasica			typical		
Helicteres rhynchocarpa				P	
Homalanthus novo-guineensis			typical		✓
Horsfieldia australiana			typical		
Hydriastele wendlandiana			typical		
Hypoestes floribunda var. *varia*				P	✓

MONSOON FOREST SPECIES	DRAINAGE LINE	SEASONALLY FLOODED	SPRING	GDE	VICTORIA
Ilex arnhemensis			typical		✓
Jasminum didymum				P	✓
Jasminum molle		typical			✓
Leea indica			typical		
Lindsaea ensifolia			diagnostic		✓
Litsea breviumbellata	diagnostic		typical		
Litsea glutinosa		typical	typical		✓
Livistona benthamii		typical			✓
Lophopetalum arnhemicum			diagnostic		
Lophostemon grandiflorus		typical		✓	✓
Lycopodium cernuum	typical		typical		
Lygodium flexuosum			typical		✓
Lygodium microphyllum			typical		✓
Macaranga involucrata	diagnostic		typical		
Macaranga tanarius			diagnostic		
Maranthus corymbosa			typical		✓
Melaleuca cajuputi				P	✓
Melaleuca leucadendra		typical		✓	✓
Melastoma affine			typical		✓
Melhania oblongifolia		typical			✓
Micromelum minutum		typical			✓
Mimusops elengi					✓
Nauclea orientalis		typical		✓	✓
Nephrolepis biserrata			typical		
Operculina aequisepala				P	✓
Opilia amentacea				P	✓
Passiflora foetida		typical			✓
Piper novae-hollandiae	diagnostic				
Planchonella DNA 47005			typical		?
Planchonia careya				P	✓
Pleomele angustifolius			diagnostic		
Polyalthia australis			typical		
Polyscias australianum	typical		typical		
Rapanea benthamiana			diagnostic		
Rhus taitensis			diagnostic		
Schefflera actinophylla			diagnostic		
Secamone elliptica		typical			✓

MONSOON FOREST SPECIES	DRAINAGE LINE	SEASONALLY FLOODED	SPRING	GDE	VICTORIA
Sersalisia sericea				P	✓
Smilax australis		typical			✓
Sterculia holtzei			typical		
Sterculia quadrifida		typical			✓
Strychnos lucida		typical			✓
Syzygium angophoroides		diagnostic	diagnostic		✓
Syzygium fibrosum			typical		
Syzygium forte			typical		
Syzygium minutuliflorum			typical		
Syzygium nervosum			typical	P	✓
Terminalia ferdinandiana				P	✓
Terminalia microcarpa			typical		✓
Terminalia petiolaris				P	
Terminalia platyphylla		diagnostic			✓
Terminalia subacroptera		typical			
Tinospora smilacina				P	✓
Tylophora cinerascens				✓	✓
Vincetoxicum cinerascens				P	
Vitex glabrata			typical		✓
Xanthostemon eucalyptoides			diagnostic		✓

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