Synthesis of knowledge to support the assessment of impacts of water resource development to ecological assets in northern Australia: asset descriptions

A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments

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This report was prepared for the Department of Infrastructure, Regional Development and Cities. The Northern Australia Water Resource Assessment is an initiative of the Australian Government’s White Paper on Developing Northern Australia and the Agricultural Competitiveness White Paper, the government’s plan for stronger farmers and a stronger economy. Aspects of the Assessment have been undertaken in conjunction with the Northern Territory Government, the Western Australian Government, and the Queensland Government.

The Assessment was guided by three committees:

(i) The Assessment’s Governance Committee: Consolidated Pastoral Company, CSIRO, DAWR, DIIS, DoIRDC, Northern Australia Development Office, Northern Land Council, Office of Northern Australia, Queensland DNRME, Regional Development Australia - Far North Queensland and Torres Strait, Regional Development Australian Northern Alliance, WA DWER

(ii) The Assessment’s Darwin Catchments Steering Committee: CSIRO, Northern Australia Development Office, Northern Land Council, NT DENR, NT DPIR, NT Farmers Association, Power and Water Corporation, Regional Development Australia (NT), NT Cattlemen’s Association

(iii) The Assessment’s Mitchell Catchment Steering Committee: AgForce, Carpentaria Shire, Cook Shire Council, CSIRO, DoIRDC, Kowanyama Shire, Mareeba Shire, Mitchell Watershed Management Group, Northern Gulf Resource Management Group, NPF Industry Pty Ltd, Office of Northern Australia, Queensland DAFF, Queensland DSD, Queensland DEWS, Queensland DNRME, Queensland DES, Regional Development Australia - Far North Queensland and Torres Strait

Note: Following consultation with the Western Australian Government, separate steering committee arrangements were not adopted for the Fitzroy catchment, but operational activities were guided by a wide range of contributors.

This report was reviewed by Elizabeth Bui (CSIRO), Michelle Burford (Griffith University), Barry Hart (Water Science Pty Ltd), Robyn Loomes (WA Department of Water and Environmental Regulation), Alana O’Brien (Qld Department of Agriculture and Fisheries), Julie Robins (Qld Department of Agriculture and Fisheries), Andrew Taylor (CSIRO), Chris Turnadge (CSIRO), and Michael Douglas, Sam Setterfield, Bradley Pusey, Leah Beesley and Caroline Canham (National Environmental Science Program Northern Hub).

Photo: A close up of Neosilurus aters. Source: B Ebner, James Cook University.
Director’s foreword

Sustainable regional development is a priority for the Australian, Western Australian, Northern Territory and Queensland governments. In 2015 the Australian Government released the ‘Our North, Our Future: White Paper on Developing Northern Australia’ and the Agricultural Competitiveness White Paper, both of which highlighted the opportunity for northern Australia’s land and water resources to enable regional development.

Sustainable regional development requires knowledge of the scale, nature, location and distribution of the likely environmental, social and economic opportunities and risks of any proposed development. Especially where resource use is contested, this knowledge informs the consultation and planning that underpins the resource security required to unlock investment.

The Australian Government commissioned CSIRO to complete the Northern Australia Water Resource Assessment (the Assessment). In collaboration with the governments of Western Australia, Northern Territory and Queensland, they respectively identified three priority areas for investigation: the Fitzroy, Darwin and Mitchell catchments.

In response, CSIRO accessed expertise from across Australia to provide data and insight to support consideration of the use of land and water resources for development in each of these regions. While the Assessment focuses mainly on the potential for agriculture and aquaculture, the detailed information provided on land and water resources, their potential uses and the impacts of those uses are relevant to a wider range of development and other interests.

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Preface

The Northern Australia Water Resource Assessment (the Assessment) provides a comprehensive and integrated evaluation of the feasibility, economic viability and sustainability of water and agricultural development in three priority regions shown in Preface Figure 1:

- Fitzroy catchment in Western Australia
- Darwin catchments (Adelaide, Finnis, Mary and Wildman) in the Northern Territory
- Mitchell catchment in Queensland.

For each of the three regions, the Assessment:

- evaluates the soil and water resources
- identifies and evaluates water capture and storage options
- identifies and tests the commercial viability of irrigated agricultural and aquaculture opportunities
- assesses potential environmental, social and economic impacts and risks of water resource and irrigation development.

Preface Figure 1 Map of Australia showing three Assessment areas
Northern Australia defined as that part of Australia north of the Tropic of Capricorn. Murray–Darling Basin and major irrigation areas and large dams (> 500 GL capacity) in Australia shown for context.
While agricultural and aquacultural developments are the primary focus of the Assessment it also considers opportunities for and intersections between other types of water-dependent development. For example, the Assessment explores the nature, scale, location and impacts of developments relating to industrial and urban development and aquaculture, in relevant locations.

The Assessment was designed to inform consideration of development, not to enable any particular development to occur. As such, the Assessment informs – but does not seek to replace – existing planning, regulatory or approval processes. Importantly, the Assessment did not assume a given policy or regulatory environment. As policy and regulations can change, this enables the results to be applied to the widest range of uses for the longest possible time frame.

It was not the intention – 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 (e.g. impacts of irrigation development on terrestrial ecology) are discussed with reference to and in the context of the existing literature.

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 most reliably informs discussion and decision concerning regional development when read as a whole.

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

- Technical reports; that present scientific work at a level of detail sufficient for technical and scientific experts to reproduce the work. Each of the ten activities (outlined below) has one or more corresponding technical reports.

- Catchment reports; that for each catchment synthesise key material from the technical reports, providing well-informed (but not necessarily-scientifically trained) readers with the information required to make decisions about the opportunities, costs and benefits associated with irrigated agriculture and other development options.

- Summary reports; that for each catchment provide a summary and narrative for a general public audience in plain English.

- Factsheets; that for each catchment provide key findings for a general public audience in the shortest possible format.

The Assessment has also developed online information products to enable the reader to better access information that is not readily available in a static form. All of these reports, information tools and data products are available online at http://www.csiro.au/NAWRA. The website provides readers with a communications suite including factsheets, multimedia content, FAQs, reports and links to other related sites, particularly about other research in northern Australia.

Functionally, the Assessment adopted an activities-based approach (reflected in the content and structure of the outputs and products), comprising ten activity groups; each contributes its part to create a cohesive picture of regional development opportunities, costs and benefits. Preface Figure 2 illustrates the high-level links between the ten activities and the general flow of information in the Assessment.
What water and soil resources are available to enable regional development?

Preface Figure 2 Schematic diagram illustrating high-level linkages between the ten activities (blue boxes)
Activity boxes that contain multiple compartments indicate key sub-activities. This report is a technical report. The red oval indicates the primary activity (or activities) that contributed to this report.
Executive summary

Land and water resource developments have the potential to modify rivers, floodplains, estuaries and coastal waters, changing habitats and ecological processes that support native flora and fauna. The Ecology activity of the Northern Australia Water Resource Assessment (the Assessment) will assess the impacts of potential changes in flow on freshwater, estuarine and marine ecosystems due to potential new infrastructure across three focus regions of northern Australia: the Mitchell, Fitzroy and Darwin catchments.

The activity has three major themes:

1. identification and prioritisation of ecological assets
2. asset descriptions, including a conceptual model developed to characterise flow–ecology relationships, the potential impacts of water resource development and other non-flow drivers of change
3. analysis of potential impacts using quantitative and qualitative methods.

A range of functional groups, species, habitats and ecosystem processes were identified as assets. Species were classified as being of recreational, commercial or conservation value. Twenty-eight assets were selected; 13 were classed as freshwater and the remaining 15 were classed as marine.

Each of the 28 assets:

• is water dependent, with that dependency being supported from water sourced within catchments. For species and functional groups, part or all of their life cycle is supported; for habitats, their condition is supported; and for ecological processes, part or all of these are supported
• is distributed across catchments, and is therefore suitable for representing flow responses of ecosystems within catchments
• has a dependency on water that is relatively well understood, such that sufficient knowledge is available to describe their relationships with flow, either conceptually or through more quantitative methods
• is representative of the range of flow requirements and responses in diverse and distributed water-dependent ecosystems
• has significance, being of conservation, recreational or commercial value.

This technical report synthesises the assets within the Assessment study area. The knowledge of these assets underpins the asset analysis technical report. Each asset is accompanied by a description of its significance, its distribution through the Assessment study areas, and an understanding of its relationship to flow. Information that captures other important drivers of change to the asset is also included. The reviews are accompanied by conceptual models and references to publicly available literature.
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Part I  Asset descriptions
1 Background

1.1 Activity overview

Land and water resource developments have the potential to modify rivers, floodplains, estuaries and coastal waters, thereby changing habitats and ecological processes that support native flora and fauna. Northern Australia has diverse and highly valued natural ecosystems that provide commercial, recreational and cultural value, and maintain ecological functions and habitats for plants and animals (Abel and Rolfe, 2009). The ecology of aquatic ecosystems is fundamentally linked to the seasonality of the wet–dry tropical climate that regulates the flow regime (i.e. variability of flow discharge), as well as the landscape it drains (Abel and Rolfe, 2009; Aquatic Ecosystems Task Group, 2012b; Warfe et al., 2011). The natural flow regime and connectivity between aquatic ecosystems is critical for sustaining freshwater and marine biodiversity and natural ecological processes. Many of northern Australia’s aquatic systems are in excellent condition, with the impact of changes in surface and groundwater typically being localised.

The freshwater ecosystems in northern Australia have a rich biodiversity, supporting 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 estuaries of northern Australia also support a rich biodiversity, and are critical in supporting productive fisheries, where increased freshwater inflow to estuaries increases fishery production (Aquatic Ecosystems Task Group, 2012b). Catchment flows also support high-value commercial and recreational marine fisheries, such as the Northern Prawn Fishery, as well as fisheries for barramundi (Lates calcarifer), mud crab (Scylla serrata and S. olivacea) and a suite of other species important to commercial, recreational and Indigenous fisheries, which have high cultural significance. Several species of conservation significance, including dugongs, sea turtles and a variety of sharks and rays – and habitats, such as mangrove forests and seagrass beds – are also dependent on coastal and estuarine systems and the support of catchment flows and nutrient inputs.

The Ecology activity seeks to develop a knowledge base to help assess the potential impacts of changes to flow regimes on freshwater, estuarine and marine ecosystems due to potential new infrastructure across the three focus study areas of northern Australia.

The information in this report underpins the information and analysis in the Northern Australia Water Resource Assessment (the Assessment) Catchment and Case Study reports.
1.2 The assessment ecology activity

The Ecology activity has three major themes:

1. identification and prioritisation of ecological assets
2. asset descriptions, including a conceptual model developed to characterise flow–ecology relationships, the potential impacts of water resource development and other non-flow drivers of change
3. analysis of potential impacts using quantitative and qualitative methods.

1.2.1 IDENTIFICATION AND PRIORITISATION OF ASSETS

Using expert knowledge and species data, a review and prioritisation of assets was undertaken for each study area. Assets considered for the analysis have some level of water dependency, such that they have some level of dependency on ground or surface water flows resulting in either periodic or sustained inundation.

While some assets were specific within a study area, the majority of assets were present across the three study areas. A diversity of assets were selected with a variety of flow requirements to represent the range of potential changes in flow regimes.

For the purposes of the Assessment, assets are defined as:

- being listed threatened, vulnerable or endangered species or communities (Commonwealth, state and territory listings)
- being formally recognised in international agreements (e.g. Ramsar Convention)
- providing vital habitat
- being near natural, rare or unique habitats
- supporting significant biodiversity
- having recreational, commercial or cultural value.

In selecting priority assets, CSIRO wanted to ensure they were:

- representative – to capture a range of flow requirements for biota and ecological processes
- distinctive – to enable a broad representation of water requirements
- describable – with sufficient peer-reviewed evidence available to describe relationships with flow
- significant – ecologically, conservationally and recreationally.

A range of functional groups and species were identified, as were habitats and ecosystem processes. Species were classified as being of recreational, commercial or conservation value. Twenty-eight assets were selected, with 13 classed as freshwater and the remaining 15 classed as marine. Distribution maps were generated for all assets, with the details of data sources and limitation detailed in Appendix B.
1.2.2 ASSET DESCRIPTIONS

The asset descriptions contained within this technical report comprise a description of the asset, its distribution, and for species, its habitat requirements. Conceptual models were developed to illustrate the interactions between flow—ecology relationships for freshwater and marine assets and other potential drivers of change, such as sedimentation, water quality changes, invasive species and climate change. The models were based on expert knowledge and literature, and represent a synthesis of existing knowledge. The conceptual models supported analyses of flow change as a consequence of water harvesting or placement of a dam.

1.2.3 ANALYSIS

Using a range of methods, analyses of change were undertaken using inputs from the inundation and river system models. An analysis of assets was only conducted where there was sufficient knowledge. The analysis method chosen for analysis was reflective of the knowledge and time available. Analysis methods and outputs are presented in Technical Report 2 (Pollino et al., 2018).

1.3 Background to ecology in study areas

Below is a short overview of ecological information for each catchment. More complete descriptions of the study areas can be found in Chapter 3 of the Assessment’s Catchment reports.

1.3.1 MITCHELL

The Mitchell catchment has ecological assets that are of conservation, recreational and commercial value. The catchment has three wetlands of national significance with significant biodiversity value. These are the:

- Mitchell River fan aggregation
- South-east Karumba Plain aggregation
- Spring Tower complex.

The Mitchell River fan aggregation comprises deeply incised stream lines with numerous permanent waterholes and floodplains, and provides habitat for a wide range of waterbirds (Environment Australia, 2001). The south-east Karumba Plain aggregation contains varied habitats, including tidal flats, stream channels, and ephemeral and permanent wetlands. This supports important waterbird breeding habitat, including the second-largest summer population of wader birds in Australia, and is recognised as having high wilderness value (Environment Australia, 2001). The Spring Tower complex contains spring-fed freshwater cave systems and is recognised as a good example of a karst wetland; these have restricted
distribution in Australia. The Spring Tower complex contains relict fauna and flora, including vine thickets and blind amphipods (Environment Australia, 2001).

The Mitchell catchment contains an extensive network of wetlands and waterholes, which support fish, invertebrates, crocodiles, frogs, turtles and waterbirds. The extensive floodplain of the catchment provides an important source of nutrients for a sub-set of fish species, including those feeding on benthic algae (Hunt et al., 2012). Fish, crayfish, prawns and shrimps access carbon from the floodplain as a source of energy. In turn, these animals are an important food source for large predators, particularly in waterholes during the dry season (Hunt et al., 2012). Collectively, studies in the Mitchell catchment have demonstrated the importance of connectivity between the river and floodplains for both large and small fishes and higher-level predators, including crocodiles (Hunt et al., 2012; Jardine et al., 2017).

The Mitchell catchment has the second-highest fish species richness nationally (Pusey, 2011). This is partially attributable to its extensive and diverse inland freshwater permanent aquatic habitats, which are largely undisturbed (Pusey, 2011). Freshwater fishes perform central ecological functions and structure ecological communities within floodplain river ecosystems (Jardine et al., 2012b). A sub-set of these fishes are large-bodied, diadromous species (species that migrate between freshwater and seawater), which provide the basis for recreational and subsistence fisheries and are of cultural significance (Close et al., 2014; Ebner et al., 2016). Species such as barramundi, threadfin salmon (family Polynemidae) and mud crab are particularly important to commercial and cultural fisheries, and support fishing tourism in the south-eastern Gulf of Carpentaria (Bayliss et al., 2014). The freshwater sawfish (Pristis pristis) is of conservation significance in the catchment. Other significant fauna in the estuarine and coastal waters of the Mitchell catchment include dugongs (Dugong dugon), sea snakes, speartooth sharks (Glyphis glyphis), sea turtles (Cheloniidae spp) and sawfish. White banana prawns (Fenneropenaeus merguiensis) are of considerable value to the Northern Prawn Fishery (approximately $106.8 million per year in 2015 for three prawn species) and are highly dependent on river flow.

Some areas of the Mitchell catchment have undergone land use changes, including intensive cropping and horticulture in the upper catchment. The upper catchment also has extensive cattle grazing, which has affected vegetation and led to some areas of river bank erosion and the establishment of weeds (Tait et al., 2015). Past mining has also disturbed sediment loads and potentially changed local water quality. Exotic fish have been found in the catchment, with the greatest concern being the spotted tilapia (Tilapia mariae).
1.3.2 FITZROY

The Fitzroy catchment has ecological assets that are of conservation, recreational and commercial value. The catchment has three areas with wetlands of national significance:

- Camballin Floodplain
- Geikie Gorge
- Gladstone Lake (Environment Australia, 2001; SKM, 2009).

Camballin Floodplain supports internationally significant waterbird populations (Department of Water, 2009). Geikie Gorge wetland is located within Geikie Gorge National Park. Mornington Sanctuary, which is adjacent to Gladstone Lake, is found in the upper catchment, and has a high biodiversity, supporting 13 threatened animal species and 10 rare or threatened flora species (Centre of Excellence in Natural Resource Management, 2010).

Important habitat types within the catchment include riparian rainforest, wetland systems, mound springs, mudflats and permanent inchannel waterholes (Centre of Excellence in Natural Resource Management, 2010; SKM, 2009). While the Fitzroy River channel ceases to flow in the dry season, some of the permanent inchannel waterholes are hydrologically connected to groundwater during the dry season, and are considered the only persistent water source for terrestrial and aquatic fauna, including waterbirds (Vogwill, 2015), fish, invertebrates, crocodiles, frogs and turtles. The floodplain of the Fitzroy River extends over much of its length (Pusey and Kath, 2015) and provides an important source of nutrients for a sub-set of fish species, including those feeding on benthic algae (Douglas et al., 2005). Fish, crayfish, prawns and shrimps access carbon from the floodplain as a source of energy. In turn, these animals are an important food source for large predators, particularly in waterholes during the dry season (Douglas et al., 2005).

The Fitzroy catchment has a high aquatic biodiversity (Vogwill, 2015). Fauna of interest include barramundi, freshwater sawfish, dwarf sawfish (*Pristis clavata*) and the saltwater crocodile (*Crocodylus porosus*); these species are of regional or national interest (Bartolo, 2006; Centre of Excellence in Natural Resource Management, 2010). The Fitzroy catchment includes 35 of 43 fish species known from the Kimberley, with 18 of these being endemic (Department of Water, 2009) – one of the highest examples of fish species endemism in northern Australia (Hermoso et al., 2011). Local endemic species of evolutionary and conservation significance to the Fitzroy include Greenway’s grunter (*Hannia greenwayi*), Barnett River gudgeon (*Hypseleotris kimberleyensis*) and freshwater sawfish (Morgan et al., 2011b; Morgan et al., 2004).

Freshwater fishes perform central ecological functions and structure ecological communities within floodplain river ecosystems (Jardine et al., 2012b). In the Fitzroy catchment there are several species of large-bodied diadromous species (species that migrate between freshwater and seawater) that provide the basis for recreational and subsistence fisheries, and are of cultural significance (Close et al., 2014; Ebner et al., 2016). Species such as barramundi, threadfin and mud crab are particularly important to commercial and cultural fisheries, and support recreational fishing (Bayliss et al., 2014). The Fitzroy features the
following Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act)-listed endangered fish species: freshwater sawfish, dwarf sawfish and northern river shark (Glyphis garricki). King Sound and the adjacent Fitzroy catchment are the only known nursery areas for freshwater sawfish in the Kimberley (Thorburn and Morgan, 2005).

The Fitzroy catchment end-of-system, King Sound, is high-value estuarine–coastal habitat. It is a large, semi-enclosed water body that is fringed by broad tidal mudflats. The tidal mudflats and mangroves provide nursery habitat for a wide variety of fish (Loneragan et al., 1997a) and crustaceans, which are important prey for inshore dolphins. Australian snubfin dolphins (Orcaella heinsohni) and the Indo–Pacific humpback dolphin (Sousa chinensis) use the tidal rivers, while bottlenose dolphins (Tursiops truncatus) are found in deeper waters.

The Fitzroy River catchment’s land use is dominated by rangeland grazing, with limited areas of mining development and irrigated agriculture (Pusey and Kath, 2015). Assessments of land condition in the catchment show widespread declines in land condition and erosion as a consequence of grazing (Kubicki et al., 1979). River, wetland and riparian habitats are affected by grazing, with cattle congregating in and around sources of water and in the shade of riparian vegetation (Pusey and Kennard, 2009). This causes a decline in local physical habitat and water quality, and potentially downstream habitat through introduction of sediment and nutrients. Grazing pressures have degraded the riparian zone, leading to a decline in the endangered purple-crowned fairy wren (Malurus coronatus) population. This species is now locally extinct in the lower reaches of the Fitzroy River, and the largest remaining population occurs in the upper reaches of the catchment (Skroblin and Legge, 2012). The mountain white gum (Eucalyptus mooreana; also known as Moores gum), which is listed as vulnerable under the EPBC Act, has also largely declined in the catchment as a consequence of grazing. While information is limited on the extent of weeds, they are well established in some areas (Centre of Excellence in Natural Resource Management, 2010). Little research has been conducted on the marine science of the King Sound (Pusey and Kath, 2015).

1.3.3 DARWIN

The Darwin catchments have ecological assets that are of conservation, recreational and commercial value. It includes four other catchments:

- Finniss
- Adelaide
- Mary River
- Wildman River.

The catchments have numerous and extensive wetlands and floodplain systems, including five wetlands of national significance, from a total of 33 nationally significant wetlands in the Northern Territory (Environment Australia, 2001). These four catchments support high levels of biodiversity. The richness of this biodiversity is attributed to the integrity, extent
and heterogeneity of the wetland habitats throughout the study area (Department of Land Resource Management, 2015; Environment Australia, 2001).

The Finniss catchment has two wetlands of conservation significance. The Finniss River Coastal floodplain supports breeding for waterbirds, magpie geese (*Anseranas semipalmata*) and saltwater crocodiles, and is habitat for threatened species such as great knot (*Calidris tenuirostris*), eastern curlew (*Numenius madagascariensis*) and marine turtles (Northern Territory Government, 2009). Fog Bay supports migratory shorebirds and marine turtles, many of which are threatened (Northern Territory Government, 2009). The upper Finniss catchment has a legacy mining site, the Rum Jungle Mine.

Darwin Harbour, which has extensive and diverse mangrove areas, supports specialised bird species. There are also extensive mudflat areas and the harbour supports a diverse range of marine species, including dugongs, dolphins, marine turtles and a large variety of fish. The harbour is habitat for threatened species, such as curlew sandpiper (*Calidris ferruginea*), red goshawk (*Erythrotriorchis radiatus*), flatback turtle (*Natator depressus*) and floodplain monitor (*Varanus panoptes*) (Northern Territory Government, 2009).

The Adelaide catchment has the Adelaide coastal river floodplain, which is a large, seasonally inundated freshwater floodplain with a mix of wetland, grass and sedge and open woodland communities, and pockets of monsoon forest. This provides breeding habitat for waterbirds, including magpie geese. Mangroves are found in the lower reaches of the Adelaide River, as are migratory shorebirds using wetlands and mudflats. The floodplain is habitat for threatened species, including the northern quoll (*Dasyurus hallucatus*), plains death adder (*Acanthophis hawkei*) and Gouldian finch (*Erythrura gouldiae*) (Northern Territory Government, 2009).

The Mary River has the Mary River coastal floodplain, which is a mix of dry and wet habitats that supports waterbirds and magpie geese, and is a breeding area for the barramundi. This floodplain is habitat for threatened species, including the brush-tailed rabbit-rat (*Conilurus penicillatus*) and the bar-tailed godwit (*Limosa lapponica*). There are barrages in place to prevent saltwater intrusion (Northern Territory Government, 2009) into the freshwater wetland systems.

The Wildman River catchment neighbours and incorporates part of the Kakadu National Park, which contains a diversity of wetlands, habitats and species. It is a Ramsar-listed wetland and provides habitat for more than 75 threatened species.

The Adelaide River Floodplain and Mary Floodplain systems are recognised as an important breeding area for waterbirds and crocodiles, and are among the most important breeding sites for magpie goose in Australia (SKM, 2009). About 27% of the study area is protected within national parks and reserve systems (Department of Land Resource Management, 2015). Protected areas within the study area include Litchfield National Park (Finniss River), Djukbinj National Park (Adelaide River) and Mary River National Park (Mary River).
Coastal floodplains, mangroves and mudflats are characteristic at the end of catchments (Territory Natural Resource Management, 2016). The Northern Territory’s mangroves are considered to be unusually diverse and extensive (Woinarski, 2004). Important inland habitats through the catchments include extensive floodplains, wetlands and inchannel waterholes, riparian zones and groundwater-dependent monsoon forests (Warfe et al., 2010). The extensive floodplain wetlands support aquatic fauna, including waterbirds, fish invertebrates, crocodiles, frogs and turtles. Seasonal flooding of the catchments sustains off-river wetlands and leads to a boom in productivity, while the groundwater baseflow maintains permanent waterholes as important refuge habitat (Warfe et al., 2011). Many permanent waterholes through the river systems are also in part replenished by groundwater, with the waterholes creating refugia in the dry season. Floodplains provide an important source of nutrients for a sub-set of fish species, including those feeding on benthic algae (Douglas et al., 2005). Fish, crayfish, prawns and shrimps access carbon from the floodplain as a source of energy. In turn, these animals are an important food source for large predators, particularly in waterholes during the dry season (Douglas et al., 2005).

The Northern Territory has a relatively simple patterning of vegetation over large areas, but with localised variation (Woinarski et al., 2005). Examples of this variation are the highly diverse monsoon forest patches, comprising only a small area (Woinarski et al., 2005). Spring-fed monsoon forests of high conservation value in the Wildman catchment are vulnerable to increased use of groundwater resources. Riparian zones are another variant and are only represented as narrow strips of vegetation that have high biodiversity and productivity. Riparian zones are rich in birdlife (Woinarski et al., 2000) and are sensitive to changes in both surface water and groundwater regimes (Pusey and Kennard, 2009). Melaleuca forests and woodlands occur in seasonally inundated areas, especially on the floodplains of the lower reaches of major river systems (Woinarski, 2004).

The Darwin catchments have a high aquatic fish biodiversity. Freshwater fishes perform central ecological functions and structure ecological communities within floodplain river ecosystems (Jardine et al., 2012b). There are several species of large-bodied diadromous species (species that migrate between freshwater and seawater), which provide the basis for recreational and subsistence fisheries and are of cultural significance (Close et al., 2014; Ebner et al., 2016). In the absence of definitive local studies of fish ecology in the study area, the movement and migration ecology of the fish fauna is perhaps best inferred from the Alligator rivers study area. This research revealed considerable seasonal migration between lowland floodplain and main channel habitats (Bishop et al., 1990). As shown by fish assemblage studies in all northern Australian river systems since, wet-season-related connectivity of aquatic habitats is an important driver of fish assemblage dynamics.

Species such as barramundi, threadfins and mud crab are particularly important to commercial and cultural fisheries, and support recreational fishing (Bayliss et al., 2014). The Adelaide River features the speartooth shark, a species listed as vulnerable under the EPBC Act (Burrows, 2008). Significant fauna in the Darwin catchments include saltwater and freshwater crocodiles (*Crocodylus johnstoni*), the dwarf sawfish, the freshwater sawfish, the northern river shark and the pig-nosed turtle (*Carettochelys insculpta*). The ecology of many
of these species is highly dependent on the quality and quantity of water resources, and on the maintenance of habitat complexity. Many species, such as Australian snubfin dolphins, barramundi, sawfish and mud crabs, have life histories that span freshwater through estuarine to marine environments, and are valuable to the commercial, recreational and Indigenous fishery sectors (Bayliss et al., 2014).

The top end of the Northern Territory has the world’s largest intact savanna, with a diverse array of plants, mammals, birds, reptiles and amphibians (Territory Natural Resource Management, 2016). The Northern Territory has about 20% of Australia’s eucalypt forests and woodlands, with low clearing rates (Woinarski, 2004), although there are some areas of clearing in the Darwin catchments (Ziembicki et al., 2014). Dry-season fires are a feature of tropical savannas of northern Australia, with the northern half of the Northern Territory areas being burnt annually or biennially (Williams et al., 1999). Low-frequency, high-intensity fires cause high mortality of trees, unlike low-intensity, high-frequency burning fires (Williams et al., 1999). The riparian zone is particularly sensitive to fire, far more so than the surrounding savanna (Douglas et al., 2015).

The top end of the Northern Territory is generally not subjected to extensive urban or agricultural expansion (Territory Natural Resource Management, 2016). There are small areas of irrigation through the area, areas of extensive cattle grazing, horticulture and former rice growing areas on the Adelaide River. The Fogg and Harrison dams on the Adelaide River are infrastructure now managed for conservation purposes. The dams provide dry season refuges for birds. Invasive weeds, such as olive hymenachne (Hymenachne amplexicaulis) and water hyacinth (Eichhornia crassipes), are currently or formerly present, and control measures are in place at both dams (Northern Territory Government, 2014). There are many invasive aquatic weeds that affect wetlands in the Northern Territory, including mimosa (Mimosa pigra), Gamba grass (Andropogon gayanus), para grass (Brachiaria mutica), rubber vine (Cryptostegia grandiflora) and candlebush (Senna alata) (Territory Natural Resource Management, 2016). Feral animals also affect aquatic habitats, with pigs and buffalo affecting rivers, floodplains and wetlands (Northern Territory Government, 2014).
2 Overview of assets

A key input to the Northern Australia Water Resource Assessment (the Assessment) was the identification of a suite of asset types that would be representative of ecosystem responses to changes in flow, and that were consistent with those that could be considered as part of a water resource planning exercise, or an assessment of environmental impact. This process was undertaken in consultation with state and Commonwealth jurisdictions and other research institutions, particularly those in the National Environmental Science Program Northern Hub.

The broad types of assets considered in the prioritisation process were habitats; individual species of conservation, commercial or recreational significance; groups of species that share characteristics of response (a functional group); or important ecosystems that maintain the health of a river and support ecological functions. All these assets are sensitive to changes in flow, being sustained by either surface or groundwater flows. Threatened and endangered communities are not exclusively used as assets, simply because they are not representative of water needs and ecological processes across a catchment.

Each of these assets is:

- water dependent, with that dependency being supported from water sourced within catchments. For species and functional groups, part or all of their life cycle is supported; for habitats, their condition is supported; and for ecological processes, part or all of these are supported
- distributed across catchments and is therefore suitable for representing flow responses of ecosystems within catchments
- well understood in terms of their dependency on water, in that there is sufficient knowledge available to describe their relationships with flow, either conceptually or quantitatively representative of the range of flow requirements/responses in diverse and distributed water-dependent ecosystems
- significant, being of conservation, recreational or commercial value.

The assets selected include biological aspects that cover the spectrum of flow dependencies, such as life history, habitat and connectivity requirements (Bunn and Arthington, 2002). This coverage is important because flow influences assets in different ways, including triggering growth, reproduction, dispersal and migration processes. It can also provide opportunities to improve condition and build resilience for dry periods. Water-dependent ecological communities have adapted to flood regime characteristics, such as frequency, sequencing, timing and duration (Arthington et al., 2006; Poff et al., 1997), and river regulation has been shown to change these characteristics (Gehrke et al., 1995; Puckridge et al., 1998; Puckridge et al., 2000). Consequently, flow changes have the potential to interact with species life-history traits, thereby changing the distribution,
composition and condition of ecological communities across the landscape (Abell, 2002; Ballinger and MacNally, 2006), with subsequent impacts on biodiversity.

Assets were also selected on the basis of whether they represent a range of flow needs within the catchment. These flow needs include changes to flood and dry flows, and change to end-of-system flows. Diversity of flow, which is often represented in a natural flow regime, represents a range of ecological flow dependencies, including supporting life-history requirements, provision of refugia habitat and system productivity (Bunn and Arthington, 2002). The connectivity of flows and connectivity in rivers is another key part of the analysis. While broad-scale generalised ecological conceptual frameworks (e.g. the river continuum concept (Vannote et al., 1980), and the flood pulse concept (Junk et al., 1989) were used to underpin the analysis and understanding of connectivity processes, such frameworks are not directly applicable at a local scale. Thus, connectivity is evaluated using an asset-based approach.

Threatened and endangered communities are not used as a criteria to define assets, simply because they are not representative of water requirements across catchments. For an investigation of impacts on threatened and endangered species, a thorough analysis of their flow requirements and the diversity of threats that may cause impacts are needed. Data on the distribution of threatened species are also typically poor, due to their rarity and lack of specific investigations.

In the following subsections an evidence base has been constructed that synthesises accessible peer-reviewed evidence from multiple sources, as well as expert knowledge. Peer-reviewed evidence includes journal papers as well as grey literature. Expert knowledge is also sourced from the project team and, through an informal peer review process, where the authors engaged with experts in universities and state governments. The evidence is intended to provide a collation of knowledge, which can be improved over time. Data was used, where possible, to map the distribution of species (Appendix B) and understand their relationship to flow.

Other than flow, ecology can also be affected by water quality, access to groundwater, soil characteristics, physical changes (e.g. grazing impacts), disease and invasive species. While the analysis of these factors are out of scope for this Assessment, their influence is included in the asset descriptions.

### 2.1 Selection of assets for the assessment

A list of priority assets was collated for where there is some water dependency on either surface water or groundwater, in consultation with state and Commonwealth jurisdictions and other research institutions. These assets cover both the freshwater and marine environments, and consist of species of significance, functional groups, important habitats or ecosystem processes. Terrestrial systems (with rainfall as their primary water source) were not considered in the asset selection process. Table 2-1 lists the asset types and names.
<table>
<thead>
<tr>
<th>CATEGORY</th>
<th>TYPE</th>
<th>ASSET</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freshwater</td>
<td>Habitats</td>
<td>Floodplain wetlands</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Waterholes – inchannel</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Riparian vegetation</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Monsoon rainforest</td>
</tr>
<tr>
<td></td>
<td>Species of significance</td>
<td>Magpie goose</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Barramundi</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sawfish</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Whipray</td>
</tr>
<tr>
<td></td>
<td></td>
<td>River sharks</td>
</tr>
<tr>
<td></td>
<td>Functional groups</td>
<td>Migratory fish</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Stable flow spawners</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Turtles or long-necked turtles</td>
</tr>
<tr>
<td></td>
<td>Processes</td>
<td>Fluvial geomorphology</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Floodplain and inchannel productivity</td>
</tr>
<tr>
<td>Marine</td>
<td>Habitats</td>
<td>Mangroves</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Seagrass</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Salt flats</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Coral</td>
</tr>
<tr>
<td></td>
<td>Species of significance</td>
<td>White banana prawns</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mullet</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mud crabs</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Longbums</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Crocodiles</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Snubfin dolphin</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Threadfin</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grunter</td>
</tr>
</tbody>
</table>
3 Asset descriptions: Freshwater

Below is an evidence-based synthesis of information that seeks to describe the assets within the Northern Australia Water Resource Assessment (the Assessment). This knowledge is used to underpin the asset analysis technical report (Pollino et al., 2018).

Each asset is accompanied by a description of its significance, its distribution through the Assessment study areas, and an understanding of its relationship to flow. Information that captures other important drivers of change to the asset is also included. The reviews are accompanied by conceptual models and references to publicly available literature.

3.1 Category: Habitats

3.1.1 Wetlands

Asset description

Across northern Australia there are a diversity of wetland types. These wetlands are highly dynamic systems, with filling and drawing-down cycles closely associated with the natural flooding patterns of the tropical north. These flooding cycles are in turn driven by a regional monsoonal climate in which the majority (95%) of annual rainfall and runoff occurs during the summer period from November to April (CSIRO, 2009).

As with most wetland systems, the inundation of wetlands and their connectivity with the main river channel is known to be a driver of patterns of primary production within rivers. Disruptions to connectivity lead to reductions in the exchange of material (nutrients and organic carbon) between rivers and their floodplains (Junk et al., 1989; Robertson et al., 1999). Disruptions to connectivity are likely to have significant impacts on riverine productivity (Cook et al., 2015; Gawne et al., 2007; Nilsson et al., 2005; Tockner et al., 1999) and consequently on the patterns of primary production within the receiving river as flows cease and refuge holes form.

In general, floodplain wetlands function as nutrient sinks. During flood events, sediment, nutrients and dissolved organic carbon are deposited onto floodplains and associated wetlands (Brinson, 1993; Tockner et al., 1999). During inundation, nutrients and carbon are released from organic matter derived from riparian vegetation (Baldwin, 1999; Robertson et al., 1999). The release of nutrients and carbon promotes an increase in the biomass of algae (primary production) and secondary production (Keckeis et al., 2003; Nielsen et al., 2016; Tockner et al., 1999). In a connected system, as water returns to the river after transiting the floodplain, a significant amount of carbon, nutrients, algae and secondary production (e.g. zooplankton) may be exported (Nielsen et al., 2016) and made available for use in food webs in the receiving river (Balcombe et al., 2012; Junk et al., 1989; Lehman et al., 2008;
Tockner et al., 1999). These resources are then available to support food webs within refuge pools as flows cease within rivers.

Prolonged inundation of wetlands promotes the productivity and biomass of macrophyte growth (Finlayson, 1991; Pettit et al., 2011; Warfe et al., 2011), which may provide important habitat for floodplain fauna, such as magpie geese (*Anseranas semipalmata*). Although carbon derived from macrophytes may not directly feed into aquatic food webs, it is important in soil dynamics and is incorporated into terrestrial food webs via herbivores such as magpie geese (Baldwin et al., 2013; Douglas et al., 2005; Pettit et al., 2011). Sedge species such as *Eleocharis* spp. are particularly important, as they provide crucial food and nesting resources for magpie geese (Finlayson, 2005; Pettit et al., 2011).

Aquatic production in tropical rivers is primarily driven by hydrology and the predictable annual flooding that occurs in the tropics. This predictable hydrological cycle influences the amount of primary production and the availability of nutrients that are required to support food webs (Junk et al., 1989).

Tropical rivers have strong hydrological connections with their associated floodplains, which enable floodplain production to subsidise less productive river habitats. The inundation of floodplains provides a boost to the overall annual energy budget within riverine systems, which drive food webs either via planktonic (algal) or heterotrophic (bacterial/fungal) pathways. Algal production is believed to be the major source of carbon for food webs in tropical river systems (Douglas et al., 2005; Pettit et al., 2011), and in some cases has been shown to support more than 50% of the biomass of fish and invertebrates within tropical rivers. While there is a prevailing view that carbon from terrestrial sources (aquatic macrophytes or riparian vegetation) are less important in most tropical rivers (Douglas et al., 2005; Pettit et al., 2011), recent research has demonstrated that terrestrial vegetation incorporated in the food webs of tropical rivers, can provide an important food resource for consumers (Jardine et al., 2012a; Leigh et al., 2010; Pettit et al., 2012a). The importance of inputs of terrestrial carbon may vary with connectivity (Pettit et al., 2016a).

The term ‘wetland’ groups together a wide range of habitats that share a number of common features. The most important of these are continuous, seasonal or periodic standing water or saturated soils.

Primarily, wetlands are classified using the Ramsar Convention; as ‘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 to the depth of which at low tide does not exceed six metres’ (Secretariat Ramsar Convention, 2013). The definition used by the Queensland Wetland strategy, ‘Areas of permanent or periodic/intermittent inundation, whether natural or artificial, 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 6 metres’ (Department of Environment and Resource Management, 2011) has been adopted for the Assessment, with the exclusion of references to brackish or salt and marine water. This definition also excludes reference to marshes, fens and
peatlands, which are not terms commonly used in Queensland (or elsewhere in northern Australia).

Wetlands are therefore defined as areas:

- of permanent or periodic/intermittent inundation, whether natural or artificial, with water that is static or flowing and fresh
- where the land supports, at least periodically, plants or animals that are adapted to and dependent on living in wet conditions for at least part of their life cycle
- where the substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers
- where the substratum is not soil and is saturated with water, or covered by water, at some time.

This definition excludes riverine wetlands (inchannel pools); floodplains that are intermittently covered by flowing water, but do not meet the hydrophytes and soil criteria; and the riparian zone above the saturation level (Department of Environment and Resource Management, 2011).

Wetlands retained by this definition are classified by the categories defined by the Aquatic Ecosystems Task Group (2012a) as either:

- **Palustrine wetlands** – primarily vegetated non-channel environments of less than 8 ha. They include billabongs, swamps, bogs, springs, soaks, etc., and have more than 30% emergent vegetation
- **Lacustrine wetlands** – large, open, water-dominated systems (e.g. lakes) larger than 8 ha. This definition also applies to modified systems (e.g. dams), which are similar to lacustrine systems (e.g. deep, standing or slow-moving waters).

**Distribution in the Assessment study areas**

**Queensland**

The distribution and occurrence of wetlands associated with the Mitchell catchment in Queensland rivers is well documented, with approximately 8827 wetlands identified (Table 3-1) ([https://wetlandinfo.ehp.qld.gov.au/wetlands/facts-maps/basin-mitchell/](https://wetlandinfo.ehp.qld.gov.au/wetlands/facts-maps/basin-mitchell/)). The majority of these wetlands in the Mitchell catchment are palustrine (Kennard, 2010). The Mitchell River Fan aggregation is listed in the Directory of Important Wetlands (Figure 3-1). The Mitchell River Fan comprises a diversity of floodplain wetlands consisting of seasonal, semi-permanent and permanent habitats (Environment Australia, 2001).

**Table 3-1 Distribution of wetlands in the Mitchell catchment, Queensland**

<table>
<thead>
<tr>
<th>SYSTEM</th>
<th>AREA (km²)</th>
<th>% WETLANDS AREA</th>
<th>% TOTAL AREA</th>
</tr>
</thead>
<tbody>
<tr>
<td>Artificial and highly modified</td>
<td>59.9</td>
<td>2.3</td>
<td>0.1</td>
</tr>
<tr>
<td>Lacustrine</td>
<td>17.1</td>
<td>0.7</td>
<td>0.0</td>
</tr>
<tr>
<td>Palustrine</td>
<td>781.3</td>
<td>30.4</td>
<td>1.1</td>
</tr>
</tbody>
</table>
Lacustrine wetlands are widely spread within the Assessment area, occurring in the catchments of the Adelaide, Mary and Finniss rivers. Palustrine wetlands are less common (Kennard, 2010). The Directory of Important Wetlands includes several wetlands in each of the catchments that are listed as nationally important (Environment Australia, 2001); however, there is no information on the extent and allocation of other wetlands on the respective floodplains (Table 3-2). Important wetlands in this study area are major floodplain–tidal systems typical of the top end, with primarily marginal lakes (Figure 3-2) (Environment Australia, 2001).
Table 3-2 Number and area of nationally important wetlands in the Darwin catchments

<table>
<thead>
<tr>
<th>SYSTEM</th>
<th>AREA (HA)</th>
<th>NO. WETLANDS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adelaide River</td>
<td>239,800</td>
<td>2</td>
</tr>
<tr>
<td>Finniss River</td>
<td>131,700</td>
<td>3</td>
</tr>
<tr>
<td>Mary River</td>
<td>1,983</td>
<td>1</td>
</tr>
<tr>
<td>Wildman River</td>
<td>unknown</td>
<td>unknown</td>
</tr>
</tbody>
</table>

Source: (Environment Australia, 2001)

Figure 3-2 Distribution of wetlands in the Darwin catchments

Western Australia

Lacustrine wetlands are widespread in the Fitzroy catchment (Kennard, 2010). Close et al. (2012a) and Karim et al. (2016) identified 30 palustrine wetlands greater than 6 ha in area that were perennial and non-perennial, but recognise that there are many smaller wetlands on the adjacent floodplain.
The Directory of Important Wetlands (Environment Australia, 2001) includes three wetlands of national importance with an aerial extent of 30,230 ha. Two of these, the Camballin Floodplain and Gladstone Lake, are floodplain systems, while the third, Geikie Gorge, is within the river channel. Gladstone Lake is a perennial lake of approximately 25 ha during the dry season, expanding to 100 ha during the wet season (Environment Australia, 2001). The Camballin Floodplain is a wetland complex consisting of two large swamps (Le Lievre and Moulamen swamps), 17-Mile Dam, seven seasonal wetlands and Snake Creek Billabong (Figure 3-3) (Environment Australia, 2001).

**Figure 3-3 Distribution of wetlands in the Fitzroy catchment**

**Perenniality**

Across northern Australia, 83% of lacustrine wetlands are perennial (permanent or near permanent). In contrast, only 16% of palustrine wetlands are perennial (Kennard, 2010).
**Conceptual model**

The conceptual model for wetlands in northern Australia is shown in Figure 3-4, with the accompanying narrative given in Table 3-3. Threats to wetlands are derived from changes in the water regime that will modify connectivity and change the extent and suitability of habitat, and changes to the physical habitat through modification of land use and the introduction of invasive plants and animals (Finlayson and Rea, 1999). Other threats (not included in the model) are derived from economic development in wetlands, bureaucratic obstacles, lack of information or poor access to information, and poor general awareness of the values and benefits derived from wetlands (Finlayson and Rea, 1999). The identified threats will influence the biodiversity and ecological functioning of wetlands at both local and landscape scales.
Figure 3-4 Conceptual model showing the relationship between threats, drivers, effects and outcomes for wetlands in northern Australia
## Conceptual model narrative

Table 3-3 Knowledge underpinning the conceptual model of wetlands in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4 EFFECT</th>
<th>5 OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Infrastructure</td>
<td>Connectivity</td>
<td>Flooding during the wet season provides lateral connectivity between rivers and wetlands which allows the exchange of nutrients and biota. This exchange promotes productivity and recruitment of wetland-riverine biota.</td>
<td>Decreased exchange of nutrients</td>
<td>Decreased productivity</td>
<td>In general, floodplain wetlands function as nutrient sinks. During flood events, sediment, nutrients and dissolved organic carbon are deposited onto floodplains and associated wetlands (Brinson, 1993; Tockner et al., 1999). During inundation, nutrients and carbon are released from organic matter derived from riparian vegetation (Baldwin, 1999; Robertson et al., 1999). The release of nutrients and carbon promotes an increase in the biomass of algae (primary production) and secondary production (Keckeis et al., 2003; Nielsen et al., 2016; Tockner et al., 1999). In a connected system, as water returns back to the river after transiting the floodplain, a significant amount of carbon, nutrients, algae and secondary production (e.g. zooplankton) may be exported (Nielsen et al., 2016) and made available for use in food webs in the receiving river (Balcombe et al., 2012; Junk et al., 1989; Lehman et al., 2008; Tockner et al., 1999). These resources are then available to support food webs within refuge pools as flows cease within rivers. Retention of water in wetlands and the loss of connectivity between wetlands and rivers will reduce the movement of algae, nutrients and carbon into the main river channel. Typically, rivers in northern Australia are low in nutrient status and are likely to be sensitive to changes in nutrients loads (Douglas et al., 2005). These effects are likely to become more apparent in the dry season as rivers reduce to a series of isolated refuge pools. The loss of connectivity between wetlands and rivers is known to have impacted on the movement of fish populations worldwide (Bain et al., 1988). For many fish species, individual growth and population condition is associated with flooding (King et al., 2009). This is either by the provision of breeding cues, the accessibility of suitable habitat or through flow-mediated changes in food availability. While ponding of water will not prevent the movement of fish into wetlands, it will prevent movement back into the main channel of adults and juveniles. As water is removed from the ponded wetlands for agricultural large numbers of fish may become stranded and ultimately die (Jones and Stuart, 2008).</td>
</tr>
<tr>
<td>Water extraction</td>
<td>Inundation patterns</td>
<td>This exchange promotes productivity and recruitment of wetland-riverine biota. Connectivity and associated inundation patterns may be altered by imposing infrastructure (levees, block banks, regulators) that prevent lateral connectivity and or promote permanent inundation or rapid drawdown.</td>
<td>Decreased migration between wetland and river channel</td>
<td>Reduced recruitment of wetland reliant biota</td>
<td></td>
</tr>
<tr>
<td>Water extraction</td>
<td>Connectivity</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Climate change</td>
<td>Inundation patterns</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Climate change</td>
<td>Connectivity</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/STRESSOR</td>
<td>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4 EFFECT</td>
<td>5 OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
</tr>
<tr>
<td>----------------------</td>
<td>------------------</td>
<td>-----------------------------------------------</td>
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<td>-----------</td>
<td>--------------------------</td>
</tr>
<tr>
<td><strong>Fire</strong></td>
<td>Fire had to do with the density of monocotyledon (e.g., grasses) plants and promotes an increase in dicotyledons (e.g., trees) Frequency and intensity of fire may lead to a contraction of non-savanna communities and some woodlands</td>
<td>Changed extent and quality of habitat</td>
<td>Changes in the abundance and distribution of biotic components</td>
<td>Increasing periods of drought will increase the potential for fires and will have a significant effect on wetland vegetation (Covich et al., 1997) with potential effects favouring the occurrence of introduced invasive plant species (D’Antonio, 2000; Hogenbirk and Wein, 1991).</td>
<td></td>
</tr>
</tbody>
</table>

| **Land use** | Inundation patterns Inundation patterns may be altered by imposing structure (leves, block banks, regulators) that prevent lateral connectivity and or promote permanent inundation or rapid drawdown | Decreased exchange of nutrients | Decreased productivity | Changed land use practices and urban development leads to increased nutrient, sediment and other contaminants being washed into wetland and riverine systems (Bayley, 1995; Brodie and Mitchell, 2005; Douglas et al., 2005). Increased sedimentation and associated increases in turbidity reduce primary production which in turn will modify food webs (Davies-Colley and Smith, 2001). Increase nutrient loads are likely to lead to a changes in the algal community (Allan, 2004; Douglas et al., 2005) and potentially eutrophication may occur in of the refuge pools as rivers cease to flow (Brodie and Mitchell, 2005; Davis and Koop, 2006; Hamilton and Gehrke, 2005). Increases in contaminates may increase mortality rates and decrease reproductive capacity (Allan, 2004). |

<p>| Fire | Frequency and intensity of fire may lead to a contraction of non-savanna communities and some woodlands | Changed extent and quality of habitat | Changes in the abundance and distribution of biotic components | Fires in northern Australia are frequent and extensive (Russell-Smith et al., 1997). Increasing periods of drought will increase the potential for fires and will have a significant effect on wetland vegetation. The intensity of fires increases with the duration of the dry period and has significant effects on the structure of plant communities (Williams et al., 1999) with potential effects favouring the |</p>
<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4 EFFECT</th>
<th>5 OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
</table>
| Sedimentation         | Sedimentation     | Sedimentation reduces habitat quality via the deposition of fine sediments. Sedimentation will also smother dormant propagules preventing germination and hatching.  
|                       | Sedimentation     | Sedimentation as a consequence of anthropogenic land use activities will lead to the deposition of fine particles on existing sediments (Lake et al., 2000). Not only will this reduce the range of habitats for benthic biota but will smother dormant seeds of plants and eggs of invertebrates preventing germination and hatching (Gleason et al., 2003). |
| Eutrophication        | Increased eutrophication reduces light penetration and leads to a decline in the diversity of submerged plants.  
|                       | Eutrophication    | Retention of water in wetlands and the loss of connectivity between wetlands and rivers will reduce the movement of algae, nutrients and carbon into the main river channel. Typically, rivers in northern Australia are low in nutrient status and are likely to be sensitive to changes in nutrients loads (Douglas et al., 2005). These effects are likely to become more apparent in the dry season as rivers reduce to a series of isolated refuge pools. Consequences of eutrophication include the development of noxious algal blooms, odours and anoxic conditions which have the potential to be lethal to waterbirds and aquatic biota (Davis and Froend, 1999). |
| Invasive plants       | Competition       | Invasive plants often form monotypes, which alter habitat structure, lower biodiversity, change nutrient cycling and productivity and modify food web.  
|                       | Invasive plants   | Wetlands in northern Australia are becoming increasingly threatened by invasive plants such as Mimosa, Salvinia and introduced pasture grasses. Wetlands are susceptible to invasive plants due to water and nutrient availability (Rea and Storrs, 1999). Invasive species alter habitat quality and depending on the species may inhibit the growth of native species (Rea and Storrs, 1999). |
| Invasive animals      | Grazing/trampling | Degradation of wetland habitats contribute to erosion and prevent native plants from establishing.  
|                       | Grazing/trampling | Introduced feral animals (i.e. buffalo) cause pugging of surface sediments, removal of native vegetation and increased erosion. In turn this has led to an increase in the distribution and abundance of invasive plant species (Finlayson et al., 1997). |
| Sea level rise        | Salinisation      | Increased salinisation results in a loss of freshwater adapted plants.  
|                       | Salinisation      | Coastal areas of northern Australia are composed of estuarine and freshwater wetlands (Eliot et al., 1999). Changes in salinity within freshwater wetlands will result in changes in freshwater plant communities (Nielsen and Brock, 2009; Nielsen et al., 2008) and changes in the distribution of biota (birds/fish/turtles) associated with freshwater wetlands (Eliot et al., 1999). |
3.1.2 WATERHOLES

Asset description

Northern Australia contains rivers with highly seasonal flow regimes (McJannet et al., 2014; Petheram et al., 2008). As flows recede during the dry season, sections of rivers often retract to a series of discrete, disconnected waterholes (McJannet et al., 2014; Waltham et al., 2013). For the purpose of the Assessment, waterholes are defined as locations within seasonal watercourses that retain water during periods of low surface flow. This definition is exclusive of large or terminal lakes, and wetlands that form on the floodplain. Waterholes can include bodies of water maintained or supported by groundwater inputs. During dry-season low-flow or cease-to-flow periods, the size, quality and connectivity of waterholes remaining within the landscape decreases (Department of Environment and Resource Management, 2010; McJannet et al., 2014). The waterholes that remain provide important ecosystem functions, including providing refuge habitat for water-dependent flora and fauna, and are an important cultural resource during the dry season (Centre of Excellence in Natural Resource Management, 2010; McJannet et al., 2014).

Changes in the flow regime associated with water resource development, surface and groundwater extraction and climate change have the potential to alter the natural filling and drying cycles of waterholes (McJannet et al., 2014; Waltham et al., 2013). Water resource development can affect waterholes through changes in hydrology associated with increases in the intensity and duration of low-flow periods, changes to the magnitude and timing of seasonal high-flow events and through the establishment of a flow regime that creates a perennial stream from a previously ephemeral river channel. Such changes could have significant impacts on the flora and fauna that depend upon the persistence of waterholes to retain populations during dry periods, and the natural cycle of reconnecting flows required to restore populations and enhance habitat across the landscape (James et al., 2013; Lymburner and Burrows, 2008), including the need for natural low flows to maintain the ecological balance of the habitat.

The ecological function of waterholes operates over both local and landscape scales (Department of Environment and Resource Management, 2010). At the local scale, the quality of habitat within a waterhole encompasses all factors required to sustain species, including water quality, habitat availability, food-chain structure and community composition (Department of Environment and Resource Management, 2010; DSITIA, 2014). Within each waterhole, as dry conditions prevail, likely outcomes include resources becoming scarcer (increased competition, loss of food-chain links and decreased total productivity), higher predation, decreases in habitat quality (higher temperatures and greater fluctuations, oxygen deprivation, increased concentration of pollutants and salinity, loss of specific habitat) (Department of Environment and Resource Management, 2010; DSITIA, 2014). The size of waterholes can provide an indication of the resilience of the refuge to periods without surface flows, as waterhole size is often associated with both
breadth of habitat and waterhole persistence during dry conditions (Department of Environment and Resource Management, 2010).

At the landscape scale, the number of waterholes remaining between reconnecting flows and the distance or presence of barriers between waterholes will affect the ability to restore populations and species across the landscape (Department of Environment and Resource Management, 2010). High flows and connectivity to the floodplain is important for nutrient exchange and recharge of groundwater stores to support riparian vegetation and dry season baseflow inputs in many locations (Waltham et al., 2013). Within study areas with few waterholes, suitable refuge habitat is likely to be of greater ecological value than study areas where waterholes are more abundant (Lymburner and Burrows, 2008; McJannet et al., 2013). Longer durations between connecting flows and increased extraction of water from waterholes increases the probability of loss of individual waterholes, as well as the probability of loss of species from within each remaining waterhole, thereby leading to cumulative regional impacts on biodiversity (Department of Environment and Resource Management, 2010).

Creating instream barriers can create new waterhole habitats that can provide persistent or refuge habitat for some species from previously ephemeral habitat. This could result in changes to community structure by establishing refuges for large, predatory fish species where none previously occurred.

**Distribution in the Assessment study areas**

Waterholes are distributed across all three Assessment study areas (Figure 3-5, Figure 3-6 Figure 3-7). Factors influencing the distribution and persistence of waterholes include geomorphic features, location and connectivity to sources of ground or hyporheic water. Waterhole mapping of persistent waterholes was undertaken using remote-sensing data. Waterholes were mapped at the end of each dry season using Landsat imagery (as described in Sims et al., 2016). An in-channel mask containing a 500 m buffer from the watercourse was 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 was calculated. This was to allow for the fact that a waterhole can vary in shape and location through time. See Sims et al. (2016) for further details regarding the method used to create the waterholes mapping dataset.
Figure 3-5 Distribution of persistent waterholes in the Fitzroy catchment
Figure 3-6 Distribution of persistent waterholes in the Darwin catchments
Figure 3-7 Distribution of persistent waterholes in the Mitchell catchment

**Conceptual model**

The conceptual model for waterholes (Figure 3-8, with the accompanying narrative given in Table 3-4) includes a range of threatening processes to the biodiversity and ecosystem function values of waterholes in northern Australia. The threatening processes include changes to hydrology associated with water resource development and extraction of ground and surface water, as well as other threatening processes leading from changes in land management practices, invasive species and climate change. Waterhole biodiversity and ecological function values are the model endpoint, encompassing the waterhole’s ability to support existing, aquatic-dependent flora and fauna and current ecological function across both local and landscape scales.
Figure 3-8 Conceptual model showing the relationship between threats, drivers, effects and outcomes for waterhole biodiversity and ecosystem function in northern Australia
Conceptual model narrative

Table 3-4 Knowledge underpinning the conceptual model of waterholes in northern Australia

* Many of the stressors associated with ephemeral river reaches still occur in perennial river reaches, but these are not considered as ‘waterhole impacts’ as in-channel waterholes occur in ephemeral river reaches (e.g. invasive species, changes in land use management and instream connectivity)

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>Ephemeral river reaches and river regulation</td>
<td>Changes in timing and magnitude of wet season flows</td>
<td>Wet-season flows provide recharge and flushing of waterholes. High flows are important for longitudinal and lateral connectivity allowing movement of species and nutrient exchange from floodplains and wetlands.</td>
<td>Delayed wet-season flows may extend the drawdown period of some waterholes. Reduced wet-season flows may limit connectivity and nutrient exchange from floodplains. Changes in timing of wet-season flows may disrupt migration patterns and life-cycle cues of some species.</td>
<td>Species requiring seasonal high flows may lose supporting ecosystem function including migration pathways and life-cycle cues resulting in reduced population condition. Water quality, productivity and food webs can be modified through changes to flow regimes.</td>
<td>Flooding is an important factor that sustains many environmental assets by providing connectivity across the floodplain and facilitating migration (Close et al., 2012b). Floods reset water bodies so that they become biophysically very similar by the end of the wet season, and then diverge strongly during the dry season (Pettit et al., 2012b). Extending the length of time between flushing events from floods could be detrimental to the temperature, oxygen and nutrient status of pools (McJannet et al., 2013).</td>
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<td>High flows remove sediment build up from waterholes through scouring</td>
<td>Regulation leading to reduced high-flow volumes and flow velocities lowers rates of scouring from waterholes allowing sediments to build up and infill waterholes.</td>
<td>Sediment build up and waterhole infilling leads to loss of habitat quality in refuge waterholes.</td>
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<td>Increased dry-season flows downstream of storages</td>
<td>Biodiversity of waterholes is adapted to natural drying and filling cycles</td>
<td>Increased baseflows during the dry season will result in changed habitat conditions as ephemeral rivers transition to perennial.</td>
<td>River regulation in ephemeral rivers can result in community changes as species suited to the new environmental conditions begin to dominate the community and risk to population through changes in predation.</td>
<td>Regulation of flow via dams and diversions can reduce the frequency of high-flow events, which are essential for scouring accumulated sediments from waterholes. Loss of waterhole habitats owing to in-infilling by sediments has been documented in many locations (McJannet et al., 2014).</td>
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<td>Homogenisation of the flow regime</td>
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<td></td>
<td>Variability in flows is important in maintaining resilience by maintaining a range of biological and physical characteristics that interact with life-history traits of the biota (Pettit et al., 2012b).</td>
</tr>
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<td>1 HABITAT AND THREATS</td>
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<tr>
<td>Reduced connectivity associated with the creation of barriers</td>
<td>Longitudinal connectivity is important for meta-population dynamics, movement and migration supporting critical life-cycle processes for some species</td>
<td>Creation of barriers can reduce instream connectivity and longitudinal movement of individuals along the river. This can have implications for migration and meta-population dynamics. Movement by individuals supports meta-population process in waterholes including recovery of populations from waterholes which previously have lost populations.</td>
<td>Species requiring instream connectivity may lose migration pathways and access to breeding habitat resulting in reduced population condition. Loss of instream connectivity may limit meta-population processes and recovery of lost populations.</td>
<td>Synthesis of literature.</td>
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<tr>
<td>Creation of permanent standing water behind weirs, barrages etc.</td>
<td>Biodiversity of waterholes is adapted to natural drying and filling cycles</td>
<td>Changes in habitat condition, structure and ecosystem processes. Changes in predation levels.</td>
<td>River regulation in ephemeral rivers can result in community changes as species suited to the new environmental conditions begin to dominate the community and risk to some species through changes in predation.</td>
<td>Synthesis of literature.</td>
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**Ephemeral river reaches and water extraction (upstream)**

<p>| Reduced dry-season flows | Waterhole persistence throughout the dry season is associated with the intensity and duration of the low flows | A decrease in dry-season flows (base flows and hyporheic flow) of waterholes is likely to enhance the drawdown process. Extended cease-to-flow periods increases the time for waterhole drawdown putting additional stress on waterhole biodiversity. | Smaller or less permanent waterholes become increasingly vulnerable under reduced dry-season flows as a consequence of drawdown. This can have biodiversity impacts associated with loss of waterholes from the landscape and lower habitat quality within remaining waterholes. The period of disconnection may increase reducing movement and nutrient exchange. | A strong relationship ($R^2=0.72$) between time since zero flow and pool area was observed (Mclennan et al., 2013). The duration of zero flow is closely tied to the persistence of instream pools (Mclennan et al., 2013). Extending the length of time between flushing events from floods could be detrimental to the temperature, oxygen and nutrient status of pools (Mclennan et al., 2013). Waterholes less than 1 m deep experience diurnal temperature amplitudes around 5°C; much higher than the ~ 2°C observed in waterholes over 2 m deep (Waltham et al., 2013). As temperatures increase beyond species thermal preferences, growth rates decline and fish become more |</p>
<table>
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<tr>
<th>1 HABITAT AND THREATS</th>
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<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
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<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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<tbody>
<tr>
<td><strong>Ephemeral river reaches and water extraction</strong> (using waterholes or local groundwater as water sources)</td>
<td>Direct waterhole drawdown</td>
<td>The volume of water within a waterhole relates to the persistence of the waterhole during the dry season</td>
<td>Faster drawdown will lead to increased ephemerality in waterholes not sufficiently large or connected to buffer water extraction.</td>
<td>Decreased water permanence and/or degradation of waterhole habitat quality leading to a lowering of biodiversity and ecosystem function values.</td>
<td>Natural reduction in water levels (due to evaporation and seepage losses) would be sufficient to significantly increase daily temperature maxima and would decrease the likelihood of a cool bottom layer forming (which would mean that fish would be unable to escape the heat). Depth reductions due to water extraction would have the same effect (Waltham et al., 2013).</td>
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<td>Lowering of the groundwater table or reduced hyporheic water surrounding waterholes</td>
<td>Connection to groundwater sources can sustain surface water persistence</td>
<td>Disconnection of waterholes from groundwater and loss of surface water expression.</td>
<td>Decreased water permanence and/or degradation of waterhole habitat quality leading to a lowering of biodiversity and ecosystem function values.</td>
<td>Permanent pools in the Fitzroy River are very likely to be significantly affected by substantial groundwater drawdown (Centre of Excellence in Natural Resource Management, 2010). Some waterholes were already exhibiting signs of drought stress towards the end of the investigation. For example, several Flinders catchment waterholes which had held water for the duration of the previous dry season were actually drying out by the end of this investigation, even though they had received enough wet-season inflow to temporarily refill them on at least one occasion. This suggests that seepage losses were much greater this year than they had been the previous year, and implies that the watertable had fallen to the point where groundwater levels were now below the level of the streambed (Waltham et al., 2013).</td>
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<td>Riparian vegetation often depends upon groundwater or hyporheic water during the dry season</td>
<td>Loss of connectivity to groundwater or hyporheic water for riparian vegetation can lead to stress or loss of riparian vegetation communities.</td>
<td>Where disconnection to groundwater sources occurs, riparian vegetation can become stressed or lost. Loss of riparian vegetation can lead to further impacts on waterholes including changes in thermal dynamics, habitat modification and erosion.</td>
<td>Synthesis of literature.</td>
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<tr>
<td><strong>Ephemeral river reaches and drying climate change</strong></td>
<td>Reduced dry-season flows increased drawdown</td>
<td>Climate predictions vary, increased aridity scenarios pose specific threat to waterholes based upon drying and filling properties of waterholes</td>
<td>Extended low-flow or cease-to-flow periods increases the time for waterhole drawdown putting additional stress on waterholes. A decrease in dry-season flows (base flows and hyporheic flow) of waterholes is likely to enhance the drawdown process. Increases in evapotranspiration rates can increase additionally the rate of waterhole drawdown and thermal stress.</td>
<td>Smaller or less permanent waterholes become increasingly vulnerable under reduced dry-season flows as a consequence of drawdown. This can have biodiversity impacts associated with loss of waterholes from the landscape and lower habitat quality within remaining waterholes. Thermal stress can be increased in association with both warmer climate and with lower volumes of water.</td>
<td>Low flows under dry climate change scenarios are likely to be altered significantly. Some areas are likely to experience considerable increases in the duration of low and zero flows, which may have major ecological impacts (Close et al., 2012b). Large reductions in rainfall under a drier future climate may severely compromise the persistence of key aquatic refugia in some rivers (McJannet et al., 2014). Combining climate change with development pressures can exacerbate changes to low-flow conditions (Close et al., 2012b). Shallower waterholes had much larger diurnal temperature fluctuations and this will affect the suitability of these waterholes as aquatic habitats (Waltham et al., 2013).</td>
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<tr>
<td><em><em>Perennial river reaches</em> and extraction or drying climate change</em>*</td>
<td>Creation of ephemeral reaches from perennial reaches</td>
<td>Aquatic biodiversity adapted to natural drying and filling cycles</td>
<td>Transition of perennial reaches to ephemeral habitat will change the community structure and existing connectivity.</td>
<td>Changes in community composition. Extent of ephemeral habitat further isolating upstream aquatic communities and impacting regional biodiversity.</td>
<td>Synthesis of literature.</td>
</tr>
<tr>
<td><strong>Ephemeral river reaches and change in land use and intensification</strong></td>
<td>Changes in nutrient, pesticide and sediment runoff from agriculture</td>
<td>Water quality is an important determinant of habitat quality for many species and ecosystem functions</td>
<td>Increases in erosion and sedimentation increases turbidity and can increase waterhole infilling. Nutrients can increase eutrophication, increasing the risk of algal blooms and blackwater events in waterholes. Pesticides can accumulate in waterholes and concentrate during waterhole drawdown to harmful levels.</td>
<td>Reduction in habitat quality can result from increases in turbidity resulting in a decline in the photosynthetic active radiation zone resulting in lower primary productivity. Degradation or loss of aquatic macrophytes can result from turbidity increases leading to reduction in habitat quality. Degradation or loss of existing riparian vegetation communities by physical bank disturbance can change habitat structure. Changes in nutrient and pesticide levels reduce habitat quality and evidence of increased productivity due to enhanced nutrient inputs (i.e. cultural eutrophication) has been observed in most north Queensland catchments where intensive agriculture has been pursued (Waltham et al., 2013). Variables related to waterhole clarity and light penetration accounted for most of the explained variation in biotic assemblage composition between catchments (Waltham et al., 2013). Previous agricultural irrigation developments in tropical Australia have been associated with decreased river and offshore water quality. These reductions in water quality are directly related to the removal of pre-existing ground cover and the application of fertilisers and pesticides.</td>
<td>Evidence of increased productivity due to enhanced nutrient inputs (i.e. cultural eutrophication) has been observed in most north Queensland catchments where intensive agriculture has been pursued (Waltham et al., 2013). Variables related to waterhole clarity and light penetration accounted for most of the explained variation in biotic assemblage composition between catchments (Waltham et al., 2013). Previous agricultural irrigation developments in tropical Australia have been associated with decreased river and offshore water quality. These reductions in water quality are directly related to the removal of pre-existing ground cover and the application of fertilisers and pesticides.</td>
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<td>Changed fire regimes</td>
<td>Riparian vegetation can be sensitive to changes in the fire regime</td>
<td>Increases in frequency or intensity of fire regimes can lead to degradation and loss of riparian vegetation and changes in habitat structure.</td>
<td>Degradation or loss of existing riparian vegetation community. Changes in habitat structure. Increases in solar heating of waterholes due to loss of shading.</td>
<td>Fertiliser and pesticide applications are in part absorbed and used by crops, however, during rain events quantities of nutrients and pesticides, as well as exposed soils, are washed into adjacent rivers (Waltham et al., 2013).</td>
<td>Synthesis of literature.</td>
</tr>
<tr>
<td>Degradation through livestock access to waterholes</td>
<td>Waterhole bank vegetation stabilises erosion</td>
<td>Livestock can affect waterhole quality by degrading riparian vegetation, increasing erosion and causing turbidity through physical disturbance of the sediment.</td>
<td>Increases in turbidity result in a decline in the photosynthetic active radiation zone, resulting in lower waterhole primary productivity. Degradation or loss of aquatic macrophytes can result from increases in turbidity. Degradation or loss of existing riparian vegetation communities by physical disturbance or grazing can change habitat structure and can increased solar heating of waterholes.</td>
<td>Impact from cattle grazing on riparian vegetation was evident at all waterholes (Waltham et al., 2013). Cattle and other feral animals using waterholes may have more detrimental long-term effects such as eliminating susceptible plant species and providing enhanced dispersal opportunities for aquatic invasive plants by carrying propagules to and from waterholes in mud or faeces (Pettit et al., 2012b). Shallow waterholes are more susceptible to high turbidity because of wind-derived resuspension of sediments which can be exacerbated by cattle disturbance, with cattle having greater access to these shallow waterholes (Pettit et al., 2012b). Increase in nutrient concentrations may lead to the eutrophication of the waterhole, resulting in algal blooms and a reduction in water quality (Pettit et al., 2012b). Providing access to river systems for cattle not only had direct water quality impacts through the addition of faecal material and nutrients, but the access tracks themselves concentrate sheet flow during rainfall and contribute high loads of sediments (Waltham et al., 2013). Trampling vegetation and soil compaction also reduces the extent and quality of riparian vegetation (Waltham et al., 2013).</td>
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<td>Increased fishing pressure</td>
<td>Waterholes provide important dry-season refuge for fish species</td>
<td>Access to waterholes provides fishing opportunities which facilitate removal of target species from the refuge habitat.</td>
<td>Targeting of waterholes for recreational, cultural or commercial fisheries can reduce population sizes of target species and change community composition through removal of predator species.</td>
<td>Freshwater fisheries are ‘predisposed’ to intensive harvest because fish are constrained to relatively restricted habitats (Close et al., 2014). It is also reasonable to predict that the removal of large fish, especially apex predators, will result in changes to community structure, loss of biodiversity and ecosystem function (Close et al., 2014). The spatial dynamics of fishing pressure, together with the restriction of fish populations to isolated dry-season refugia, suggests that target species will be predisposed to the effects of fisheries harvest (Close et al., 2014). Despite remoteness, low population densities and limited accessibility, the data indicated that the abundance of some targeted species, particularly <em>L. calcarifer</em> and probably <em>H. jenkinsi</em>, were correlated with accessibility and showed a negative relationship to fishing pressure, whereas non-target species showed no discernible trend (Close et al., 2014).</td>
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<tr>
<td>Ephemeral river reaches and invasive species</td>
<td>Invasive aquatic macrophyte species</td>
<td>Aquatic macrophytes are important for providing habitat structure and primary productivity</td>
<td>Clogging of waterholes, shading of submerged macrophytes and changes in habitat structure, competition with native aquatic macrophytes.</td>
<td>Loss of important habitat and food-web structure. Changes in community composition from habitat change.</td>
<td>The biggest difference in the composition of macrophyte community in the present study was seen at Fishhole Creek waterhole, with the dominance of the floating exotic herb <em>Eichhornia crassipes</em>, which resulted in this waterhole having the lowest species richness and diversity. This plant is native to South American floodplains and grows and reproduces very rapidly, quickly occupying all available space (Pettit et al., 2012b).</td>
</tr>
<tr>
<td>Invasive riparian plant species</td>
<td>Riparian vegetation is important for providing habitat structure and in performing ecosystem functions including</td>
<td>Degradation of native riparian vegetation. Changes in community composition.</td>
<td>Loss or degradation of native vegetation communities.</td>
<td>Synthesis of literature.</td>
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<td>1 HABITAT AND THREATS</td>
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<td>stabilisation of banks</td>
<td>Invasive aquatic fauna (e.g. fish, turtles, frogs)</td>
<td>Degradation of native species populations. Changes in community composition.</td>
<td>Changes in community composition. Increases in competition, predation levels on native species.</td>
<td>The cane toad has become a considerable ecological problem in northern Australia (Waltham et al., 2013).</td>
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<tr>
<td>Invasive terrestrial fauna (e.g. pigs)</td>
<td>Waterhole habitat condition is important for providing the environmental conditions required by many species</td>
<td>Degradation and loss of riparian vegetation. Erosion and turbidity increases. Increased predation (for example turtle eggs).</td>
<td>Degradation or loss of existing riparian vegetation community. Degradation or loss of aquatic macrophytes association with turbidity increases. Increases in predation levels (for example turtle eggs) leading to a decline in native species.</td>
<td>Overall riparian condition was lower for the waterholes in the Flinders catchment due to greater impacts from feral animals compared to the Gilbert catchment waterholes (Waltham et al., 2013). Feral pigs may also have more direct impacts through consuming buried plant rhizomes and tubers as well as predating on turtles and mussels. Aestivating in the mud, which become accessible as the waterholes dry out at the end of the dry season (Pettit et al., 2012b). Cattle and other feral animals using waterholes may have more detrimental long-term effects such as eliminating susceptible plant species and providing enhanced dispersal opportunities for aquatic invasive plants by carrying propagules to and from waterholes in mud or faeces (Pettit et al., 2012b).</td>
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3.1.3 RIPARIAN VEGETATION

Asset description

In northern Australia, riparian vegetation forms distinct, green bands between riverine and terrestrial biomes (Dixon et al., 2011) and is associated with a network of intermittent, ephemeral watercourses and streams leading to permanent rivers that drain the vast tropical savannas. While a conspicuous part of the landscape, these systems are the most poorly understood component of tropical savannas, especially in relation to the links between biology and flow regimes (Pettit et al., 2016b) and their interaction with fire (Douglas et al., 2015).

Riparian zones act as the vegetated interface between terrestrial and aquatic ecosystems. They are dependent on strong seasonal surface and groundwater flow regimes with high inter-annual variability and are vulnerable to disturbance. The zones are strongly associated with wetlands (Section 3.1.1) and monsoon forest patches (Section 3.1.4). Riparian zones are regarded as highly diverse, dynamic and complex (Naiman and Decamps, 1997), acting as a thermal buffer to streams. They influence instream primary production; nutrient interception, storage and release; improving bank stability; the provision of coarse woody material as habitat and substrate for fish, invertebrates and microalgae; channel morphology and habitat diversity (Pusey and Arthington, 2003). The riparian vegetation, especially litter fall during the dry season, adds terrestrial organic matter, and nutrients (especially phosphorous and dead organic matter carbon) during the first flushes of the wet season. These kickstart algal growth critical to the tropical food web (Pettit et al., 2017). This period between the dry and wet seasons is a key period of material transport and primary production, with most of large consumers’ growth from feeding occurring at this time.

Riparian zones act as both sources and sinks of energy and matter for aquatic ecosystems, which are linked longitudinally, laterally and vertically as open ecosystems under the influence of hydrological and geomorphic processes (Tabacchi et al., 1998; Ward, 1989). Hydrological and geomorphic processes are primary ecosystem drivers, whereas chemical and biological factors act as secondary response variables that are constrained by the primary drivers (Tabacchi et al., 1998). Geomorphic factors include soil physics, slope, catchment area gradient, groundwater influences and channel morphology (Tabacchi et al., 1998).

The riparian zones of semi-arid and monsoonal northern Australia are different to those of the wet tropics, with fewer species (Pettit et al., 2016b), but often with species more akin to those in northern rainforests and monsoonal forest patches than the surrounding savanna. Woinarski et al. (2000) describe the riparian zones of northern Australia as being collated to nine floristic variables that summarise the cover of the main distinctive riparian vegetation elements:

• pandanus (the palm-like Pandanus spiralis and P. aquaticus)
• eucalypt (principally the river red gum, *Eucalyptus camaldulensis*, and coolibah, *E. microtheca*)

• melaleuca (paperbarks of the species *Melaleuca viridiflora*, *M. cajuputi*, *M. leucadendra* and *M. argentea*)

• bamboo (the very tall ‘woody’ graminoid *Bambusa arnhemica*)

• casuarina (the she-oak *Casuarina cunninghamiana*)

• lophostemon (swamp mahogany, *Lophostemon lactifluus*, and northern swamp box, *L. grandiflorus*)

• rainforest trees and shrubs including *Nauclea orientalis*, *Ficus* spp., *Terminalia microcarpa*, *T. platyptera*, *Acacia auriculiformis*, *Barringtonia acutangula*, *Xanthostemon eucalyptoides*, *Syzygium* spp., *Carallia brachiate* and *Cathormium umbellatum*)

• graminoids (all grass and sedge species other than bamboo)

• weeds.

In the Daly region (Northern Territory), six species contribute 82% of the standing basal area, being *M. argentea*, *M. leucadendra*, *E. bella*, *C. umbellatum*, *N. orientalis* and *C. cunninghamiana* (Erskine et al., 2003).

Using the information on riparian vegetation on northern Australia, particularly in or close to the NAWRA regions (DSITIA, 2014; Eamus et al., 2006; Erskine et al., 2003; O’Grady et al., 2006; Pettit and Froend, 2001; Pusey and Kath, 2015; Pusey and Kennard, 2009; Storey et al., 2001; Tooth and Nanson, 2000; Woinarski, 2004; Woinarski et al., 2000), as well as vegetation mapping, a list of potential riparian species was generated. Where possible, species were grouped by genus or family, otherwise they were grouped based on having similar water requirements where possible.

**Distribution in Assessment study areas**

The distribution of selected vegetation species are shown in Figure 3-9 to Figure 3-38, with the species groups defined in Table 3-5.

**Table 3-5 Vegetation species groups included in the species distribution maps**

<table>
<thead>
<tr>
<th>GROUP</th>
<th>SPECIES</th>
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</table>
| Group A | *Eucalyptus camaldulensis*  
*Eucalyptus coolibah*  
*Eucalyptus microtheca*  
*Eucalyptus victrix* |
| Group B | *Melaleuca argentea*  
*Melaleuca glomerata*  
*Melaleuca leucadendra*  
*Melaleuca viridiflora* |
<table>
<thead>
<tr>
<th>GROUP</th>
<th>SPECIES</th>
</tr>
</thead>
</table>
| Group C | Lophostemon lactifluus  
|        | Lophostemon suaveolens  
|        | Lophostemon grandiflorus |
| Group D | Terminalia platypylla  
|        | Terminalia ferdinandiana  
|        | Terminalia microcarpa |
| Group E | Bambusa arnhemica  
|        | Barringtonia acutangula  
|        | Carallia brachiata  
|        | Phragmites karka  
|        | Typha domingensis |
| Group F | Casuarina cunninghamiana  
|        | Cathormion umbellatum  
|        | Corymbia bella  
|        | Ficus coronilata/Ficus coronulata  
|        | Lysiphyllum cunninghamii  
|        | Nauclea orientalis  
|        | Pandanus spiralis  
|        | Pandanus aquaticus  
|        | Planchnia careya  
|        | Strychnos lucida  
|        | Xanthostemon eucalyptoides |
Figure 3-9 Distribution of *Acacia* spp. in the Fitzroy catchment
Figure 3-10 Distribution of *Eucalyptus* spp. in the Fitzroy catchment
Figure 3-11 Distribution of *Ficus* spp. in the Fitzroy catchment
Figure 3-12 Distribution of *Melaleuca* spp. in the Fitzroy catchment
Figure 3-13 Distribution of Group A vegetation species in the Fitzroy catchment
Figure 3-14 Distribution of Group B vegetation species in the Fitzroy catchment
Figure 3-15 Distribution of Group C vegetation species in the Fitzroy catchment
Figure 3-16 Distribution of Group D vegetation species in the Fitzroy catchment
Figure 3-17 Distribution of Group E vegetation species in the Fitzroy catchment
Figure 3-18 Distribution of Group F vegetation species in the Fitzroy catchment
Figure 3-19 Distribution of *Acacia* spp. in the Darwin catchments
Figure 3-20 Distribution of *Eucalyptus* spp. in the Darwin catchments
Figure 3-21 Distribution of *Ficus* spp. in the Darwin catchments
Figure 3-22 Distribution of *Melaleuca* spp. in the Darwin catchments
Figure 3-23 Distribution of Group A vegetation species in the Darwin catchments
Figure 3-24 Distribution of Group B vegetation species in the Darwin catchments
Figure 3-25 Distribution of Group C vegetation species in the Darwin catchments
Figure 3-26 Distribution of Group D vegetation species in the Darwin catchments
Figure 3-27 Distribution of Group E vegetation species in the Darwin catchments
Figure 3-28 Distribution of Group F vegetation species in the Darwin catchments
Figure 3-29 Distribution of *Acacia* spp. in the Mitchell catchment
Figure 3-30 Distribution of *Eucalyptus* spp. in the Mitchell catchment
Figure 3-31 Distribution of *Ficus* spp. in the Mitchell catchment
Figure 3-32 Distribution of *Melaleuca* spp. in the Mitchell catchment
Figure 3-33 Distribution of Group A vegetation species in the Mitchell catchment
Figure 3-34 Distribution of Group B vegetation species in the Mitchell catchment
Figure 3-35 Distribution of Group C vegetation species in the Mitchell catchment
Figure 3-36 Distribution of Group D vegetation species in the Mitchell catchment
Figure 3-37 Distribution of Group E vegetation species in the Mitchell catchment
Riparian vegetation and the physical environment

The timing and quantity of water available to the riparian zone is critical for determining its structure, function and resilience, such that all aspects of the flow regime (i.e. precipitation, runoff and evapotranspiration) exert some control over how riparian vegetation access water (Tabacchi et al., 1998). There are three potential sources of water used by riparian vegetation: access in the upper non-saturated profile (soil water) derived from rainfall events, access at depth (groundwater) and access in the riverbank (O’Grady et al., 2006).

Riparian zones both structure and are structured by their physical environment. Complex interactions between hydrology, geomorphology, light, temperature and fire influence the structure, dynamics and composition of riparian zones (Naiman and Decamps, 1997; Tabacchi et al., 1998). The width of the riparian zone, the level of control that the streambed vegetation has on the stream environment, and the diversity of functional attributes (e.g. instream flow, biogeochemical cycles) are related to the size of the stream, its position within the drainage network, the hydrological regime and the local geomorphology Naiman and Decamps (1997).
Most riparian zones in northern Australia are relatively narrow, with perennial flowing rivers of the study area generally having banks that consist of a series of benches below the floodplain created by high wet-season flows, as well as areas of groundwater seepage (Erskine et al., 2003). As these banks increase in elevation from the low-flow level to that of the surrounding savanna, access to water by trees and the species present change. A study in the Daly River catchment of the Northern Territory found bank slopes were dominated by six species with Melaleuca trees, which are tolerant of high energy flooding, found at elevations less than 5 m above the river. These trees were mostly using groundwater (O’Grady et al., 2006), which accounts for 50% of riparian-zone dry-season water requirement (Lamontagne et al., 2005). Moving upslope, there is less reliance on groundwater, with other tree species in that study more opportunistic rather than obligate on groundwater dependence. Two species, A. auriculiformis and C. cunninghamiana, occurred across the riparian gradient and highlighted the opportunistic nature of groundwater use by trees in the riparian zone. At low elevations, both species used groundwater, but at higher elevations in the landscape both species used soil water. Water use was highest in the wet season reflecting water availability and the prevalence of deciduous and semi-deciduous species (O’Grady et al., 2006).

Riparian vegetation also structures the environment, by stabilising river banks and influencing the channel structure. At a basin scale, the underlying geology, geomorphic history and discharge of sediment influences the structure of the floodplains (Tabacchi et al., 1998). Riparian vegetation modifies sediment transport either by physically entrapping materials, which appears to be most important in relatively low-gradient environments, or by altering channel hydraulics (Naiman and Decamps, 1997).

In northern Australia, trees at sufficient densities from the genus Melaleuca can have a significant influence on dryland channel process, form and adjustment by moderating erosion and depositional processes (Tooth and Nanson, 2000). The hydraulics of the entire channel can also be modified drastically during both low and high flows as a result of trees invading sand and gravel bars. Such invasions can occur in response to changes in climate conditions or following man-made changes to river regimes (Tabacchi et al., 2000).

Sediment retention is highly correlated with backwater flow patterns and vegetation cover, which can also increase water residence times for drainage (Naiman and Decamps, 1997; Tabacchi et al., 2000). This promotes exchange between surface and groundwater, although fragmentation of riparian vegetation can increase the intensity of the river-to-floodplain pulse (Tabacchi et al., 2000). Riparian vegetation also acts as a source of woody debris to a stream. This debris can cause water-flow diversions, channel avulsions, congestions in the main channel, reduced connectivity between the main channel and secondary water bodies and enhanced local erosion (Tabacchi et al., 2000). Wood is an important element of aquatic habitat structure and provides diversity to stream habitats, as flow-mediated scour can create localised pools in otherwise shallow run habitats (Pusey and Kennard, 2009). Microalgae is also deposited on instream wood structures, and is an important food source for many fish species, such as the bony herring.
Biogeochemical processes within the riparian zone and the river channel are influenced by basin hydrological (the water cycle of evaporation and precipitation) and hydraulic (water under pressure) processes (Naiman and Decamps, 1997). Fine sediments that are mobilised by basin runoff have higher concentrations of easily moved nutrients, the removal of which occurs as a consequence of the interactive processes of deposition and erosion, infiltration, dilution, and adsorption or desorption reactions with forest soil and litter (Naiman and Decamps, 1997). Being highly productive, riparian vegetation takes up nutrients from soil and groundwater. Stored nutrients are released subsequently during decomposition, releasing organics (Naiman and Decamps, 1997). Coarse and dissolved organic matter contribute to streams as available sources of energy to support productivity and food webs. Organic matter from riparian vegetation is a source of nourishment for aquatic organisms.

A large proportion of riparian species are capable of vegetative resprouting, where roots and trunks broken during floods resprout, allowing these species to quickly dominate the canopy after a flood (Pettit et al., 2016b). Riparian vegetation may also be disturbed by floods and re-establish in new locations, but ultimately reproduction, dispersal of propagules and age structure of riparian trees is dependent on the natural flood regime (Pettit and Froend, 2001). Seedling establishment, phenology, and the composition and structure of the community reflect the duration of flooding (Pettit et al., 2017).

Seed fall is a key process in seed dispersal. Seed fall needs to coincide with seasonal changes in river levels, such that the timing of seed production and seed fall coincides with the favourable conditions for recruitment, for example when floodwaters are receding and fresh soil is exposed (Pettit and Froend, 2001). In monsoonal areas of tropical northern Australia, many species of eucalypts have short reproductive cycles, with flowering occurring at the end of the dry season (Brooker and Kleinig 1994) and seed release at the end of the wet season (Pettit and Froend, 2001), while the native bamboo (*B. arnhemica*) can grow for decades before flowering and the death of the mature stand (Franklin, 2004). Changes in timing of flows and sequencing of flows can impact on establishment of seedlings.

Conceptual models capturing the cycle of recruitment and the relationship to hydrology have been proposed for the Ord River in WA (Figure 3-39) (Pettit and Froend, 2001).
Riparian zones and biota

Riparian plant communities are composed of specialised and disturbance-adapted species (Naiman and Decamps, 1997). Riparian vegetation typically disperses seeds in phase with the seasonal retreat of floodwaters, ensuring moist seedbeds for successful germination and plant colonisation (Naiman and Decamps, 1997). Many species disperse seed via flowing water, and some also have vegetative reproduction. Riparian vegetation of the wet–dry tropics in Australia are comprised of specifically adapted flora not usually found in the surrounding savanna country. The riparian zones of northern Australian rivers are unusually narrow compared to elsewhere in Australia, because of the intense water stress that develops in terrestrial areas during the dry season.

As networks distributed over large areas, riparian zones are key landscape components in maintaining biological connections along extended and dynamic environmental gradients (Naiman and Decamps, 1997). They provide habitat for fish and macroinvertebrates within the stream channel and afford protection for small mammals and birds on land (Naiman and Decamps, 1997). Despite being characterised by disturbance within landscapes, riparian zones provide relative resource stability for biota, particularly in arid and semi-arid landscapes (Woinarski, 2004).

The riparian zones of northern Australian support a rich diversity, particularly in bird fauna (Pusey and Kennard, 2009; Woinarski et al., 2000). This is well pronounced in areas with extensive cover of rainforest plants and *Melaleuca* (Woinarski et al., 2000). Riparian zones provide critical habitat and resources for birds in terms of feeding, refuge, watering, nesting and rearing (Pusey and Kennard, 2009). Riparian zones support a high abundance of insects.
that provide food for a variety of animals (Pusey and Kennard, 2009). Riparian zones also provide food for aquatic animals.

**Threats to riparian vegetation**

Riparian zones are characterised by heterogeneity promoted through flood processes, which results in a high vulnerability to disturbance (Naiman and Decamps, 1997). The ecological integrity of riparian zones across northern Australia is threatened by a range of existing processes, including clearing, weed invasions and disturbance by livestock (Woinarski et al., 2000). The deterioration of riparian habitats has been cited as a principal reason for local and regional declines in the northern Australian endemic riparian specialists, the buff-sided robin (*Poecilodryas cerviniventris*) and the purple-crowned fairy-wren (*Malurus coronatus*) (Woinarski et al., 2000).

The riparian zones of northern Australia have evolved with periods of high flow and may change if such flows are not appropriately managed (Pettit et al., 2016b). Lack of large floods can limit seed dispersal, while regulated systems support some species but not others. Downstream of the Ord River Dam in WA, major physical and ecological changes have occurred, including the growth of dense thickets of vegetation along the river’s margins. This has led to a consequent narrowing of the river channel (Cluett and Radford, 2003) and changes in species composition of the riparian vegetation because of a reduction in the flood disturbance regime (Pettit and Froend, 2001).

Land clearing from catchment development can result in large-scale vegetation clearing, which can change runoff characteristics and reduce groundwater recharge (Tooth and Nanson, 2000). This, along with groundwater pumping, can reduce groundwater levels. Reduction in groundwater levels may directly affect riparian vegetation during the dry season, and may make vegetation more fire prone as a result (Erskine et al., 2003). Stream banks largely devoid of riparian vegetation are often highly unstable, and bank erosion is more prevalent on non-vegetated banks exposed to currents than on vegetated banks (Naiman and Decamps, 1997).

Riparian zones (including floodplains and wetlands) in arid and semi-arid zones can experience high salinity as a consequence of the highly evaporative conditions and the variability of inflows, which provide dilution and flushing of the stored salt (Jolly et al., 2008). Many arid and semi-arid environments are experiencing extended periods of high salinity as a consequence of catchment development, with salts being mobilised in areas of rising groundwater tables. This can be due to intensive irrigation water application or vegetation clearing. Salinity can exert ecological impacts in a wide range of other, indirect or non-lethal ways, altering productivity, species distributions and community interactions (Herbert et al., 2015).

The challenge for managing river flows and protecting riparian zones will be to try and replicate the natural flow regimes (Pettit et al., 2016b), with management of groundwater and fire likely to be key to regulating the riparian boundary (Petty and Douglas, 2010).
Fire

Fire is very common in the dry season across the vast savanna landscapes of northern Australia. The impact of fire is influenced by climatic conditions (wind, relative humidity and temperature) and landscape factors, such as the quality and quantity of the fuel load and slope. These factors, in turn, define the physical properties of the fire, such as the rate of spread and height of flames. Fire typically enters the riparian zone from the surrounding uplands, with penetration patchy or absent due to the riparian zone characteristics reducing the conditions suitable for fire. These include greater canopy cover and reduced evaporation maintaining higher relative humidity, increased moisture of the litter layer, and reduced air temperatures (Douglas et al., 2016). Riparian zones can therefore act as a fire barrier. The effects of fire are usually greatest in the smaller zones of headwaters where the conditions are reduced and more similar to the surrounding savanna. Therefore, impacts decrease with channel size.

Flooding can interact with fire by depositing large amounts of accumulated woody fuel (Pettit and Naiman, 2006). It can also open up areas in which fire tolerant grassy species grow, creating conditions for bigger fires (Douglas et al., 2003). These fires subsequently remove vegetation and can therefore increase erosion. If fire occurs after the critical germination periods that follow after flooding, young individuals will be killed and lost from the system, potentially altering stand composition. Fire can also reduce flowering, seeding and fruit bearing depending on its timing.

Many of the tree species growing in riparian zones are sensitive to fire, especially thin-barked trees and vines. Fire will often kill small shrubs and young plants that dominate the riparian zone, and if they do not have the ability to resprout after fire like many of the savanna species, they will be lost and rely on recruitment. Not only will individual fires affect the plant species, but the fire regime – which includes the frequency, timing, intensity, spatial scale and pattern of repeated fires – has the ability to dramatically alter the plant species structure and floristic composition, especially when fire occurs late in the dry season (Andersen et al., 2005). The same authors found that unburned riparian zones had more species in common with rainforest patches than savanna species.

Climate change also has the potential to change the flow regime of the riparian zone, which could dry it out and leave it more vulnerable to fire (Tabacchi et al., 1998).

Weeds

Riparian zones, especially those affected by disturbances, such as flooding, fire and grazing, can act as key pathways for the spread of invasive weeds (e.g. Gamba grass, (Petty et al., 2012)). Gamba grass and other grasses (mission grass, Pennisetum polystachion; molasses grass, Melinis minutiflora; and guinea grass, Megathyrsus maximus) can provide increased fuel loads, resulting in more damaging fires. Woody weeds, such as mimosa, can act to dominate the riparian zone with a single species and exclude fire. Other woody weeds, such as rubber vine, can choke riparian zones and reduce species diversity (Grice, 1997).
Grazing

Australia has no long-term history of grazing by ungulates, and yet introduced species pose a major threat to riparian systems through hoof trampling and erosion, to the spread of invasive plants through disturbance (Jones and Vesk, 2016). Northern Australia is also affected by undulate grazers, such as feral buffalo, donkeys, horses, pigs and cattle. With less than 1% of northern Australia irrigated, the creeks and rivers provide good forage in the annual dry season and produce disproportionate use of riparian zones. The less profitable subtropical and semi-arid regions of northern Australia have also seen low-density expansive grazing, with recent increases in herd size and a growing export market. Grazing and trampling over vast areas, and particularly the riparian zones, can alter nutrient availability (N) and affect sensitive microbes and macroinvertebrates, leading to negative feedback into infiltration, runoff and water relations.

Conceptual model

The conceptual model for riparian vegetation is shown in Figure 3-40, with the accompanying narrative given in Table 3-6.

![Figure 3-40 Conceptual model showing the relationship between threats, drivers, effects and outcomes for riparian vegetation in northern Australia](image)

This example considers understanding outcomes associated with changes in hydrology (blue pathways).
### Conceptual model narrative

**Table 3-6 Knowledge underpinning the conceptual model of riparian vegetation in northern Australia**

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irrigation infrastructure</td>
<td>Change in seasonal inundation and flooding</td>
<td>Instream irrigation infrastructure can change the flow regime, by altering the timing, duration and height of flow peaks. This can alter the flooding regime of riparian zones. Offstream irrigation infrastructure can change the runoff processes and alter hydrological conductivity.</td>
<td>Change in flooding regime can impact on the maintenance and recruitment of vegetation that have a requirement for surface water to fulfil their life history. Prolonged inundation/intensive irrigation can cause salinity. Saline groundwater occurs in many arid and semi-arid areas.</td>
<td>Reduced maintenance and recruitment opportunities for vegetation Reduced/altered diversity of riparian zones Change in instream diversity of habitats Change in biogeochemical processes Changes instream structure</td>
<td>Water resource development changes riparian zones. Physical and ecological changes that can occur include the growth of dense thickets of vegetation along the river’s margins, with a consequent narrowing of the river channel (Cluett and Radford, 2003), and changes in species composition of the riparian vegetation because of a reduction in the flood disturbance regime (Pettit and Froend, 2001). Many arid and semi-arid environments are experiencing extended periods of high salinity as a consequence of catchment development, with salts being mobilised in areas of rising groundwater tables. This can be due to intensive irrigation water application or due to vegetation clearing (Herbert et al., 2015). Salinity can exert ecological impacts in a wide range of other, indirect or non-lethal, ways, altering productivity, species distributions, and community interactions (Herbert et al., 2015).</td>
</tr>
<tr>
<td>Change in groundwater (GW) height</td>
<td></td>
<td>Groundwater pumping lowers watertables and can isolate vegetation roots, which could otherwise use groundwater as a water source.</td>
<td>Decline in groundwater tables, below roots of trees in the riparian zone can cause a decline in tree health and potential death. Reduced groundwater in the dry season can also lead to vegetation being more vulnerable to fire.</td>
<td>Decline in the condition of vegetation, tree death.</td>
<td>Riparian zones act as the vegetated interface between terrestrial and aquatic ecosystems which are dependent on surface and groundwater flow regimes, and are vulnerable to disturbance (Woinarski et al., 2000). The ability of soils and sediments to hold water and the existence of tributary and groundwater flows are equally important in determining vegetative distribution (Naiman and Decamps, 1997). Reduction in groundwater levels may directly impact on riparian vegetation during the dry season, and it may make vegetation more fire prone as a result (Erskine et al., 2003).</td>
</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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</tr>
<tr>
<td>Change/Intensification in land use</td>
<td>Vegetation clearing/loss</td>
<td>Intensification of agriculture can lead to direct clearing of vegetation, including in riparian zones.</td>
<td>Land clearing can have a direct impact on runoff characteristics and can therefore alter recharge. Land clearing can also impact on the function of the riparian zone as a productive part of the landscape that supports native biodiversity, both aquatic and terrestrial. Land clearing can also cause changes in water quality, through increased input of sediments to streams and potentially nutrients, through change of biogeochemical processes. Land clearing in the riparian zone can also change the structure of the channel, with potential for erosion.</td>
<td>Altered hydrology, biodiversity, water quality and stream-side stability.</td>
<td>Land clearing from catchment development can result in large-scale vegetation clearing, which can change runoff characteristics and reduce groundwater recharge (Tooth and Nanson, 2000). Being highly productive, riparian vegetation uptakes nutrients from soil and groundwater stored nutrients are released subsequently during decomposition releasing organics (Naiman and Decamps, 1997). Riparian vegetation acts as a thermal buffer to streams. This has an influence on instream primary production, nutrient interception, storage and release, enhancement of bank stability, the provision of coarse woody material as habitat and substrate for fish, invertebrates and microalgae, channel morphology and habitat diversity (Pusey and Arthington, 2003).</td>
</tr>
<tr>
<td>Climate change</td>
<td>Change in fire regime — intensity, frequency</td>
<td>Fire is uncommon in riparian zones given that these areas are typically wetter parts of the landscape.</td>
<td>Fire has the potential to cause a decline in maintenance and recruitment of riparian vegetation.</td>
<td>Decline or loss of vegetation and changes in biodiversity, water quality and stream-side stability.</td>
<td>Climate change has the potential to change the flow regime to the riparian zone, which could dry it out and leave it more vulnerable to fire (Tabacchi et al., 1998).</td>
</tr>
<tr>
<td>Invasive species</td>
<td>Competition — resources, including space Herbivory, habitat disturbance</td>
<td>The introduction of invasives, such as weeds or feral animals, such as pigs, can change the character and function of riparian zones.</td>
<td>Invasives can change the character and function of riparian zones. Disturbed landscapes can result in excessive growth of generalists, resulting in weed invasions.</td>
<td>Impact on recruitment of vegetation through being outcompeted for space or through herbivory. Decline and loss of vegetation recruitment and maintenance, resulting in a loss of native diversity.</td>
<td>Riparian corridors connect a stream to its valley and more broadly within a basin and at a local scale by the geomorphic and vegetation mosaics (Tabacchi et al., 1998). Despite strong hydrological and geographic influences on riparian vegetation, ecological influences (such as competition, herbivory, soils, and disease) are significant in shaping communities (Naiman and Decamps, 1997).</td>
</tr>
</tbody>
</table>
3.1.4 MONSOON RAINFOREST

Asset description

The monsoon rainforest patches of northern Australia (also known as monsoonal vine forests) are an example of groundwater-dependent forests that rely on year-round water availability. They are associated with areas in the landscape where water is trapped or the watertable is close to the surface. They occur around the coastline, springs, watercourses and deep rocky gullies, or areas that are naturally fire protected by topographic features such as steeper, rockier terrain (dry monsoonal forests comprising deciduous species) and especially in sheltered gorges or the moist substrate (Bowman, 2000).

There is considerable variation in rainforest patches, including vine forests on sandstone substrates, vine forests and thickets on rocky outcrops, and vine forests on lowland plains (especially fringing rivers and wetlands) to coastal vine forests and thickets (Russell-Smith and Stanton, 2002). The latter form a threatened community on the swales and sand dunes of the Dampier Peninsula, WA (Belfield and Beames, 2010). This asset description focuses on wet rainforest patches of tall trees located on the flat lowland sites with year-round soil water and canopy cover of 90 to 95% (Bach, 2002), rather than the dry thickets of predominantly deciduous species on seasonally dry sites or thickets growing on coastal sandy soils. The wet sites reflect other habitats described in this section and are most likely influenced by changes in water management.

These moist, heavily shaded habitats, and the concentration of fleshy fruit-bearing plant species, supports a variety of animals dependent upon these forests. This includes fruit-eating birds, such as the pied imperial pigeon (*Ducula bicolor*), rose-crowned fruit dove (*Ptilinopus regina*), figbird (*Sphecotheres viridis*), yellow orioles (*Oriolus flavicintus*), common koel (*Eudynamys scolopacea*) and great bowerbird (*Chlamydera nuchalis*) (Price et al., 1999), and bats (black flying fox, *Pteropus alecto*). These birds and bats move between monsoonal forest patches and require many patches over a large area to maintain their populations (Price et al., 1999). Woinarski (1993) also found that bird species from eucalypt open forests, wet tropical rainforests, mangroves, paperbark woodlands and generalist species all occupy the rainforest patches of Kakadu National Park. It is suggested that the distance between patches may be the reason that there is only one mammal species, the Cape York rat (*Rattus leucopus*), with exclusive dependence on these habitats in Cape York (Bowman and Woinarski, 1996). Other threatened species, such as the northern quoll (*Dasyurus hallucatus*), golden-backed tree-rat (*Mesembriomys macrurus*) and Carpentarian rock-rat (*Zyzomys palatalis*), also use this habitat (Australian Rainforest Foundation, 2011). For these reasons this habitat is considered as an important asset in the Darwin catchments.

Distribution in Assessment study areas

When looking at satellite imagery, the distinct, thick canopies of the trees growing in the monsoon vine forests stand out from the sparsely wooded, eucalypt savanna that covers much of northern Australia. While some vegetation associated with river edges and springs
is found in the Fitzroy and Mitchell River catchments, this habitat is most common in the Darwin catchments and can be found from Melville Island to Tennant Creek (Russell-Smith, 2001). Although there is 2700 km² of this habitat in the Northern Territory, it only accounts for 0.2% of the Northern Territory, and is made up of 15,000 patches ranging in size from 1 to 4000 ha with a median size of 3.6 ha (Northern Territory Department of Land Resource Management, n.d; Price et al., 1999). Many of these patches contain very small populations of mostly less than 50 plants (Russell-Smith and Lee, 1992). The majority of these patches and those with the greatest structural and floristic diversity occur in the narrow coastal and sub-coastal belt, and become more scattered inland of the Northern Territory (Russell-Smith and Stanton, 2002).

Bowman and Panton (1993) found that the rainforest patches of the Darwin area are susceptible to development, fire, weed invasion and cyclones, with a 60% reduction in cover of pre-1945 rainforest extent. Russell-Smith and Bowman (1992) also found impacts of introduced cattle and Asian water buffalo, pigs and flood damage on rainforest patches.

Figure 3-41 shows the distribution of monsoon vine forest in the Darwin catchments.
Description of habitat type and processes

A number of hypotheses have been proposed as to why these small patches exist in the extensive savannas. Bowman (1992) suggested small patches may be fragments of a formerly continuous vegetation. Studies have suggested that the contraction of this continuous vegetation to today’s distribution of small patches could be the result of the developing rainfall seasonality and aridity over the last 10 million years (Russell-Smith and Stanton, 2002). Other hypotheses include local variation in fire, soil chemistry, soil water, microclimate, soil micro-organisms and allelopathy (Bowman and Panton, 1993; Ondei et al., 2017).

Access to water is critical to the type and functioning of tropical tree stands. Even with the tropical, monsoonal wet season delivering 800 to 2000 mm rainfall, many trees must survive...
primarily on soil water (1 to 2 metres of soil, Liedloff and Cook, 2007) during the long period each year with very little or no rainfall (May to October). Areas where water is available for longer periods of the year through topographic features (e.g. deep rocky gullies) or where the watertable comes close to the surface (e.g. springs) support a different suite of tree species and associated fauna from the majority of tropical savannas, and therefore provide important refuges.

As plant roots can access groundwater throughout an otherwise long, rain-free dry season, these forests support dense stands of tall trees, palms and vine thickets with greater numbers of stems, higher litter cover and lower grass abundance than the surrounding fire-dominated savanna landscape. They vary from tall, closed forests to deciduous thickets of less than 2 m tall (Tropical Savannas CRC) with plant species richness and number of trees per patch relatively low and closely related to rainforests of eastern Australia (Russell-Smith and Lee, 1992).

The structure of monsoon rainforests with persistent green vegetation, shaded canopies and lack of a grass layer also acts as a barrier to fire, which is very common during the dry season in the surrounding dry savanna. The presence of frequent fire in the savannas drives the species composition towards fire and water-stress-tolerant species, while the monsoonal vine forests often contain fire-sensitive species less able to survive fire. Rainforest species, while producing more seedlings, lack lignotubers, have slower growth rates and some lack resprouting capacity, making them susceptible to repeated fires (Russell-Smith and Setterfield, 2006).

The small size of most of the monsoon rainforest vegetation patches in the Darwin catchments means that they are sensitive to any impacts on their edges caused by land disturbance, invasive weeds, feral animals or fire. Any process that alters the period of time that trees can access water can also influence the monsoon vine forests and reduce their long-term viability. To date, the integrity of these stands has been maintained in the landscape, despite wet-season rainfall variation and current disturbances, such as storm damage, invasive grasses and feral animals. However, this habitat is under increasing pressure and threats.

Changes to water availability beyond that naturally experienced poses a threat. A reduction in water availability during the dry season, especially towards the end of the season, would likely have consequences for the wet monsoon vine forests, resulting in stress or even death of species requiring continuous access to soil water. This is especially true of the individuals towards the edge of the forest, where soil conditions are similar to the surrounding savanna. The resulting loss of trees opens up the rainforest patch, altering the microclimate, allowing for soil water evaporation and allowing the colonisation of grasses, therefore increasing the chances of fire in this habitat. Therefore, the current water relations support trees and maintain a fire-excluded habitat.

For water resource management to negatively affect monsoon rainforests, the scale of change would need to alter the duration of spring flow or the height of watertables that may naturally be close to the surface (<5 m) where these forests are supported. To assess
future impacts on monsoon forests, changes in watertable height over time and future spring flow rates and duration need to be predicted, taking into account both water management and predicted climate change.

**Conceptual model**

Figure 3-42 shows the conceptual model for the monsoon vine forest asset, with the accompanying narrative given in Table 3-7.

![Conceptual model](image)

**Figure 3-42** Conceptual model showing the relationship between threats, drivers, effects and outcomes for the monsoon vine forest asset in northern Australia

This example considers understanding outcomes associated with changes in hydrology (blue pathways).
### Conceptual model narrative

**Table 3-7 Knowledge underpinning the conceptual model of monsoon rainforests in northern Australia**

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4 EFFECT</th>
<th>5 OUTCOME</th>
<th>6 EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extraction/ regulation</td>
<td>Lowering of watertable</td>
<td>Monsoon vine forest relies on soil water or shallow watertable access during dry season</td>
<td>If the watertable drops below a threshold the monsoon forest will become stressed. Extended periods may lead to reduced fruiting and even mortality of trees.</td>
<td>The death of mature evergreen trees in the forest opens the canopy and allows grass growth potentially leading to fire.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Reduced spring flow</td>
<td>Springs may keep surrounding areas moist</td>
<td>Moist or green vegetation excludes fire if this vegetation dries out early in the dry season the monsoon vine forest edges are exposed to fire.</td>
<td>Loss of trees on edge of forest. Encroachment of grasses into forest.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Increased seasonal drought</td>
<td>Springs may keep surrounding areas moist</td>
<td>Reduced growth rate of young trees exposes them to fire.</td>
<td>Inability of stand to increase/replace gaps at edge. Russell-Smith (2001).</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Loss of critical conditions</td>
<td>Monsoon savanna ecotone</td>
<td>Changes to the ecotone will alter microclimate and soil conditions (mycorrhizas, soil fertility and soil water) critical for rainforest seedling establishment.</td>
<td>Explains why rainforest species have not be found in fire protected savanna, and rainforest has difficulty spreading. Bowman and Panton (1993).</td>
<td></td>
</tr>
<tr>
<td>Recruitment</td>
<td>Access to pollination and seed dispersal</td>
<td>Bird/bat species critical for pollination and seed dispersal</td>
<td>These species require large areas of rainforest with critical distances between patches and a sequence of flowering and fruiting species.</td>
<td>Reduced seed dispersal and pollination. Negative feedback into rainforest health and maintenance. Price et al. (1999).</td>
<td></td>
</tr>
<tr>
<td>Fire</td>
<td>Late dry season, relatively intense fires increase with land management</td>
<td>Monsoon savanna ecotone</td>
<td>Destructive to rainforest patches especially at the edges.</td>
<td>Reduce size of patches and expose them to other disturbances. Russell-Smith and Bowman (1992).</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Frequent late dry-season fires increase under land management</td>
<td>Monsoon savanna ecotone</td>
<td>Rapidly erode rainforest edges. Low-intensity fires have limited effect due to species regenerative abilities.</td>
<td>Rainforest reduced in size with very slow recovery. Russell-Smith and Bowman (1992). Banfai and Bowman (2006).</td>
<td></td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/STRESSOR</td>
<td>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4 EFFECT</td>
<td>5 OUTCOME</td>
<td>6 EVIDENCE AND PATHWAYS</td>
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<tr>
<td><strong>Feral animals</strong></td>
<td>Pig, buffalo, cattle damage</td>
<td>Rainforest fine surface roots</td>
<td>Trampling and digging damages fine surface roots, compaction of ground, damage to trees</td>
<td>Reduced rainforest health leading to changes in microclimate and other invasion of weeds and fire</td>
<td>Russell-Smith and Bowman (1992) Bowman and McDonough (1991) limited pig effect</td>
</tr>
<tr>
<td><strong>Climate change</strong></td>
<td>Increasing moisture or CO₂ enrichment</td>
<td>Rainforest</td>
<td>Increase in monsoon rainforest extent</td>
<td>Monsoon rainforest patches found to expand</td>
<td>Bowman et al. (2010) Banfai and Bowman (2006)</td>
</tr>
</tbody>
</table>
Predicted implications of management

As this habitat generally occurs at groundwater discharge locations where the watertable does not fluctuate greatly during the year, it is less susceptible to lowering watertable levels than the surrounding savanna on recharge areas where the watertable can drop to 10 to 15 m below the surface (Turnadge et al., 2018). Turnadge et al. (2018) found the watertable depth around discharge points may drop to 3 m below the surface during the late dry season, which is probably still readily available to the roots of monsoon forest tree species.

The only threat may be that the total recharge of the catchment is reduced through water management, but given the Wildman catchment has a 3 GL balance (Turnadge et al., 2018), this is also unlikely.

3.2 Category: Species of significance

3.2.1 MAGPIE GOOSE

Asset description

The magpie goose is native to Australia and New Guinea and has an estimated population size of 3.5 million in the Northern Territory alone (Traill et al., 2009). Once prevalent throughout south-east Australia, the Australian distribution of the magpie goose has largely contracted to northern Australia in association with a range of synergistic threatening processes, including habitat loss, invasive species and drought (Bayliss and Yeomans, 1990; Colley, 1999; Traill and Brook, 2011). The magpie goose occurs within sub-coastal wetlands across the tropical north of Australia, where during the wet season, they make nests from emergent vegetation including sedges and grasses (Colley, 1999). During the dry season, magpie geese gather on floodplains in large numbers and feed on the tubers of the sedge *Eleocharis dulcis* (Traill and Brook, 2011). Magpie geese move between river systems in response to seasonal rainfall to find breeding and foraging sites (Traill et al., 2010; Wilson, 1997). Studies in the Northern Territory indicate that long-distance movements by magpie geese are rare, with most movements occurring only locally (Traill et al., 2010; Wilson, 1997).

The timing of rainfall and monsoonal flooding provide cues for nesting and hatching in magpie geese (Warfe et al., 2011). Research has shown that the best habitat quality is the result of a complex combination of features that provide suitable nesting sites and opportunity. These include availability of food resources, materials for construction of nests, protection from predators and flood resilience (Colley, 1999). Monsoon rainfall and flooding occurring in the early wet season increases macrophyte growth, providing habitat and foraging resources (Traill and Brook, 2011). Lower flood peaks and changes in flood timing are associated with reduced breeding success, recruitment and survival (Warfe et al., 2011; Whitehead and Saalfeld, 2000). Feeding on *Eleocharis* spp. is most preferential when water levels are between 10 to 20 cm (Wilson, 1997) and before the ground dries and hardens.
Hence, wetlands retaining water are an important dry-season refuge for magpie geese (Bayliss and Yeomans, 1990), which often aggregate in large numbers at important sites during the dry season when resources are otherwise more limited. Loss of dry-season wetland inundation results in reduced body condition and lower survival for magpie geese (Traill and Brook, 2011). The species is considered relatively well understood and is economically and culturally important to both Indigenous and non-Indigenous Australians (Traill et al., 2010).

**Distribution in Assessment study areas**

The distribution of the magpie goose extends across northern Australia throughout all three Assessment study areas (Figure 3-43, Figure 3-44 and Figure 3-45). Historically, the population was prevalent in south-eastern Australia, but is now classed as near-threatened in Victoria, vulnerable in NSW and extinct or endangered in SA. The species also has populations within New Guinea. Within the Assessment study areas, the Darwin catchments contains the most significant populations of the magpie goose.
Figure 3-44 Distribution of magpie goose (Anseranas semipalmata) in the Darwin catchments
Figure 3-45 Distribution of magpie goose (Anseranas semipalmata) in the Mitchell catchment.
Conceptual model

Figure 3-46 shows the conceptual model for the magpie goose, with the accompanying narrative given in Table 3-8.

Figure 3-46 Conceptual model showing the relationship between threats, drivers, effects and outcomes for the magpie goose in northern Australia
### Conceptual model narrative

#### Table 3-8 Knowledge underpinning the conceptual model of the magpie goose in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/ STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floodplain wetlands</td>
<td>Changes to wet season/ inundation/ flooding regime</td>
<td>Timing of monsoonal rainfall and flooding important in maintaining ecological processes</td>
<td>Alteration of flood dynamics can disrupt cues for nesting and hatching in magpie geese (Warfe et al., 2011). Early onset monsoon associated with strong macrophyte growth providing habitat and foraging resources. Period of floodplain inundation is important in influencing the survival of the Magpie goose food source <em>E. dulcis</em> (Traill and Brook, 2011). Fecundity and dry-season survival likely associated with rainfall (Bayliss and Yeomans, 1990).</td>
<td>Reduction in either rainfall or flood peaks, or changes in flood timing leading to reduced breeding success, survival and population size of magpie geese (Warfe et al., 2011; Whitehead and Saalfeld, 2000).</td>
<td>Magpie goose population size demonstrates trends closely coupled with trends in rainfall and river flows (Warfe et al., 2011; Wilson, 1997). Magpie goose nest density and timing are strongly correlated with time of wet season, with nesting almost completely failing in unfavourable (El Nino) years (Whitehead and Saalfeld, 2000). Clutch sizes increase in years with early wet-season onset (Whitehead and Saalfeld, 2000). Correlation between population increase and rainfall at intermediate levels indicate relationship with habitat and food (Bayliss and Yeomans, 1990).</td>
</tr>
<tr>
<td>Changes to dry-season water persistence</td>
<td>During the transition from the wet to dry, floodplain water contracts to waterholes These provide important refuge habitat for magpie geese (Warfe et al., 2011)</td>
<td>Reduction in persistence of ephemeral aquatic habitat increases pressure on magpie goose populations.</td>
<td>Magpie geese migrate to areas with quality resources, becoming highly dense at locations with suitable habitat characteristics (Warfe et al., 2011). Loss of suitable early dry-season habitat increases competition at suitable sites resulting in reduced body condition, important for survival during the dry season (Traill and Brook, 2011).</td>
<td>Magpie geese comprised about 77% of the total waterbird biomass in Magela Creek floodplains during the dry season (Pettit et al., 2011). Magpie geese often aggregate on floodplains during the dry season and disperse to other habitats during the wet season (Pettit et al., 2011). Different wetland/river systems become important under varied conditions; the wetland</td>
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<tr>
<td>HABITAT AND THREATS</td>
<td>DRIVER/STRESSOR</td>
<td>SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>EFFECT</td>
<td>OUTCOME</td>
<td>EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Saltwater intrusion</td>
<td>Require freshwater vegetation for foraging and nesting (Catford et al., 2013; Traill et al., 2009)</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion kills salt-intolerant vegetation and transitions the community to salt-tolerant species (Catford et al., 2013; Traill et al., 2010). Floodplain water quality (including salinity) likely to be important in influencing the survival of the Magpie goose key food source <em>E. dulcis</em> (Traill and Brook, 2011).</td>
<td>Loss of salt-intolerant vegetation leads to population decline and changed distribution (Catford et al., 2013).</td>
<td>Loss of salt-intolerant vegetation leads to population decline and changed distribution (Catford et al., 2013).</td>
<td>East Alligator River has already experienced tidal intrusion up to 4 km inland, resulting in a nine-fold increase in the area of saline mudflats and a loss of salt-intolerant semi-aquatic vegetation, including 64% of the original Melaleuca forest (Catford et al., 2013). Modelling has shown that a loss of just 5% of the wetland habitat currently available to magpie geese could reduce populations to just a few thousand individuals and possibly initiate an ecological bottleneck or ‘point of-no-return’ (Catford et al., 2013).</td>
</tr>
</tbody>
</table>

Magpie geese forage on *E. dulcis* during the early tropical dry. Magpie geese gain body condition and accrue reserves to survive the late dry season (Traill and Brook, 2011).

Mosaic is considered important (Finlayson et al., 2006). Magpie geese abandoned preferred forage sites when tuber density fell below $\sim 4$ m$^{-2}$ (Traill and Brook, 2011).

Loss of floodplain inundation regimes likely impact tourism, recreational and Indigenous values (Warfe et al., 2011).

During wet periods, waterbirds can disperse throughout the available habitat. During dry periods waterbirds concentrate on the remaining more permanent water bodies (Kingsford and Norman, 2002).
<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
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<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Floodplain wetlands</td>
<td>Change in intensification in land use and management and Invasive species (e.g. flora)</td>
<td>Habitat degradation/homogenisation</td>
<td>Habitat heterogeneity important in providing resources through variation in time, and for meeting different needs of the species (Whitehead and Saalfeld, 2000)</td>
<td>Homogenisation or loss of wetlands to improvement to pasture or invasive species risks losing year-round habitat needs (Whitehead and Saalfeld, 2000).</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
</tr>
<tr>
<td>Floodplain wetlands</td>
<td>Invasive species (e.g. fauna)</td>
<td>Competition for resources and homogenisation/degradation of habitat</td>
<td>Invasive species, including the feral pig, directly compete with Magpie geese for key food resources (Pettit et al., 2011) Invasive weeds, habitat degradation by pig and buffalo (Bayliss and Yeomans, 1990; Traill et al., 2009)</td>
<td>Reduction in quality of habitat and resources required to maintain population health.</td>
<td>Reduction in population size or changes in distribution/habitat use. Magpie geese abandoned preferred forage sites when tuber density fell below $\sim 4m^{-2}$ (Traill and Brook, 2011).</td>
</tr>
<tr>
<td>Floodplain wetlands</td>
<td>Harvest; consumptive use</td>
<td>Reduced survival</td>
<td>Sufficient long-term adult survival and recruitment rates to maintain population size</td>
<td>Long-term harvest rates that exceed population growth rates, including consideration of environmental variability in growth rates, will reduce population sizes of magpie geese (Brook and Whitehead, 2005; Dillingham and Fletcher, 2008).</td>
<td>Long-term harvest rates that exceed population growth rates lead to decreases in population size and reduced long-term population viability (Brook and Whitehead, 2005). Investigation of population harvest rates with models has demonstrated mixed conclusions regarding the sustainability of the current predicted harvest rate on long-term population persistence of magpie geese (Dillingham and Fletcher, 2008). Overharvest has implications on viability of both species persistence and human utility (Brook and Whitehead, 2005).</td>
</tr>
</tbody>
</table>
3.2.2 BARRAMUNDI

Asset description

Barramundi (Lates calcarifer) is a large (>1 m) catadromous, opportunist fish (order Perciformes) that occurs throughout northern Australia, including all Assessment catchments. The species occurs throughout the west Indo–Pacific region, including rivers, lagoons, swamps and estuaries across northern Australia, from the Noosa River in Queensland to the Ashburton River in WA (Froese and Pauly, 2014). It is a voracious predator capable of modifying the estuarine fish and crustacean community in which it lives (Milton et al., 2005; Russell and Garrett, 1985; Tanimoto et al., 2012). Barramundi are found in ‘catchment to coast’ habitats including freshwater, estuarine and marine habitats at various times within their life history (Russell and Garrett, 1985). Throughout their range they are a very productive fish species that are important as a commercial, recreational and Indigenous fish resource.

Barramundi is arguably the most important fish species to commercial, recreational and Indigenous fisheries throughout wet–dry tropical Australia. Their body form and quality of flesh makes them a prized table fish, and commercial fisheries exist in coastal rivers of the Gulf of Carpentaria, Northern Territory and northwest WA (Pilbara/Kimberley region). The species makes up a substantial component of total catch in northern Australia (Savage and Hobsbawn, 2015). Catches in the Gulf of Carpentaria have been steadily increasing from around 400 t in the mid-1990s to a peak of 977 t in 2011, while the annual mean catch over the past five years is 793 t. Recent fishing effort has been relatively stable at about 10,000 days per year (Jerry et al., 2013). The species is particularly important in the south-east Gulf of Carpentaria during the annual dry season, when road access to townships is open for tourists travelling from both interstate and from within Queensland. The 2010 statewide recreational fishing survey estimated the total catch in the south-east Gulf of Carpentaria to be 124,553 fish. Assuming a mean weight of 3 kg, this equates to a total catch biomass of around 373 t. However, data from the 2010 survey indicates that 83% of barramundi in the study area are released. Interestingly, logbook catches indicated that barramundi are of little importance to the charter fishery, with catches peaking at around 2 t in the early 2000s, but the mean catch in the past five years being only 470 kg.

Barramundi are also a fish of cultural significance for the Indigenous community and an important food source (Jackson et al., 2012; Toussaint et al., 2005). Indigenous communities use the movements of barramundi as indicators of seasonal weather calendars (Green et al., 2010). All of these values are closely related to the reliance of barramundi on seasonal variation in river flows. The Indigenous catch of barramundi in the study region is less certain than other fisheries. The only available data was collected for all of Queensland as part of the 2001 National Recreational and Indigenous Fishing Survey (Henry and Lyle, 2003), which was an estimated catch of 5745 fish, or 23 t.

Barramundi typify a life-history strategy that could be significantly impacted by interruptions to the natural flows of the rivers in tropical northern Australia. The species is a
long-lived (32 years), fast-growing, protandrous hermaphrodite that often partitions habitat in the male and female form. Spawning occurs in the estuary and young male fish move into floodplain and freshwater habitats when suitable flows provide access (Russell and Garrett, 1985). Recent work has proposed three primary life-history strategies employed by barramundi (Crook et al., 2017). Some male adults return to the estuary to spawn after spending up to several years in freshwater habitats, while some males may delay downstream spawning migrations for several years until they have undergone the transition to females in freshwater habitats. Finally, some barramundi can remain completely in estuarine waters and complete their life cycle without entering freshwater systems (Crook et al., 2017). Many of these migrations are thought to be triggered by variation in the flow regime (Crook et al., 2017), making the species particularly vulnerable to water resource development.

The conditions of freshwater and estuarine systems are likely to be highly affected by the predatory behaviour of barramundi. They are an abundant top predator in the fish community, being a strong predator of a range of fish and crustacean prey in the coastal ecosystem (Milton et al., 2005). Penaeid prawns, another key group of fishery species in northern Australia, are a major prey item in the diet of barramundi while they reside in estuarine systems (Salini et al., 1998). In the north-east Gulf of Carpentaria estuary, it was the third most abundant predator species (by weight) (Blaber et al., 1989). While less abundant in embayments in the north-west Gulf of Carpentaria, the dominance of fish and commercially important crustacean prey in the diet of barramundi continued (Brewer et al., 1995). Consequently, changes to river discharge that affect barramundi movements and abundance are likely to have flow-on effects to other key assets in northern Australia.

Although data are not available by fishery, the vast majority of the catch comes from the N3 inshore gill-net fishery, while a small proportion of the catch comes from the N12/13 offshore gill-net fishery.

**Distribution in Assessment study areas**

Figure 3-47, Figure 3-48 and Figure 3-49 show the distribution of barramundi across northern Australia throughout all three Assessment catchments and marine study areas.
Figure 3-47 Distribution of barramundi (*Lates calcarifer*) in the Fitzroy catchment and marine study area
Figure 3-48 Distribution of barramundi (*Lates calcarifer*) in the Darwin catchments and marine study area
Figure 3-49 Distribution of barramundi (*Lates calcarifer*) in the Mitchell catchment and marine study area
Square symbols (dark green) are from the species density grid in Queensland WildNet (Department of Science, 2017). The smaller round symbols (light green) are for all other catches collected from other datasets (e.g. Atlas of Living Australia (2016), Northern Australia Fish Atlas (TropWATER, 2017), Jardine et al. (2012) and Pusey et al. (2017)). Larger round symbols (medium green) are 5 nm-buffered records from Queensland Department of Agriculture and Fisheries (recreational fisheries catch).

**Habitat use**

The nature of habitat use by barramundi is strongly tied to the life-history stage of each individual. Barramundi demonstrate a semi-catadromous (non-obligatory) life-history strategy. Spawning occurs in the lower estuary and adjacent coastal zone before the onset of the wet season, but can extend between September and February. Their downstream movement to these areas is possibly stimulated by rising water temperature, increasing photoperiod and first-season low flows that connect riverine waterholes and reduce salinity in the upper estuary (Robins and Ye, 2007). Larvae spend about three weeks in inshore marine waters and brackish waters to optimise their development (Robins and Ye, 2007). Although juvenile barramundi can survive as permanent estuarine residents, they thrive in semi-permanent wetlands, tidal creeks and freshwater riverine habitats (Russell and...
Garrett, 1985). Post-larvae and small juveniles attempt to access freshwater habitats adjacent to, and upstream of, the estuary when river flows allow (Halliday et al., 2012).

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 allow fish to 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). Around one year of age, individuals move out of these habitats and may move upstream into freshwater reaches (Russell and Garrett, 1988) or return to the estuary (Milton et al., 2008), where they may reside for several years.

Juvenile and adolescents remain in ephemeral or perennial freshwater habitats from months to years until flood-moderated connectivity liberates them to return to the river before emigrating downstream to the estuary and near-shore zones, often as adults, to spawn. Consequently, the annual wet season and subsequent runoff 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; Robins et al., 2005).

As the life cycle of the species is tied to ontogenetic change in habitat use (Pusey et al., 2004a), there is considerable separation of male and female barramundi (Halliday et al., 2012). Juveniles that move into fresh water are overwhelmingly male (Davis, 1987), with the transition to female occurring among larger males after they have returned to the estuary for potentially multiple spawning events (Grey, 1987), while some transitions are delayed by several years and actually occur in fresh water (Crook et al., 2017). Mature females generally do not return to freshwater reaches in Australia (Davis, 1987). Movement in barramundi appears to be cued by changes in flow, while gametogenesis is cued by increases in water temperature and photoperiod (Halliday et al., 2012).

**Conceptual model**

Figure 3-50 shows the conceptual model for the barramundi, with the accompanying narrative given in Table 3-9.
Figure 3-50 Conceptual model showing the relationship between threats, drivers, effects and outcomes for barramundi in northern Australia
### Conceptual model narrative

#### Table 3-9 Knowledge underpinning the conceptual model for barramundi in northern Australia

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
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<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
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<tbody>
<tr>
<td>Estuarine habitats</td>
<td>Reduction in seasonal inundation/flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows. Timing of monsoonal rainfall and flooding important in maintaining access to juvenile habitats (seasonally inundated freshwater habitats, supra-littoral salt flats and floodplain).</td>
<td>Reduction in flood flow volume and duration reduces the flooding of salt flats and supra-littoral habitats, as well as seasonally inundated palustrine wetlands; all habitats of early juvenile barramundi (Russell and Garrett, 1985). Interruption of emigration cues of adult barramundi from freshwater reaches to rivers to the lower estuary to spawn. Reduced growth and survival of juvenile barramundi due to inability to access fresh and brackish water habitats where insects, crustaceans and fish are consumed. Loss of brackish ecotone and the persistence of the ecotone during the post-wet season reduces the habitat of crustacean and teleost prey species of barramundi (Salini et al., 1998). Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the flood plume. Reduced extent of key juvenile and post-larval lateral estuarine and freshwater habitats connected through high volume flood events.</td>
<td>Reduction in year-class strength of barramundi recruits and subsequent population. Adult male and female barramundi remain resident in riverine reaches and do not encounter conditions appropriate to spawning and egg/larval survival, i.e. the lower estuary marine environment. Possible reduction in prey abundance in hypersaline estuary. Limitation of primary production in the flood plume and near-shore waters due to nutrient limitation. Reduced productivity in key zone for barramundi larvae. Reduced habitat extent may reduce post-larval survival and recruitment through a loss of key feeding grounds.</td>
<td>Reduction in barramundi recruitment and subsequent adult abundance, as evidenced by commercial catch usually lagged by 1-4 years (Halliday et al., 2010; Halliday et al., 2012; Robins et al., 2005). Barramundi eggs and early larvae require near marine-salinity waters for survival (20-30 ppt optimal) (Maneewong, 1987). Larvae immigrate from near-shore marine waters to brackish estuaries to access supra-littoral and palustrine habitats inundated during the monsoon season (Halliday et al., 2012; Russell and Garrett, 1985). Barramundi consume fish and crustacean prey with high frequency of occurrence of penaeidawns, mullet and Engraulidae (Salini et al., 1998); all species known to thrive in brackish waters. The inundation of salt flats through high volume flood events provides a source of both carbon and nitrogen for near-shore waters (Burford et al., 2011). Larvae are reliant on near-shore zone as they spend about three weeks in inshore marine and brackish waters to maximise growth and development before moving upstream (Robins and Ye, 2007).</td>
</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
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<td>Loss of connectivity between spawning and juvenile habitats (marine, estuarine and riverine reaches of rivers) due to the placement of barriers in the upper estuary or lower river reach; or to either low or nil flow causing the fluvial separation of waterholes above the estuary</td>
<td>Juvenile barramundi migrate upstream from estuaries to freshwater reaches of rivers and palustrine habitats where they reside for 1 to 5 years. Growth and survival in seasonally accessible habitats likely is optimal (Welch et al., 2014).</td>
<td>Physical barriers block or impede upstream migration of juvenile barramundi to key freshwater habitats, and subsequent emigration as adults downstream to the estuary. Fishways assist migration over barriers with varying levels of success (Stuart and Mallen-Cooper, 1999).</td>
<td>Reduced access to critical habitat for juvenile and sub-adult barramundi. Reduced long stream habitat extent. Reduced abundance of adult barramundi reaching estuary to spawn. Given barramundi are a long-lived species, a single seasonal loss/reduction in spawning success will not severely impact the population; however, repeatedly low spawning success may have population wide consequences.</td>
<td>Barramundi have been observed using fishways to negotiate river barriers such as dams (Stuart and Mallen-Cooper, 1999). In some study areas, adult barramundi living in freshwater reaches shown to not participate in spawning (Milton and Chenery, 2005) or delay downstream emigration for spawning (Crook et al., 2017). Lower spawning stock makes populations more heavily reliant on adults who conform to saline life history and remain in estuary to spawn.</td>
<td></td>
</tr>
<tr>
<td>Change in inter-annual sequencing of floodplain habitats to riverine and estuarine habitats.</td>
<td>Juvenile barramundi that access floodplain habitats or long stream habitats benefit from access to seasonally extensive and productive habitat. But in subsequent years, they must return to the estuary and marine habitats as adults and to spawn. Early wet-season flows support access to freshwater habitats during the early portion of the spawning season.</td>
<td>Loss of annual connectivity of floodplain, upstream and estuarine habitats causes mortality of juvenile and early adult barramundi in refuge and diminishing pondage. Adult barramundi cannot access lower-estuarine spawning habitat, thus reducing recruitment potential from the adult stock. Multiple years of poor recruitment will undoubtedly have social economic and ecological implications.</td>
<td>Strong positive correlation between juvenile barramundi abundance and early wet-season annual rainfall (Griffin, 1987).</td>
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</table>

Barramundi have been observed using fishways to negotiate river barriers such as dams (Stuart and Mallen-Cooper, 1999). In some study areas, adult barramundi living in freshwater reaches shown to not participate in spawning (Milton and Chenery, 2005) or delay downstream emigration for spawning (Crook et al., 2017). Lower spawning stock makes populations more heavily reliant on adults who conform to saline life history and remain in estuary to spawn.
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</tr>
</thead>
<tbody>
<tr>
<td>Altered estuarine salinity and seasonality</td>
<td>Change in salinity regime in extensive coastal wetland habitats in low-gradient geoslopes in tropical coastal ecosystems across northern Australia.</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion reduces the extent or presence of freshwater supralittoral habitats which are optimal habitat for juvenile barramundi.</td>
<td>Reduction in barramundi population due to decline in freshwater habitat extent and productivity at key times in the annual climate cycle. Reduced extent of fresh and brackish water habitats where insects, crustaceans and fish are key species consumed by juvenile barramundi (Russell and Garrett, 1985).</td>
<td>Preference for and higher growth rates in freshwater habitats by juvenile and sub-adult barramundi (Robins et al., 2005; Welch et al., 2014). Brackish conditions are optimal for the growth of banana prawns (Staples and Heales, 1991), a key component of barramundi diet (Salini et al., 1990). Tidal extent of Alligator river in the Northern Territory has shifted up to 4 km inland since 1950 (Winn et al., 2006). The Mary River, Northern Territory, has seen tidal creek systems extend more than 30 km in 50 years (Knighton et al., 1991) and is only likely to be exacerbated with climate change.</td>
<td></td>
</tr>
<tr>
<td>Wetland and floodplain Extraction and regulation/Change in land use and management/Climate change (drying)/Invasive species</td>
<td>Loss of seasonal inundation / habitat extent with extraction and drying climate change Annual flooding of salt flats and wetlands mark an inundation expansion habitat available to barramundi. With flooding, desiccated algal crust on salt pans invigorates and begins to photosynthesize. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016).</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation salt pan and wetland adjacent to tropical estuaries reduces key habitat that becomes available to estuarine fauna during the monsoon season. Reduction in key brackish and freshwater habitat available to post-larval and juvenile barramundi during wet season large flood flows.</td>
<td>Reduction in population size of juvenile and sub-adult barramundi and changes in distribution/habitat use. Post-larvae and juveniles have reduced survival and do not recruit into fishery. Reduction growth in estuaries due to decreased coastal productivity.</td>
<td>Juveniles thrive in semi-permanent wetlands, tidal creeks and freshwater riverine habitats (Russell and Garrett, 1985). When seasonal flows are sufficient, post-larvae and small juveniles access freshwater habitats adjacent to, and upstream of the estuary (Halliday et al., 2012). Reduction in nutrient production from flooded salt flats during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients exported from tropical estuaries with subsequent reduced coastal productivity for species with ontogenetic habitat shifts; i.e. species that emigrate from estuarine nursery habitats to near-shore/offshore adult habitats (Burford et al., 2011).</td>
<td></td>
</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Habitat degradation/ Homogenisation</td>
<td>Important juvenile habitat during flood events and monsoon season.</td>
<td>Loss of habitat due to degraded water quality due to impacts on waterholes and lagoons from invasive animals such as pigs and feral horses. Modification/draining of wetlands for human use e.g. agriculture and urban development.</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
<td>Ungulate grazing in saltmarsh habitats shown to significantly reduce fish and invertebrate abundance in North Carolina (Levin et al., 2002). In Swan Coastal plain in WA, around 70% of original wetlands have been lost through draining and filling as a result of agriculture and urban activities (Halse, 1989).</td>
<td></td>
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<tr>
<td>Altered biotic interactions</td>
<td>Barramundi have key role in food webs and ecology of tropical rivers.</td>
<td>Biotic interactions under climate warming may have different competitive outcomes, or lead to altered distributions within the estuary. Potential for increased predation by crocs/sharks/birds/adult barramundi on juveniles. Increased competition for resources as key prey for barramundi are either affected by water extraction and regulation, climate change, or are predated upon by other organisms.</td>
<td>Reduced survival and recruitment. Reduced growth as food resources decline and increased competitive interactions occur between both co-specifics and conspecifics.</td>
<td>Altered distributions of species in warming rivers has been demonstrated between salmon and bass in the Colombia river (Lawrence et al., 2014).</td>
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</tbody>
</table>
3.2.3  SAWFISH

Asset description

Sawfishes belong to a family (Pristidae) within the Elasmobranchii, which includes sharks and rays. Sawfishes attain very large sizes (in the order of 5 to 7 m total length) and live in tropical and subtropical coastal marine waters as adults (Dulvy et al., 2016; Last and Stevens, 2009). A key feature of the group is the tooth-lined rostrum (or saw), which is a flattened extension of the snout. The saw is important in the specialist, stealth-feeding strategies of these species. It is used to sense and, in some cases, strike and impale prey, including prawns and fish (Bradney et al., 2017; Morgan, 2016a; Morgan, 2016b; Wueringer et al., 2011a; Wueringer et al., 2011b).

Four species of sawfish occupy Australian waters (Table 3-10). In a number of cases, Australian waters represent a substantial portion of the known current day range of each of these species (most notably for the dwarf sawfish, *Pristis clavata* (Dulvy et al., 2016). All of these species are listed as threatened species (Table 3-10). Sawfishes give birth to small litters (~5 to 20) of live young referred to as neonates. Neonates are generally about 50 to 90 cm in total length at birth (Morgan et al., 2011a). Inshore waters, including bays and estuaries, are important nursery grounds for neonates and juvenile sawfishes up until about 4 to 6 years of age (Morgan et al., 2011a; Morgan et al., 2017a; Peverell, 2005). The largetooth (or freshwater) sawfish *P. pristis* makes use of large rivers as nursery habitat in these early years of life (Morgan, 2016b; Thorburn et al., 2007; Whitty et al., 2017; Whitty et al., 2009). Hence, the species is often referred to as the freshwater sawfish. Until recently, this species was referred to as *Pristis microdon* in Australia and in much of the relevant literature (Feutry et al., 2015).

Rationale for key species selection

Sawfishes are important assets for the Assessment, as they are not only present in the relevant study areas, but because these study areas represent important or at least potentially important strongholds for the remnant populations of these threatened fishes (Morgan et al., 2015; Peverell, 2005; Phillips et al., 2017a; Phillips et al., 2017b). As such, sawfishes are conservation-listed species internationally, and nationally (with the exception of the knifetooth sawfish, *Anoxypristis cuspidata*, which is only listed internationally) (Table 3-10). Additionally, sawfishes can be important icons, including cultural icons, for society (Ebner et al., 2016; McDavitt, 2005) and have the potential to migrate at landscape and oceanic scales. Recently, a genetic study revealed that large-scale dispersal of *P. pristis* achieved by males, in contrast to *P. clavata* and the green sawfish (*P. zijsron*), which are more segregated spatially in Australian waters (Phillips et al., 2017a). However, it is important to acknowledge that information on the movement and spatial ecology of sawfishes is incomplete, including pertaining to adult movement and migration, and the distribution of juvenile nurseries.
### Table 3-10 Sawfish species in Australian waters

<table>
<thead>
<tr>
<th>SPECIES</th>
<th>BIOME SCALE</th>
<th>MIGRATION</th>
<th>NURSERY</th>
<th>ADULT HABITAT</th>
<th>CONSERVATION STATUS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Large tooth sawfish, <em>Pristis</em> (L. 1758)</td>
<td>Diadromous</td>
<td>Lowland main channel reaches of large rivers</td>
<td>Coastal marine waters; female probably gives birth in large river estuaries</td>
<td>VulnerableEPBC, Critically endangeredIUCN</td>
<td></td>
</tr>
<tr>
<td>Green sawfish, <em>Pristis zijsron</em> (Latham, 1968)</td>
<td>Marine</td>
<td>Tidal creeks, estuaries and bays</td>
<td>Widespread but exclusively marine waters; female probably gives birth in or near estuary</td>
<td>VulnerableEPBC, Critically endangeredIUCN</td>
<td></td>
</tr>
<tr>
<td>Knifetooth sawfish, <em>Anoxypristis cuspidata</em> (White and Moy-Thomas 1941)</td>
<td>Marine</td>
<td>Inshore marine waters (to 10 m depth)</td>
<td>Marine waters to about 40 m depth</td>
<td>EndangeredIUCN</td>
<td></td>
</tr>
</tbody>
</table>


Source: (Dulvy et al., 2016; Last and Stevens, 2009; Morgan et al., 2011a; Morgan, 2016b; Morgan et al., 2015; Morgan et al., 2017a; Morgan et al., 2017b; Peverell, 2005; Phillips et al., 2017a; Phillips et al., 2017b; Whitty et al., 2017) (http://www.iucnredlist.org; http://www.environment.gov.au/cgi-bin/sprat/public/publicthreatenedlist.pl)

### Distribution in Assessment study areas

Sawfishes are generally widespread in northern Australian marine waters (Figure 3-51, Figure 3-52 and Figure 3-53) but are not necessarily abundant. *P. pristis* historically occurred on the west coast of Australia, in the Northern Territory and in Queensland, including on the east coast. In the present day, this species is rarely detected on the east coast of Queensland, but occurs as juveniles in the rivers and estuaries of the Gulf of Carpentaria, and is regularly detected in the Fitzroy River, WA (Morgan et al., 2011a; Peverell, 2005).

*P. clavata* has been recorded from north-western Australia and is perhaps most consistently detected in King Sound and in the estuary of the Fitzroy River in WA (Morgan et al., 2011a; Thorburn et al., 2008). It exists at low detection levels in the Gulf of Carpentaria and is essentially considered extinct on the eastern coast of Queensland (Peverell, 2005). In contrast, *P. zijsron* is primarily found between Onslow and One-arm Point in WA and thus has a more southerly distribution than *P. clavata* (Morgan et al., 2011a; Morgan et al., 2015). A nursery for *P. zijsron* was recently detected in the Ashburton River delta, near Onslow (Morgan et al., 2015). It is also found in the Northern Territory and the Gulf of Carpentaria (Last and Stevens, 2009; Morgan et al., 2011a; Peverell, 2005).

*A. cuspidata* is the most abundant sawfish in the Gulf of Carpentaria (Peverell, 2005) and is infrequently encountered elsewhere in northern Australia (Last and Stevens, 2009).
Figure 3-51 Distribution of sawfish in the Fitzroy catchment and marine study area
Figure 3-52 Distribution of sawfish in the Darwin catchments and marine study area
Habitat use

Largetooth sawfish, *Pristis pristis*

*P. pristis* has a marine adult phase and the juvenile phase is in freshwater or euryhaline environments (Gleiss et al., 2017; Last and Stevens, 2009; Morgan et al., 2011a; Morgan, 2016b; Morgan et al., 2004; Peverell, 2005; Thorburn et al., 2007; Whitty et al., 2017; Whitty et al., 2009). Genetic studies have shown that adult males provide the means by which this species disperses large distances in marine systems within northern Australia (Phillips et al., 2017a; Phillips et al., 2017b). Pupping occurs in estuaries and river mouths (Last and Stevens, 2009; Morgan, 2016b). In tropical Australian freshwaters, *P. pristis* occupies large pools and waterholes, mostly in the main channel of larger rivers from the Fitzroy in the Kimberley across the Northern Territory and in the Gulf of Carpentaria (Feutry et al., 2015; Last and Stevens, 2009; Morgan et al., 2004; Peverell, 2005), and occasionally in larger offchannel habitat (e.g. Morgan and Beatty (2010)). Juveniles occupy these habitats for the first four to five years of life (Morgan, 2016b). Neonates occupy shallow water relative to
larger and older juveniles (Whitty et al., 2009). These larger juveniles are relatively inactive by day, when they occupy deeper sections of pools (Whitty et al., 2017). They forage over shallow pool margins and in runs and glides at night (Whitty et al., 2017). \textit{P. pristis} is a top predator that feeds on fishes and decapod crustaceans (Thorburn et al., 2014; Thorburn et al., 2007).

\textbf{Green sawfish, \textit{Pristis zijsron}}

\textit{P. zijsron} has a marine adult phase and marine or estuarine juvenile phase (Morgan et al., 2015). Genetic studies have shown limited large-scale dispersal between disjunct populations of this species in northern Australia (Phillips et al., 2017a; Phillips et al., 2017b). Pupping likely occurs in estuaries, sheltered coastal bays or near river mouths during the wet season in Queensland (Peverell, 2005) or late spring in the Pilbara (Morgan et al., 2015). In the Ashburton delta and surrounding coastal area, neonates occupy shallow estuaries and tidal creeks, with larger juveniles moving into increasingly larger home ranges that include coastal marine waters (Morgan et al., 2015; Morgan et al., 2017a). Neonates occupy shallow water relative to larger (older) juveniles (Morgan et al., 2017a). The species feeds on fishes and crustaceans (Last and Stevens, 2009).

\textbf{Dwarf sawfish, \textit{Pristis clavata}}

\textit{P. clavata} has a marine adult phase and marine or estuarine juvenile phase (Morgan et al., 2011a; Thorburn et al., 2008). Genetic studies have shown limited large-scale dispersal between disjunct populations of this species in northern Australia (Phillips et al., 2017a; Phillips et al., 2017b). Pupping likely occurs in estuaries or sheltered coastal bays during the wet season (Peverell, 2005). In King Sound, this species occupy tidal waters, where it feeds on fishes and crustaceans (Morgan et al., 2011a). It is found in shallow and sometimes deeper waters (Morgan et al., 2011a; Peverell, 2005; Stephenson, 2003).

\textbf{Knifetooth sawfish, \textit{Anoxypristis cuspidata}}

\textit{A. cuspidata} is a marine species (Peverell, 2005). Inshore marine waters of the Gulf of Carpentaria are likely important habitat for juveniles of this species and adults are detected offshore (Peverell, 2005). (Peverell, 2005) deduced from catches of adults in surface nets that \textit{A. cuspidata} is both benthic and capable of occupying the water column. The species feeds on squid, fishes and crustaceans (Peverell, 2005).

\textbf{Conceptual model}

Figure 3-54 shows the conceptual model for \textit{P. pristis} in large rivers, with the accompanying narrative given in Table 3-11.
Figure 3-54 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.
### Conceptual model narrative

**Table 3-11 Knowledge underpinning the conceptual model of *Pristis pristis* in northern Australia**

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>River regulation</td>
<td>Change in timing of flow and connectivity of waterways</td>
<td>Change in system connectivity and habitat availability.</td>
<td>Upstream and downstream dispersal of <em>P. pristis</em> minimised if water harvested on tail end of flow or number of high-flow days reduced.</td>
<td>Neonates of <em>P. pristis</em> become aggregated lower in the catchment where they are vulnerable to intraspecific competition and predation.</td>
<td>The number of flow days is important for large-scale <em>P. pristis</em> migration in the Fitzroy River catchment (Morgan, 2016a). Shallow sand banks and narrow glides and narrow runs are important night time feeding habitats for juvenile <em>P. pristis</em> (Whitty et al., 2017). Crocodiles and bull sharks predate and maim <em>P. pristis</em> in a river environment (Morgan et al., 2017b).</td>
</tr>
<tr>
<td>Instream barrier (on tributary)</td>
<td>Reduced connectivity</td>
<td>Reduced dispersal of <em>P. pristis</em> Reduced productivity of amphidromous species including freshwater prawns and shrimps (major prey species for <em>P. pristis</em>) Top predators’ density elevated immediately downstream of barrier (e.g. bull sharks)</td>
<td>Reduced productivity of amphidromous species including freshwater prawns and shrimps (major prey species for <em>P. pristis</em>) Top predators’ density elevated immediately downstream of barrier (e.g. bull sharks)</td>
<td>Reduced growth and survivorship of juvenile <em>P. pristis</em> (This is dependent on the tributary being a sawfish nursery/migration pathway; this is likely most important in tributaries of catchments where pupping rates are high in the absence of failing recruitment by <em>P. pristis</em> adults)</td>
<td><em>P. pristis</em> and predatory bull sharks congregate downstream of Camballin Barrage on the main channel of the Fitzroy River, WA (Morgan, 2016a; Morgan et al., 2017b). The timing of amphidromous migrations and the density of prawns and shrimps is described for the Daly River (Novak et al., 2017; Novak et al., 2015).</td>
</tr>
<tr>
<td>Instream barrier (on main channel)</td>
<td>Reduced connectivity</td>
<td>Reduced dispersal of <em>P. pristis</em> Reduced productivity of amphidromous species including freshwater prawns and shrimps (major prey species for <em>P. pristis</em>) Top predators’ density elevated immediately downstream of barrier (e.g. bull sharks)</td>
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<td>Reduced growth and survivorship of juvenile <em>P. pristis</em></td>
<td><em>P. pristis</em> and predatory bull sharks congregate downstream of Camballin Barrage on the main channel of the Fitzroy River, WA (Morgan, 2016a; Morgan et al., 2017b). The timing of amphidromous migrations and the density of prawns and shrimps is described for the Daly River (Novak et al., 2017; Novak et al., 2015).</td>
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<tr>
<td>1. HABITAT AND THREATS</td>
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<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<td>--------------------------</td>
</tr>
<tr>
<td>Water extraction</td>
<td>Reduced flow and surface water availability</td>
<td>Water drawdown and reduced flow</td>
<td><em>P. pristis</em> pup production correlates with discharge in the Fitzroy catchment</td>
<td>Reduced recruitment of neonate <em>P. pristis</em></td>
<td>There is a positive correlation between total Fitzroy River discharge and neonate production of <em>P. pristis</em> (Morgan, 2016b). The limited year to year difference in the composition and abundance of fishes in offchannel habitats in the mid-Fitzroy River catchment have been attributed to stability provided by groundwater input (Morgan, and Beatty, 2010). Thus, water extraction from these systems is likely to negatively affect resident fish assemblages, and potentially prey available to <em>P. pristis</em>. The majority of energy uptake by main channel fishes is derived from the floodplain rather than the main channel in the Mitchell River catchment (Hunt et al., 2012).</td>
</tr>
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</table>

| Climate change         | Reduced flow and surface water availability | As above                                        | As above                                        | As above                                        | As above |

| Land use               | Direct riparian clearing and stocking of cattle | Reduced bank stability and increased erosion, via trampling and grazing of seedlings, long-term loss of large trees | Loss of day time refuge for 1 to 5-year old *P. pristis* adults, also potential loss of refuge for nocturnal prey species (e.g. *Neosilurus* spp., *L. calcarifer*, prawns) in the form of standing trees with undercut roots and instream wood loads. | Loss of daytime habitat and prey production. | Structure including undercut banks and large woody debris is considered an important mediator of diel shifts and habitat use of *P. pristis* and its prey in pools-runs-glides in main channel of Fitzroy River (Whitty et al., 2017). |

<p>| Fishing pressure       | Recreational fishers, subsistence fishers, illegal fishing | Reduced numbers of large fishes including top predators through direct harvest and bycatch | <em>P. pristis</em> are angled and bycatch in rivers and estuaries. Additionally, anglers selectively remove large-bodied fishes and prawns that are prey of <em>P. pristis</em>. (Note this applies to other species of sawfishes in marine systems) | Reduced growth, abundance and survivorship of <em>P. pristis</em> | (Close et al., 2014; Jellyman, 2016; Morgan, 2016a) |</p>
<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
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<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Interactions</td>
<td>Interactions</td>
<td>There are endless permutations in this category. Some of the more relevant examples include: Increased intensive land use usually results in changes or increases in particular recreational activities as a function of increased staffing in remote areas. This includes recreational fishing pressure. Similarly, increased surface water storage creates localised fishing opportunities, which can lead to fish stocking initiatives and spread of alien species. Additionally, all of the drivers listed above in this table occur in concert with climate change.</td>
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</table>
3.2.4  FRESHWATER WHIPRAY

Asset description
The giant freshwater whipray (*Urogymnus dalyensis*) (Last et al., 2016), previously *Himantura chaophraya/dalyensis* and listed as vulnerable by the International Union for the Conservation of Nature (IUCN) (2003) is a little-known species of stingray in the family Dasyatidae, found in a number of large rivers and associated estuaries in northern Australia. The species represents a potential flagship species for large river ecosystems in northern Australia as a function of its large size (i.e. greater than 2 m disc width in freshwater (Ebner et al., 2016); large individuals probably evade detection with light gear in conventional scientific surveys). There is likely to be minimal exchange between rivers where it occurs, and so separate subpopulations may exist across northern Australia. The species is a top predator, feeding on fishes and crustaceans (Ebner et al., 2016). The ecology and distribution of the species is poorly understood, making it difficult to ascertain the likelihood of human impact. The restricted geographic range of this whipray, rareness of individuals and the presumed limited interchange between rivers, increases the species’ inherent sensitivity to threats (Kyne, 2016).

Distribution in Assessment study areas
The freshwater whipray is likely to be endemic to fresh and estuarine waters of tropical northern Australia, although its range is not fully mapped and it may occur in the Fly River Basin of Papua New Guinea (Last and Stevens, 2009; Last and Manjaji-Matsumoto, 2008; Morgan et al., 2004). The species has been recorded from the Ord, Fitzroy and Pentecost rivers in WA; the Victoria, Daly, Adelaide, South Alligator and Roper rivers in the Northern Territory; and the Mitchell, Gilbert, Normanby and Wenlock rivers in Queensland (Kyne, 2016; Last and Stevens, 2009; Last and Manjaji-Matsumoto, 2008; Morgan et al., 2004; Thorburn et al., 2004). The Daly River (Northern Territory) and Normanby River (Queensland) systems have been identified as sites of significance for the species, given their abundance relative to other systems (Thorburn et al., 2003). In the Fitzroy, this species has been detected from the estuary to as far upstream as Geike Gorge (Morgan, 2016a; Morgan et al., 2011b; Phillips et al., 2017a; Thorburn et al., 2004). The species is naturally rare, and this rarity is further exacerbated by the limited data available on the species. Despite the limited observations of the whipray within the Assessment study areas of interest (Figure 3-55, Figure 3-56 and Figure 3-57), it is important that the species is not classed absent or limited presence, but rather data poor.
Figure 3-55 Distribution of the giant freshwater whipray (*Urogymnus dalyensis*) in the Fitzroy catchment

The limited observations of whiprays are a result of rareness rather than absence of the species.
The limited observations of whiprays are a result of rareness rather than absence of the species.
Habitat use

The freshwater whipray is classified as a brackish marginal species (Last, 1997) that occurs in freshwater and estuarine habitats (Campbell et al., 2012; Ebner et al., 2016; Morgan et al., 2011b). However, the habitat use and movement ecology of this species is not well documented, except for some telemetry work in the Wenlock River system. The species occurs low in the water column at a depth of 1 to 4 m and frequents both turbid and clear waters (Thorburn et al., 2003). The freshwater whipray is capable of migrating tens of kilometres (Campbell et al., 2012), but its life-history patterns and migration patterns, if any, remain unclear. The freshwater whipray is speculated to have a life history similar to the largetooth sawfish. Freshwater rays are born in the estuary and migrate upstream to spend their first years of life in all available freshwater reaches of the rivers and tributaries (>300 km inland) (Burrows and Perna, 2006; Thorburn et al., 2003). As they mature, whiprays move downriver and enter the estuarine environment. As adults (up to 160 to 200 cm disc) they can be found in the upper marine-influenced estuary; they have not been
recorded in euryhaline marine waters (30–40), and most are found in waters less than 10 (Thorburn et al., 2004). Marzullo et al. (2011) found the physiology of the whipray to be similar to that of marine rays. The species is likely to be tolerant of salinities greater than those it is found in.

A knowledge gap exists about the micro-habitat use of freshwater whiprays, and highlights the need for greater understanding of their use of habitat and hence the impact of the modification of flows on their populations. Only one tagging study of the movement of freshwater whiprays has been undertaken. Four tagged rays (two female, two male, all juveniles) were tracked over a 64-km section of the Wenlock River, Cape York (Campbell et al., 2012). During the dry season, each of the four rays inhabited an approximate 8 km freshwater section of the river (up to 104 km upstream) where their movement was cued by tidal and lunar cycles. During the wet season, the behaviour of the two male whiprays was cued by the diel cycle and in part by flood flows; they migrated downstream to a river reach with saline influence (Campbell et al., 2012). Although that study was limited, it suggests that the behaviour of whiprays changes during the monsoon season when flood flows dominate the flow characteristics of the river. A reduction in magnitude of flows can affect the movement of freshwater whiprays. They could become stranded during disconnection periods and become vulnerable in remnant pools, due to their innate behaviour of limited movement. Site-specific changes may limit the ability of individuals to evade both habitat alteration or degradation and exploitation. However, until more information on movement and migration pathways is known, it is difficult to determine the likely impacts of water resource development on the species. Though the Australian freshwater whipray population is most likely sustainable under current levels of catchment impacts in the tropical north, increased modification of catchment connectivity and flows may endanger the species. Cogenetic species in Asia are listed as endangered, and locally they are critically endangered.

**Threats to species**

The paucity of information on this species makes it difficult to ascertain threats. Similar to other freshwater elasmobranchs, biological limitations of the group and the physical constraints of their environment may limit the freshwater whiprays’ ability to evade both habitat alteration or degradation and exploitation (Compagno and Cook, 1995; Kyne, 2016). The species showed a high degree of site fidelity (Campbell et al., 2012) and site-specific change may affect them to a higher degree than species that regularly move large distances within a river. Freshwater whiprays typify a life-history strategy that would be significantly affected by loss of longstream connectivity in the Assessment study areas. Loss of connectivity would include both the placement of barriers to migration, and the loss of baseflow and flood flow that supports longstream movement between waterholes in the river bed.

As a large-bodied, low-fecundity species, the freshwater whipray is vulnerable to direct exploitation and deterioration or loss of main channel and floodplain environments. This species also warrants precautionary consideration in terms of fish passage and particularly upstream fish passage via fishways, in part as a function of the large body size of adults and
especially their width. Whiprays have been observed foraging in very shallow water, including in floodplain wetlands (BC Ebner, DL Morgan and S Doody, unpublished data), indicating that rapid lowering of water levels may leave this species susceptible to stranding. (Morgan, 2016a) reported one dead specimen in relation to an anoxic water event in Geike Gorge in the Fitzroy River.

It is also conceivable that freshwater whiprays consume benthic molluscs, based on the species’ ecomorphology (and the behaviour of other stingrays). This further highlights the insufficient ecological knowledge of this species and acknowledge the need to consider aspects of the food chain when conceptualising the potential threats to the freshwater whipray.

Chin et al. (2010) classified the freshwater whipray as one of the most susceptible chondrichthyan to the effects of climate change on the Great Barrier Reef and in adjacent waters of the eastern Queensland coast, where it occurs in the Normanby River and Endeavour Rivers (BC Ebner, unpublished data). This evaluation was suggested in relation to high exposure to climate change factors (rising sea levels, rising temperature, changing freshwater input, etc.) within its riverine/estuarine habitat. It was assumed that the species has high sensitivity and low adaptive capacity given its relative rarity, habitat specificity and restricted range (Chin et al., 2010).

The interruption of wet-season low flows by instream dams and the reduction of both low and high flood flows by water diversion or impoundment will have impacts on foraging, refuge habitats and seasonal movement patterns of whiprays. Freshwater whiprays require access to freshwater habitats, which would be severely impeded by river development upstream. Physical barriers to longstream connectivity, such as instream dams, barrages and road bridges constructed for access to these newly developed landscapes, interrupt the whiprays’ movement to freshwater habitats and return to the estuary. The freshwater whipray have slow growth, are long lived, and there is strong evidence for population subdivision at the estuary level. Consequently, a reduction in flow is likely to significantly reduce the population size and dynamics.

Importantly, change in the whiprays’ seasonal movement patterns may have significant implications for the sustainability of populations. Both gene flow between distinct riverine populations and reproductive behaviour could be debilitated by modified longstream movement. Although little is known about the population dynamics of whiprays, freshwater sawfish have a very similar life-history strategy (Kyne, 2016). Genetic analysis shows distinct populations of sawfish in neighbouring river catchments (Feutry et al., 2015), suggesting that sawfish movement is restricted to individual rivers. Rivers in the Gulf of Carpentaria differ, with regional populations existing and exchanging genetic material across a suite of rivers. Wet-season connectivity between rivers has been postulated as a conduit for sawfish movement between river catchments (Feutry et al., 2015). Given the similarity between the life history of freshwater sawfish and whiprays, connectivity between rivers would be crucial to maintain genetic diversity over basin-scale river systems. Reduced flows would reduce the connectivity between rivers and may modify the downstream movements of whiprays.
to flooded estuarine and coastal floodplains, where the extensive shallow-inundated salt flats allow their movement between rivers.

In addition, as for sawfish, wet-season estuarine habitats might be the mating and spawning habitat for whiprays (Burrows and Perna, 2006). Any impediment to their downstream movement would limit the reproductive potential of the population. Reduced seasonal flood flows may reduce their downstream migration.

Although many facets of the life history of freshwater whiprays remain poorly understood, change associated with water resource development, such as flow modification, barrier construction and restricted connectivity, will affect their ecology. The downstream effects of infrastructure and impoundment will include shifts in their seasonal movement patterns, limits on their reproductive behaviour, population destabilisation and reduced genetic exchange over inter-catchment and inter-basin scales.

**Conceptual model**

Based on what little is known of the life cycle and habitat use of the freshwater whipray in a few of the rivers in which it is present, from a precautionary perspective it is probably best to assume this species is capable of a mixed life-history strategy. Specifically, a component of the population in a given catchment is potentially capable of entirely completing its life cycle in fresh water (potomadromy), and another faction of the population is capable of completing its life cycle by using both the estuary at some stage in its adult phase and the freshwater reaches of a river in the juvenile phase (diadromy). The presence of extremely large adults in estuary environments (S. Peverell, J. Seymour and B. C. Ebner, unpublished data) provides some basis for prioritising the connectivity between estuary and freshwater in catchments where this species exists. It is also likely that predation on juveniles is intensified in estuary environments, where large marine sharks and crocodiles can be commonplace.

Figure 3-58 shows the conceptual model of whipray in northern Australia.
Figure 3-58 Conceptual model showing the relationship between threats, drivers, effects and outcomes for the whipray in northern Australia

Conceptual model narrative

Freshwater whipray populations are probably best maintained in rivers with good water quality (as elasmobranchs are less robust to high nitrites and nitrates than most teleosts, for instance) and rivers that are well connected, at least in the main channel in most years. The connection between river and estuary is potentially important for this species and this is supported by the limited telemetry work done to date (Campbell et al., 2012).

3.2.5 RIVER SHARKS

Asset description

River sharks belong to the genus *Glyphis* within the whaler shark family Carcharhinidae (Last and Stevens, 2009). All four of the currently recognised living species of river shark live in coastal marine and estuarine waters of the central and west Pacific Ocean, with two species known from Australian waters. The speartooth shark, *Glyphis glyphis*, is known from select northern Australian catchments including the Wenlock River on western Cape York Peninsula and in the Northern Territory, notably in the Adelaide River and Alligator River study areas. It is possibly extinct from the eastern side of Cape York (Last and Stevens, 2009), although it was recently rediscovered in Papua New Guinea along with the northern river shark (*G. garricki*) (White et al., 2015), which was taxonomically described only relatively recently (Compagno et al., 2008). *G. garricki* has a patchy distribution in the Northern Territory in the Adelaide River and Alligator Rivers region and is additionally
known from King Sound near the mouth of the Fitzroy River and from the Joseph Bonaparte Gulf and the Ord River estuary (Last and Stevens, 2009; Morgan, 2018; Thorburn and Morgan, 2004).

These two shark species attain about 2.5 to 3 m in total length. They are born at about 50 cm, but little else is known about their general biology (Last and Stevens, 2009; White et al., 2015). The teeth are broad, triangular and serrated on the upper jaw, and narrow and pointed on the lower jaw (hence the common name, speartooth shark) (Last and Stevens, 2009). The eyes are small and the pectoral fins are large (Last and Stevens, 2009). Both species are considered estuarine species. Populations of *G. glyphis* demonstrate strong genetic structuring that indicates limited interconnection of populations between study areas within northern Australia (Feutry et al., 2014).

River sharks are important assets for the Assessment, as the relevant Northern Territory and Fitzroy River catchments and the associated estuaries represent important strongholds for these species. The hydrology of the estuary, including the tidal regime and freshwater discharge, are important drivers of the ecology of populations of river sharks (Lyon et al., 2017; Pillans et al., 2009). Both these species are conservation-listed nationally and internationally as either Endangered or critically endangered (Compagno et al., 2008; Kyne and Feutry, 2017; Morgan, 2018; Pogonoski, 2003).

**Distribution in Assessment study areas**

*Glyphis* sharks are generally found in coastal marine or brackish water estuarine habitats, but their distribution is patchier in northern Australia than other elasmobranchs associated with river systems, including the bull shark (*Carcharhinus leucas*), or even the largetooth sawfish, *P. pristis*. *G. garricki* is confirmed to inhabit King Sound at the mouth of the Fitzroy River and Joseph Bonaparte Gulf, Ord River and King River estuaries in WA, and the Adelaide, South Alligator and East Alligator rivers in the Northern Territory (Last and Stevens, 2009; Morgan, 2018; Pillans et al., 2009; Thorburn and Morgan, 2004). There is also one confirmed record of an adult from Wessel Islands in the Northern Territory (Pillans et al., 2009).

*G. glyphis* is confirmed from locations in Australia in the Northern Territory (the Adelaide River; South, East and West Alligator rivers; and Murganella Creek) and Cape York Peninsula, Queensland (Port Musgrave, and Wenlock, Ducie, Hey, Embley and Normanby rivers) (Kyne and Feutry, 2017; Last and Stevens, 2009; Lyon et al., 2017; Peverell et al., 2006; Pillans et al., 2009). It was only recently discovered in the Ord River in WA (Morgan, 2018).

Importantly, the distribution of these two *Glyphis* species in Australia represent a substantial component of their entire global distribution, being found otherwise only in southern Papua New Guinea (Last and Stevens, 2009; Pillans et al., 2009; White et al., 2015). Figure 3-59 and Figure 3-60 show the distribution of *Glyphis* spp. in the Fitzroy and Darwin catchments. There were no records for the Mitchell catchment.
Figure 3-59 Distribution of *Glyphis* spp. sharks in the Fitzroy catchment
Habitat use

The most detailed study of *G. glyphis* was based on 22 months of tracking a substantial sample of neonates and juveniles by acoustic telemetry in the Port Musgrave–Wenlock River region on the north-western side of Cape York Peninsula in Queensland (Lyon et al., 2017). That study demonstrates that juvenile *G. glyphis* use upper estuarine creeks and rivers during the dry season and lower estuary embayment areas during the wet season. *G. glyphis* occupies muddy substrate, highly turbid tidal waters with salinity intermediate between fresh water and seawater, in high-turbidity, low-visibility waters, at least in the juvenile phase (Lyon et al., 2017; Peverell et al., 2006; Pillans et al., 2009). There is little known of the movement and habitat use of adults, although there are a few records from marine coastal waters (White et al., 2015). Short-term tracking of three individuals revealed movement upstream in relation to incoming tides and downstream movement with the outgoing tide within the Adelaide River (Pillans et al., 2009).
The movement and habitat use of *G. garricki* has not been studied as comprehensively as *G. glyphis*. However, *G. garricki* is relatively similar to *G. glyphis* in occupying muddy substrate, tidal embayment areas with high turbidity (Pillans et al., 2009; Thorburn and Morgan, 2004). As for *G. glyphis*, there is little known of the movement and habitat use of *G. garricki* adults, although there are a few records from marine coastal waters (Pillans et al., 2009; Thorburn and Morgan, 2004; White et al., 2015; Whitty, 2011).

**Conceptual model**

Figure 3-61 shows the conceptual model for *Glyphis* spp., with the accompanying narrative given in Table 3-12.

![Conceptual model](image)

*Figure 3-61* Conceptual model showing the relationship between threats, drivers, effects and outcomes for *Glyphis* spp. sharks in northern Australia

This example considers understanding outcomes associated with changes in hydrology (blue pathways).
### Conceptual model narrative

**Table 3-12 Knowledge underpinning the conceptual model of *Glyphis* sharks in northern Australia**

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>River regulation</strong></td>
<td>Change in timing of flow and connectivity of waterways</td>
<td>Change in nutrient and physical-chemical processes in the upper tidal sections of estuaries in the dry season</td>
<td>Upstream and downstream range of <em>Glyphis</em> spp. may be altered by water harvest if there are end-of-system changes in terms of flow. The alteration would likely be an upstream shift in distribution for <em>Glyphis</em> and unknown changes to biotic and abiotic characteristics within the habitat/range occupied by these species. The migration of juvenile <em>Glyphis</em> is a function of discharge, and therefore changed flow regimes has the potential to affect the timing of migration for juvenile <em>Glyphis</em>.</td>
<td>Reduced habitat quantity and quality would likely reduce growth and survivorship of neonate and juvenile <em>Glyphis</em> spp.</td>
<td>(Lyon et al., 2017)</td>
</tr>
<tr>
<td><strong>Instream barrier (on tributary)</strong></td>
<td>Reduced connectivity</td>
<td>Unlikely to be a direct effect on passage of <em>Glyphis</em> species since barriers are upstream of tidal areas. However, reduced productivity of amphidromous species including freshwater prawns and shrimps (major prey species for <em>Glyphis</em> spp.) may occur as a consequence of an instream barrier; changes in productivity within the estuary are also a possibility though likely to depend on the scale of hydrological and ecological change relating to damming of upper tributaries.</td>
<td>Potential for reduced growth and survivorship of juvenile <em>Glyphis</em> spp. particularly in upper estuary sections of the species range.</td>
<td>Timing of amphidromous migrations and the survivorship and density of prawns and shrimps in the Daly River is influenced by hydrology (Novak et al., 2017; Novak et al., 2015).</td>
<td></td>
</tr>
<tr>
<td><strong>Instream barrier (on main channel)</strong></td>
<td>Reduced connectivity</td>
<td>As above. Additionally, damming of main channel is far more likely to alter catchment productivity and energy dynamics than damming of minor and even major tributaries.</td>
<td>Potential for reduced growth and survivorship of juvenile <em>Glyphis</em> spp. particularly in upper estuary sections of the species range.</td>
<td>Timing of amphidromous migrations and the survivorship and density of prawns and shrimps in the Daly River is influenced by hydrology (Novak et al., 2017; Novak et al., 2015).</td>
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</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/ STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Water extraction</td>
<td>Reduced flow and freshwater supply to the upper tidal sections of estuary</td>
<td>Water drawdown and reduced flow</td>
<td>Potential for changes in benthic productivity in estuaries could affect <em>Glyphis</em>. Reduced flow and increased number of cease-to-flow days has the capacity to increase productivity and stabilise food webs in estuaries in some circumstances. Potential for increased overlap between <em>Glyphis</em> and competitor and predatory species (e.g. bull sharks)</td>
<td>Potential to alter the growth rate of <em>Glyphis</em> spp. especially in juvenile phase</td>
<td>(Lyon et al., 2017)</td>
</tr>
<tr>
<td>Climate change</td>
<td>Reduced flow and upstream encroachment of saline waters</td>
<td>Potential loss of or change to upper tidal zone (brackish water zone)</td>
<td>As above</td>
<td>As above</td>
<td>As above</td>
</tr>
<tr>
<td>Land use</td>
<td>Sedimentation, pesticide and nutrient inputs to estuary</td>
<td>Changes to chemical and physical properties of benthos in estuary and suspended solids</td>
<td>Potential for changes to primary production, secondary production and chemical-physical properties of estuary (e.g. dissolved oxygen during the dry season)</td>
<td>Potential loss or shift in suitable <em>Glyphis</em> habitat</td>
<td>(Human et al., 2018; Lyon et al., 2017)</td>
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<tr>
<td>Fishing pressure</td>
<td>Recreational fishers, subsistence fishers, illegal fishing; commercial fishing bycatch; <em>Glyphis</em> is vulnerable to netting, line fishing and capture in crab pots.</td>
<td>Reduced numbers of large fishes including top predators through direct harvest and bycatch; Additionally, there is potential for interaction between fishing and migration events in relation to changed flow regimes.</td>
<td>They are bycatch of <em>Glyphis</em> spp. in commercial and recreational fisheries. This includes recreational angling, legal and illegal netting and subsistence fishing. Changing discharge regimes and the related <em>Glyphis</em> migration patterns has potential to interact with bycatch and may warrant spatial-temporal closures for particular fishing practices.</td>
<td>Reduced abundance and survivorship of <em>Glyphis</em> spp. Altered food webs; population size and age structure changes (Particularly in terms of disfavouring large-bodied species, low-fecundity species, and long-lived slow to mature species).</td>
<td>(Kyne and Feutry, 2017; Lyon et al., 2017; Martin, 2005; Thorburn and Morgan, 2004; White et al., 2015; Whitty, 2011)</td>
</tr>
<tr>
<td>Interactions</td>
<td>Interactions between different drivers (e.g. changed hydrology of estuary, fishing pressure, nutrient loading)</td>
<td>Direct harvest by target fishing or bycatch can alter top predator structure within food webs and interact with changes in aquatic communities that are altered by changed flow regimes and altered</td>
<td>There are endless permutations in this category. Some of the more relevant examples include: Increased intensive land use usually results in changes or increases in particular recreational activities as a function of increased staffing in remote areas. This includes increases in recreational fishing pressure in rivers, estuaries</td>
<td>Altered food webs; population size and age structure changes (Particularly in terms of disfavouring large-bodied species, low-fecundity species, and long-lived slow to mature species).</td>
<td>(Human et al., 2018; Kyne and Feutry, 2017; Lyon et al., 2017; Martin, 2005; Peverell et al., 2006)</td>
</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<td>connectivity. Nutrient increases and reduced number of flow days has potential to alter benthic food webs. and coastal marine waters. Similarly, increased surface water storage creates localised fishing opportunities, which can lead to fish stocking initiatives and spread of alien species. The effects of changing hydrology and altering nutrient and pesticide levels in relation to benthic food-web processes in the estuary are potentially relevant; noting that the niche for juvenile <em>Glyphis</em> is feeding on benthic prey in highly turbid tidal waters. Additionally, all of the drivers listed above in this table occur in concert with climate change.</td>
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</table>
3.3 Category: Functional groups

3.3.1 MIGRATORY FISH

Asset description

Fishes comprise the dominant aquatic-vertebrate group in terms of species richness in tropical freshwater catchments of northern Australia. Species range in size from a few centimetres in maximum total length during adulthood to well over a metre in length in the case of scaled fishes, including barramundi, and at least 2 to 3 m in the few elasmobranchs (i.e. sharks, rays, sawfishes) that regularly inhabit fresh water (Allen et al., 2002; Ebner et al., 2016; Morgan et al., 2011a; Thorburn et al., 2014). Freshwater fishes perform central ecological functions and act as structuring agents within floodplain river ecosystems (Jardine et al., 2012a; Rayner et al., 2010). Additionally, a sub-set of these fishes, including a number of large-bodied, diadromous species (species that migrate between freshwater and seawater during their life cycle) are the basis for recreational and subsistence fisheries, are of cultural significance and are icons for northern Australian people (Close et al., 2014; Ebner et al., 2016; McDavitt, 2005).

The total number of extant Australian freshwater fish species is currently unknown, with new species discovered or described each year (Raadik, 2014; Thuesen et al., 2011; Unmack, 2013). Based on current knowledge, the most species-rich families of freshwater scaled fish in tropical Australia include the grunters (Terapontidae), eel-tailed catfishes including the black catfish and Hyrtl’s tandan (Plotosidae), forktailed catfish (Ariidae), rainbowfishes (Melanotaeniidae), hardyheads (Atherinidae), glassfishes (Ambassidae), gobies (Gobiidae) and gudgeons (Eleotridae) (Allen et al., 2002; Pusey et al., 2017; Unmack, 2013).

Freshwater migratory fishes include species with populations or subpopulations that undertake large-scale movement during their life cycle. These migrations may be associated with reproduction or exploiting available habitat and food resources. For instance, the adults of some species move to and from spawning grounds for reproductive purposes, and drifting eggs and larvae can traverse the river, floodplain or even the sea. There may be a non-reproductive function to migrations, such as when larvae migrate to juvenile nurseries, where the availability of appropriate food and shelter influence survivorship. Additionally, while some species seem to be regimental in when, or at least which, cues invoke migrations, other species are expert opportunists. By way of contrast, unseasonal rains will not invoke some species to breed (e.g. freshwater sawfish) (Peverell, 2005), whereas the widespread spangled perch, (*Leiopotherapon uniclor*) seems to colonise ephemeral habitats or marginal areas at any opportunity (Pusey et al., 2004a). Here, small-scale migrations occurring within mesohabitats are not considered.

Rationale for key species selection

All fishes depend upon water for survival. Based on published information, there is a range of migratory strategies and consequences of dispersal in fishes within tropical floodplain
rivers in Australia. However, it is important to acknowledge that this information is incomplete. Notably, the movement and migration of small-bodied species is largely unstudied. Furthermore, it is important to acknowledge that single species can exhibit multiple movement strategies within a population or among populations, and that catchment-specific characteristics can dictate the success of such migration strategies or mixed strategies (Augspurger et al., 2017; Chapman et al., 2012). Nevertheless, a sub-set of species have been selected to capture some of the complexity and heterogeneity in migration among the fish assemblages of the focal catchments in the Assessment (Table 3-13).

Species (or in some cases, closely related species) were not selected randomly, with the following attributes being relevant:

- presence of the species in the catchments of interest
- the existence of scientific understanding of the migration of the species
- specific human interest and value (e.g. fisheries species, conservation-listed species)
- species were selected to capture a variety of among species differences in migration.

The species selected were:

- barramundi (*L. calcarifer*)
- freshwater sawfish (*P. pristis*)
- bull shark (*C. leucas*)
- black catfish (*Neosilurus ater*)
- Hyrtl’s tandan (*N. hyrtlii*)
- sooty grunter (*Hephaestus fugilinosus* and *H. jenkinsi*)
- freshwater longtom (*Strongylura kreffti*)
- spangled perch (*L. unicolor*)

Collectively, these species occupy a range of freshwater habitats, including inchannel and offchannel environments, as well as upper and lower catchment areas. Some complete their entire life cycle in fresh water (i.e. black catfish, sooty grunter, spangled perch, and longtom) and others require the estuary or ocean at some stage in their development (i.e. barramundi, bull shark, largetooth sawfish). The largetooth sawfish is a threatened species (Morgan et al., 2011a; Stevens et al., 2005) that was until recently, referred to as *P. microdon* in much of the relevant literature (Feutry et al., 2015). Similarly, recent genetic work indicates that *H. fugilinosus* and *H. jenkinsi* warrant being considered as a single species (Pusey et al., 2016). With the exception of the freshwater longtom, the other species selected in the freshwater migratory fishes asset are all essentially high-flow spawners.
Table 3-13 Focal fish species selected to represent migration and system connectivity

<table>
<thead>
<tr>
<th>BIOME SCALE MIGRATION</th>
<th>FRESHWATER MACROHABITAT</th>
<th>FRESHWATER MESOHABITAT USE</th>
<th>SPECIES</th>
<th>SPAWNING/PUPPING STRATEGY</th>
</tr>
</thead>
<tbody>
<tr>
<td>Potamodromous</td>
<td>Main channel lowland</td>
<td>Spawning in shallow margins</td>
<td>Spangled perch, <em>Leiopotherapon unicolor</em></td>
<td>Paired spawning in margins during wet season</td>
</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel upland</td>
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<td></td>
</tr>
<tr>
<td></td>
<td>Ephemeral and permanent water bodies</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel lowland</td>
<td>Spawning in rifflies</td>
<td>Black catfish, <em>Neosilurus ater</em></td>
<td>Paired spawning in rifflies during wet season</td>
</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel upland</td>
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<td></td>
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<tr>
<td></td>
<td>Ephemeral creeks are nurseries</td>
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<td></td>
<td>Permanent water bodies dry-season adult refuge</td>
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</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel lowland</td>
<td>Spawning in rifflies</td>
<td>Hyrtl’s tandan, <em>Neosilurus hyrtlii</em></td>
<td>Spawning in rifflies during wet season</td>
</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel upland</td>
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<td></td>
</tr>
<tr>
<td></td>
<td>Ephemeral and permanent water bodies</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Potamodromous</td>
<td>Main channel lowland</td>
<td>Run-pool: undercut banks and woody debris association; also spawning in slack water adjacent to run/riffle</td>
<td>Sooty grunter, <em>Hephaestus spp.</em></td>
<td>Migrates to runs/rifflies to spawn during flow</td>
</tr>
<tr>
<td>Potamodromous</td>
<td>and upland</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diadromous</td>
<td>Widespread</td>
<td>Open water and edges of structure including macrophytes</td>
<td>Freshwater longtom, <em>Strongylura krefftii</em></td>
<td>Submerged macrophytes for spawning</td>
</tr>
<tr>
<td>Diadromous</td>
<td>Widespread except upstream of major barriers</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diadromous</td>
<td>Lowland main channel reaches</td>
<td>Pool-run-glide (diesel activity)</td>
<td>Freshwater sawfish, <em>Pristis pristis</em></td>
<td>Marine adult female pups near river mouth during wet season</td>
</tr>
<tr>
<td>Diadromous</td>
<td>Lowland main channel reaches</td>
<td></td>
<td></td>
<td>Marine adult female pups near river mouth</td>
</tr>
</tbody>
</table>

**Distribution in Assessment study areas**

The focal species representing the migratory freshwater fishes asset are widespread and shared across all study areas (Table 3-13). Distribution data associated with individual species is presented in Figure 3-62, Figure 3-63 and Figure 3-64. The differences in data availability and survey effort across the Assessment study areas are acknowledged in Appendix B. It is also important to recognise that the majority of the surveys used as the basis for these data are based on dry-season sampling, primarily as a function of the difficulty of surveying fishes in large rivers during the wet season and flooding period. This has ramifications for understanding the migration of freshwater fishes, since distribution data are confined to periods of no or low migration. For instance, species may move over
the Mitchell River floodplain for weeks to months, but be confined to main channel and distributary refugia during the dry season.

Figure 3-62 Distribution of focal species of the migratory freshwater fishes asset in the Fitzroy catchment
Figure 3-63 Distribution of focal species of the migratory freshwater fishes asset in the Darwin catchments
Potamodromous fishes

Focal species: Spangled perch (*L. unicolor*)

*L. unicolor* attains a maximum total length (TL) of about 300 mm, but more typically adults reach about 200 mm TL (Allen et al., 2002; Pusey et al., 2004a). This renders it a medium-bodied fish in large, lowland river systems, although it can be a relatively large-bodied species in the context of fish assemblages in small, upland streams and marginal habitats. The species has a broad temperature and dissolved oxygen tolerance (Gehrke and Fielder, 1988; Llewellyn, 1973), facilitating its capability as a habitat generalist. It is widespread in lowland and upland main channels, tributaries and distributaries and floodplain habitats (Balcombe et al., 2005; Bishop et al., 1990; Morgan et al., 2004; Pusey, 2011; Pusey et al., 2004a). *L. unicolor* is primarily active by day as a visual carnivore (Bishop et al., 1995; Davis
et al., 2011; Gehrke, 1988) and feeds on aquatic insects in the Fitzroy River (Kimberley) (Thorburn et al., 2014). It is renowned for rapid migration in ephemeral habitats (Kerezsy et al., 2011; Unmack, 2001) and spreads during flooding (Ellis et al., 2015). L. unicolor is at times piscivorous (Balcombe et al., 2005) and functions as a top predator in the absence of larger species, such as L. calcarifer and H. fugilinosus, and is a mesopredator in species-rich fish assemblages, such as those occurring in lowland main channel environments. Flooding enhances breeding, but is not necessary (Llewellyn, 1973).

**Focal species: Sooty grunter (H. fugilinosis and H. jenkinsi)**

*Hephaestus* sp. attains a maximum of 500 mm TL but more typically reaches 300 mm TL (Allen et al., 2002; Pusey et al., 2004a). *Hephaestus* sp. transitions from feeding on benthic macroinvertebrates to terrestrial invertebrates to a diverse adult diet that includes fruits, and terrestrial vertebrates (Davis et al., 2011). *Hephaestus* spp. are a riverine inhabitant that spawn in fresh water (Hogan and Nicholson, 1987) and in slack water adjacent to riffles and runs (Pusey et al., 2004a). Adult spawning migrations may be short (e.g. hundreds of metres) or substantial (tens of kilometres), depending on the proximity of suitable spawning grounds (Pusey et al., 2004a). Spawning occurs in the wet season in the catchments of interest to the Assessment (i.e. the dry tropics rather than the wet tropics).

Juveniles are abundant in shallow water streams, and riffles, runs and glides in main channel environments. Adults are most abundant in pools and deep habitats with structure (Pusey et al., 2004a). The population-level response of *Hephaestus* sp. to flows is scale dependent. For instance, juvenile recruitment is favoured by non-variable flows in the Daly River, particularly at high-flow sites within the catchment (Stewart-Koster et al., 2011).

**Focal species: Black catfish (N. ater)**

*N. ater* is a medium (commonly 300 to 400 mm TL) to large-bodied eel-tailed catfish (>500 mm TL) that feeds mostly at night as a benthic carnivore (Allen et al., 2002; Pusey et al., 2004a). Adults are typically associated with undercuts, trees roots and complex structure by day and forage in open water on sand and mud flats at night. While they may be frequently solitary, this species is aggregated (Pusey et al., 2004a). The diet is predominantly aquatic invertebrates, including molluscs, detritus and fishes (cf. review of diet studies in (Pusey et al., 2004a; Rayner et al., 2009). Spawning occurs in flowing water and particularly riffle-run sections of ephemeral and seasonal creeks and upland tributary streams (Orr and Milward, 1984).

**Focal species: Hyrtl’s tandan (N. hyrtlii)**

*N. hyrtlii* is a small to medium-bodied eel-tailed catfish (commonly to 300 mm TL, very occasionally to 450 mm TL) that feeds as a benthic carnivore (Allen et al., 2002; Pusey et al., 2004a). It is observed to school and be active by day as both juvenile and adult. This species is a habitat generalist occupying main channels, tributaries and distributaries, floodplain wetlands and dune lakes (Pusey et al., 2004a). It is a high-flow spawner (Kerezsy et al., 2011) undertaking lateral or upstream migrations as adults to spawn in flowing riffle-runs, with multiple spawning events possible per wet season (Orr and Milward, 1984; Pusey et al.,
Small and large juveniles are sometimes found in large schools moving over sand following the wet season, including during the day.

**Focal species: Freshwater longtom (S. krefftii)**

*S. krefftii* is a habitat generalist and has been found to occupy a wide range of main channel and offchannel habitats (Bishop et al., 1990; Bishop et al., 1995; Morgan and Beatty, 2010; Pusey et al., 2004a). Habitat use changes with age; small individuals mainly use floodplain lagoons, sandy creeks and corridor lagoons, which are likely to provide dispersal routes (Bishop, 2001). Bishop et al. (2001) found larger *S. krefftii* to be rare in floodplain lagoons and more common in lowland muddy lagoons in the Alligator rivers region. Despite being a habitat generalist, *S. krefftii* relies on aquatic macrophytes and overhanging vegetation for spawning and feeding. *S. krefftii* is predominantly piscivorous and has been known to launch ambush attacks on smaller fish species from the cover of overhanging vegetation (Pusey et al., 2004a; Thorburn et al., 2014). The eggs of this species have tendrils, which are presumed to stick to submerged vegetation (Lake, 1971).

**Diadromous fishes**

**Focal species: Barramundi (L. calcarifer)**

The barramundi is widespread in coastal, estuarine and freshwater habitats downstream of major barriers in catchments of northern Australia. Adults migrate to the vicinity of river mouths to spawn on mudflats (Davis, 1985). Larvae have been collected in tidal habitats; small juveniles use swamps and wetlands as nurseries, and larger juveniles and sub-adults either remain in these habitats or shift into main channel and estuary habitats (Davis, 1985; Moore, 1982; Morgan and Beatty, 2010; Russell and Garrett, 1985; 1988). It was reported that there was a general tendency for larger fish to move seaward during a tagging study (Davis, 1985), but this is not necessarily the case in the Fitzroy River in the Kimberley, where large barramundi remain in the upper reaches of the catchment (DL Morgan, unpublished data). *L. calcarifer* is a top predator in main channel and offchannel environments, as well as in estuarine and coastal waters (Pusey et al., 2004a; Salini et al., 1998; Thorburn et al., 2014).

**Focal species: Bull shark (C. leucas)**

*C. leucas* has a coastal–estuarine adult phase (Cliff and Dudley, 1991; O’Connell et al., 2007; Werry et al., 2011) and its juvenile phase represents the most widely distributed elasmobranch in Australian fresh waters (Ebner et al., 2016; Last and Stevens, 2009). Pupping occurs in estuaries and river mouths (Last and Stevens, 2009). Recruitment is a function of discharge in the Fitzroy River (Laolada, 2015). Neonates occupy low-salinity habitats where the freshwater-estuary connection is maintained (Werry et al., 2012), whereas neonates, larger juveniles and occasionally adults occupy freshwater reaches where rivers disconnect from the seas during the dry season (Morgan et al., 2004; Thorburn and Rowland, 2008). In tropical Australian freshwaters, *C. leucas* is an open-water species occupying large pools and waterholes, mostly in the main channel. Juveniles occupy deeper sections of pools by day and the upper water column during dusk, night and dawn (Laolada,
The species is a top predator that feeds on large fishes, sawfish and terrestrial and semi-aquatic vertebrates (Thorburn et al., 2014; Thorburn and Rowland, 2008).

**Focal species: Largetooth sawfish or freshwater sawfish (P. pristis)**

*P. pristis* has a marine adult phase, while the juvenile phase is in freshwater or euryhaline environments (Gleiss et al., 2017; Last and Stevens, 2009; Morgan et al., 2011a; Morgan, 2016a; Morgan et al., 2004; Peverell, 2005; Whitty et al., 2017). Pupping occurs in estuaries and river mouths (Last and Stevens, 2009; Morgan, 2016b). In tropical Australian freshwaters, *P. pristis* occupies large pools and waterholes, mostly in the main channel of larger rivers from the Fitzroy in the Kimberley across the Northern Territory and in the Gulf of Carpentaria (Feutry et al., 2015; Last and Stevens, 2009; Morgan et al., 2004; Peverell, 2005) and is occasionally found in large offchannel habitat (Morgan and Beatty, 2010). Juveniles occupy these habitats for the first four to six years of life (Morgan, 2016b). Neonates occupy shallow water relative to larger (older) juveniles (Whitty et al., 2009). These larger juveniles are relatively inactive by day, when they occupy deeper sections of pools (Whitty et al., 2017). They forage over shallow pool margins and in runs and glides at night (Morgan et al., 2017a). *P. pristis* is a top predator that feeds on fishes and decapod crustaceans (Thorburn et al., 2014).

**Conceptual model**

Figure 3-65 shows the conceptual model for migratory fish in northern Australia, with the accompanying narrative given in Table 3-14.
Figure 3-65 Conceptual model illustrating the relationship between threats, drivers, effects and outcomes for migratory fish in northern Australia
### Conceptual model narrative

**Table 3-14 Knowledge underpinning the conceptual model of migratory freshwater fishes in northern Australia**

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>River regulation</td>
<td>Change in timing of flow and connectivity of waterways</td>
<td>Change in system connectivity and habitat availability.</td>
<td>Changes to when a spawning ground (e.g. riffles for <em>Neosilurus</em> spp., margins for <em>L. unicolor</em>) is accessible, has implications for the success of spawning in different species. Dispersal of main channel specialists e.g. elasmobranchs, <em>Hephaestus</em> sp. and some generalists (e.g. <em>L. calcarifer</em>) minimised if water harvested on tail end of flow or number of high-flow days reduced.</td>
<td>Reduced recruitment of potamodromous migratory species; mainly species accessing ephemeral or seasonal habitats for spawning or nursery function. Increased recruitment and abundance may occur for some species since they are differentially favoured over species with highly synchronised spawning strategies (e.g. <em>L. unicolor</em>). Dispersal of elasmobranchs (large-bodied species) minimised if water harvested on tail end of flow or number of high-flow days reduced.</td>
<td>Orr and Milward (1984) found that both <em>N. hyrtlii</em> and <em>N. ater</em> use small tributary riffle habitat for spawning during the wet season. Adult <em>Hephaestus</em> sp. usually occupy main channel pools and runs and migrate to slack water immediately adjacent riffles for spawning (Pusey et al., 2004a). <em>L. unicolor</em> are a rapid coloniser (Kerezsy et al., 2011) and fast reproducing species (with short generation times) that can reproduce without flooding (Llewellyn, 1973), and persist at upstream extremities and in marginal areas (Pusey et al., 2004a). (Morgan et al., 2014) discuss the effect of the number of flow days as important for large-scale elasmobranch migration.</td>
</tr>
<tr>
<td>Instream barrier (on tributary)</td>
<td>Reduced connectivity</td>
<td>Reduced access to riffles for migratory fishes that are obligate high-flow spawners (e.g. <em>N. ater</em>, <em>N. hyrtlii</em>, <em>Hephaestus</em> sp.) Reduced access to floodplains and wetlands for non-obligate high-flow spawners (e.g. <em>L. unicolor</em>, <em>S. krefftii</em>) Downstream migration of larvae, juveniles and adults potentially compromised by sharp drop in stream elevation</td>
<td>Reduced recruitment and population decline of <em>N. ater</em>, <em>N. hyrtlii</em>, <em>Hephaestus</em> sp.</td>
<td>Orr and Milward (1984) found that both <em>N. hyrtlii</em> and <em>N. ater</em> use small tributary riffle habitat for spawning during the wet season. Adult <em>Hephaestus</em> sp. usually occupy main channel pools and runs and migrate to slack water immediately adjacent riffles for spawning (Pusey et al., 2004a). No specific studies on downstream fish passage in Tropical Australia, evidence taken from temperate investigations (Baumgartner et al., 2006; O’connor et al., 2005).</td>
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<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Instream barrier</td>
<td>Reduced connectivity</td>
<td>Reduced access throughout the catchment to all species</td>
<td>Reduced numbers of juvenile elasmobranchs and L. calcarifer in catchment; Elevated predation of C. leucas on P. pristis and L. calcarifer likely. Morgan et al. (2016a) provides information on elasmobranch movement in the main channel of the Fitzroy River, WA, including in relation to the Camballin Barrage. Novak et al. (2015) and (Novak et al., 2017) provide evidence of the timing of amphidromous migrations and density of prawns and shrimps in the Daly River.</td>
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<tr>
<td>Water extraction</td>
<td>Reduced flow and surface water availability</td>
<td>Water drawdown and reduced flow</td>
<td>Reduced abundance of species using floodplain wetlands during life cycle</td>
<td>Limited year to year difference in the composition and abundance of fishes in offchannel habitats in the mid-Fitzroy River catchment attributed to groundwater input by Morgan and Beatty (2010). Thus, water extraction from these systems likely to negatively affect resident fish assemblages. Hunt et al. (2012) demonstrated that the majority of energy uptake by main channel fishes is derived from the floodplain rather than main channel in the Mitchell River catchment</td>
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<tr>
<td>Climate change</td>
<td>Reduced flow and surface water availability</td>
<td>As above</td>
<td>As above</td>
<td>As above</td>
<td>As above</td>
</tr>
<tr>
<td>Land use</td>
<td>Direct riparian clearing and stocking of cattle</td>
<td>Reduced bank stability and increased erosion, via trampling and grazing of seedlings, long-term loss of large trees</td>
<td>Loss of day time refuge for adults of: Neosilurus spp., Hephastus sp., L. calcarifer and additional prey species for elasmobranchs (e.g., prawns) in the form of standing trees with undercut roots and instream wood loads. Also, loss of night time refuge for Hephastus sp.</td>
<td>Reduced abundance of bank and wood associated migratory fishes. Also, potentially adverse for P. pristis via loss of daytime habitat</td>
<td>Diel shift in habitat use in terms of P. pristis and prey in pools-runs-glides in main channel of Fitzroy River – structure including undercut banks and large woody debris is potentially an important mediator (Whitty et al., 2017).</td>
</tr>
<tr>
<td>Fishing pressure</td>
<td>Recreational fishers, subsistence fishers, illegal fishing</td>
<td>Reduced numbers of large fishes including top predators through direct harvest and bycatch</td>
<td>Large-bodied migratory fishes selectively removed from ecosystem</td>
<td>Reduced abundance of adult N. ater, Hephastus sp., L. calcarifer Reduced abundance of P. pristis and C. leucas</td>
<td>Close et al. (2014), Jellyman (2016) and Morgan et al. (2016)</td>
</tr>
<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<td>Interactions</td>
<td>Interactions</td>
<td>There are endless permutations in this category. Some of the more relevant examples include:</td>
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<td></td>
<td>between different</td>
<td>Increased intensive land use usually results in changes or increases in particular recreational activities as a function of increased staffing in remote areas. This includes recreational fishing pressure. Similarly, increased surface water storage creates localised fishing opportunities, which can lead to fish stocking initiatives and spread of alien species. Additionally, all of the drivers listed above in this table occur in concert with climate change.</td>
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<td></td>
<td>drivers</td>
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3.3.2 STABLE FLOW SPAWNING FISH

Asset description

Northern Australia is considered to be a region of high species richness for freshwater fishes. A total of 176 species of bony fish and six species of elasmobranch have been recorded in freshwaters in northern Australia (Pusey, 2011). Eighty-six of these species reside exclusively in freshwaters, while the remaining 90 require access to marine or estuarine waters for a part of their life cycle (Pusey, 2011).

Water resource development often reduces the frequency, magnitude and duration of floodplain inundation and flow events (King et al., 2015; Poff and Zimmerman, 2010). Some fish species, such as the barramundi (*L. calcarifer*) rely on high flows to initiate spawning (Pusey et al., 2004a). Potential impacts of reduced flows on high-flow spawners may include adverse impacts on cues for spawning and recruitment success. Floods increase resource availability and inundate floodplains, which provide important habitat and nursery grounds for fish species. Floods are undoubtedly important for ecosystem health and processes, as evidenced by the flood pulse concept proposed by Junk et al. (1989). In contrast, the low-flow recruitment hypothesis proposed by Humphries et al. (1999) recognises that spawning and recruitment in some fish species is independent of floods and that warm, low-flow conditions provide a favourable environment. The low-flow recruitment hypothesis proposes that recruitment success of low-flow spawning species is likely based on prey availability and concentrations of suitable small prey (i.e. zooplankton) that are available for larvae in main channels during low-flow periods.

Stable flow spawners are an important group of fish that spawn in association with stable flows (low flow, baseflow and cease-to-flow) or as a result of other triggers, such as temperature or Julian day. This diverse, functional group of fish is represented by species that will spawn in stable flows independent of flooding, and predominately spawn in the dry season. The stable flow spawning fish functional group includes species such the freshwater longtom (*S. krefftii*), mouth almighty (*Glossamia aprion*), bony herring (*Nematalosa erebi*), barred grunter (*Amniataba percodeis*), flyspecked hardyhead (*Craterocephalus stercesmuscarum stercesmuscarum*), freckled hardyhead (*C. lentiginosus*), eastern rainbowfish (*Melanotaenia splendidia splendidia*), chequered rainbowfish (*M. splendidia inornata*) and western rainbowfish (*M. australis*).

The mechanisms under which water resource development is likely to affect stable flow spawning fish species in comparison with other species, such as those that require high flows for spawning, is different. Stable flow spawning fish are important ecologically, as they are critical in the transfer of energy between trophic levels (*N. erebi*), are a structuring force on fish community dynamics (*S. krefftii, A. percodeis* and *G. aprion*) and are a key food source to higher trophic levels (*Craterocephalus* spp., *Melanotaenia* spp. and *N. erebi*). Low-flow habitats, such as backwaters, floodplain lagoons and other still, littoral habitats, are key nursery and spawning habitats for stable flow spawning fish (Humphries et al., 1999; King et al., 2015). Despite stable flow spawning fish not requiring high flows for spawning, they still
require flows for migration, as well as for the creation and maintenance of habitats. Changes in habitat availability, structure, size and quality of dry-season refugia habitats are likely to adversely affect stable flow spawning fish species.

Rationale for key species selection

Within the stable flow spawners functional group, three categories emerge with slightly differing flow and habitat requirements. The three categories include: large-bodied (predators), medium-bodied (mesopredators) and small-bodied (prey) stable flow spawners (Table 3-15). The three categories are also reflective of the Winemiller and Rose (1992) three life-history strategies. Life-history strategies are based on known life-history traits and the life-history strategy at family level constructed by Beesley (2006) and Sternberg and Kennard (2014).

Table 3-15 Key species within the three categories of stable flow spawners: large-bodied, medium-bodied and small-bodied

<table>
<thead>
<tr>
<th>LARGE-BODIED</th>
<th>MEDIUM-BODIED</th>
<th>SMALL-BODIED</th>
</tr>
</thead>
<tbody>
<tr>
<td>*Freshwater longtom (Strongylura krefftii)</td>
<td>Mouth almighty (Glossamia aprion)</td>
<td>Flyspecked (Craterocephalus stercusmuscarum stercusmuscarum) / freckleheaded (Craterocephalus lentiginosus) hardyhead (Craterocephalus spp.)</td>
</tr>
<tr>
<td>Bony herring (Nematalosa erebi)</td>
<td></td>
<td>Eastern (Melanotaenia splendida splendida) / chequered (Melanotaenia splendida inornata) / western (Melanotaenia australis) rainbowfish (Melanotaenia spp.)</td>
</tr>
<tr>
<td>*Barred grunter (Amniataba percoideus)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Periodic</td>
<td>Periodic/ Equilibrium</td>
<td>Opportunistic</td>
</tr>
</tbody>
</table>

* = little known, likely that the species aligns to the life-history strategy

Hardyheads (Craterocephalus spp.) provide a useful species group to consider as low-flow spawners for assessment of water resource development. They can be abundant in river or stream habitats, and can be especially abundant in floodplain wetlands. Specifically, there is considerable information available on the breeding and recruitment ecology of C. stercusmuscarum stercusmuscarum (Pusey et al., 2004a). The species is distributed in most of the rivers draining into the Gulf of Carpentaria, including the Mitchell River, and the Northern Territory catchments of interest in the Assessment, but is not present in the Kimberley region (Allen et al., 2002; Morgan et al., 2011b). However, an ecomorphologically similar species, C. lentiginosus, is present in the Fitzroy catchment (Morgan et al., 2011b). Therefore, these two hardyhead species are referred to collectively as Craterocephalus spp.

Similarly, no single rainbowfish species is distributed across all of three Assessment study areas. M. splendida is present and often abundant in the catchments of interest, with the exception of the Fitzroy River (Allen et al., 2002; Morgan et al., 2011b). M. australis replaces M. splendida as the common rainbowfish species within the Fitzroy River (Allen et al., 2002; Morgan et al., 2011b). The subspecies M. splendida splendida and M. splendida inornata differ on the basis of colour pattern and body depth, and are an ecomorphologically similar species (Allen, 1982; Pusey et al., 2004a). Therefore, the Melanotaenia spp. group will be used as a surrogate for two species, M. australis and M. splendida, and two subspecies,
M. splendida splendida and M. splendida inornata. The current species status of the selected rainbowfish is reflective of current literature.

Large, medium and small-sized fishes were selected to represent the low-flow spawning fish functional group. The basis for this was to represent the complexity associated with the stable flow spawners, including food-chain and predation effects. The size classes represent not only the size of the fishes, but the life-history strategies proposed by Winemiller and Rose (1992) used by the species and the variation in trophic levels and feeding guilds. The delineation for body size classes is somewhat arbitrary, with small being less than 10 cm TL, medium relating to species with an adult phase of between 10 and 40 cm TL, and large being greater than 40 cm TL. The only dry-season spawner in the latter category, revealed from scientific literature (Pusey et al., 2004a) and the experience of the authors, is the freshwater longtom, S. krefftii. It is a carnivore with strong piscivorous tendencies (Bishop et al., 1990; Pusey et al., 2004a). The three medium-bodied size species selected were the mouth almighty, G. aprion (mesopredator/piscivore), barred grunter, A. percoides (omnivore/microphagous carnivore), and bony herring, N. erebi (detritivore/herbivore) (Davis et al., 2010; Pusey et al., 2004a). The three medium-bodied species use different feeding guilds and have differing habitat and spawning requirements. All of these species are zooplanktivorous as larvae (Pusey et al., 2004a).

Species were selected to be representative of the functional group primarily on available information and their distribution throughout the Assessment study area. There are several species that are highly endemic to certain areas in northern Australia (i.e. M. eachamensis) or highly distributed throughout the study area, but with little ecological information. The impacts on water resource development on the selected species may not be reflective of all stable flow spawning fish species in northern Australia, and so it is important to consider other species, particularly highly endemic species.

**Distribution in Assessment study areas**

Figure 3-66, Figure 3-67 and Figure 3-68 illustrate the distribution of stable flow spawning fish species in the three Assessment study areas.
Figure 3-66 Distribution of stable flow spawning fish species in the Fitzroy catchment
Figure 3-67 Distribution of stable flow spawning fish species in the Darwin catchments
Habitat use

Large-bodied, stable flow spawners

Key species: freshwater longtom (*S. kreffti*)

*S. kreffti* are habitat generalists and have been found to occupy a wide range of both main channel and offchannel habitats. Habitat use changes with age; small individuals mainly use floodplain lagoons, sandy creeks and corridor lagoons, which are habitats that are likely to provide dispersal routes (Bishop et al., 1990). Bishop et al. (1990) found larger *S. kreffti* to be rare in floodplain lagoons and more common in lowland muddy lagoons in the Alligator rivers region. However, it has been found to occupy the coastal wetlands of the Northern Territory (Allsop and De Lestang, 2000). Despite being habitat generalists, *S. kreffti* rely on aquatic macrophytes and overhanging vegetation for spawning and feeding. They have been known to launch ambush attacks on smaller fish species from the cover of overhanging vegetation (Pusey et al., 2004). Lake (1971) described the eggs of *S. kreffti* as having tendrils, which are suggested to stick to submerged vegetation.
Medium-bodied, stable flow spawners

**Key species:** mouth almighty (*G. aprion*), bony herring (*N. erebi*) and barred grunter (*A. percoides*)

Similar to large-bodied stable flow spawners, medium-bodied stable flow spawners occupy a vast range of habitats. *N. erebi* occupy different main channel and off-channel habitats (Pusey et al., 2004a). *N. erebi* are a highly widespread and adaptable species. Their habitat preferences are likely influenced by their preferred food source of algae and detritus (microalgae) rather than the physical habitat (flow, substrate, cover) (Pusey et al., 2004a). *N. erebi* can persist for long periods in waterholes in the dry season (Balcombe et al., 2014) and can occur in zero-flow to fast-flowing reaches. They benefit from drying conditions, due to reduced competition for algae resources from other species, and their populations often boom after significant waterhole isolation and drying (Balcombe and Arthington, 2009). Backflow billabongs are important for spawning, although *N. erebi* can also breed in corridor water bodies (Bishop et al., 1990).

*G. aprion* occur in zero to low-flow areas with a range of substrate types, from sand to bedrock. They prefer deep pools with abundant cover, which is used to launch ambush attacks on prey (Pusey et al., 2004a). *G. aprion* is one of the few native species is suggested to have benefited from the spread of the exotic weed para grass (*Brachiaria mutica*) (Pusey et al., 2004a). Juveniles occur mostly in floodplain lagoons. Adults occur in these habitats but also occupy main channel habitats, sandy creeks and escarpment perennial streams. Eggs of *G. aprion* are incubated in the buccal cavity of males, the presence of still-water conditions is important for the transfer of eggs from the female to the buccal cavity of the male (Bishop et al., 1990).

*A. percoides* inhabit still pools and fast-flowing streams in clear or turbid water. They have been found in lotic main channel habitats, corridor and floodplain lagoons and sandy creek bed and escarpment pools (Bishop et al., 1990). Pusey et al. (1998) found *A. percoides* to be more abundant in shallow stream reaches with coarse substrate and flows greater than 0.3 m/sec in the Burdekin River, Queensland. They have only been recorded in fresh water (Pusey, 2000) and show an aversion to high-gradient tributary streams (Pusey et al., 2004a). Spawning can occur in numerous habitats from main channel water bodies to sandy creek beds and floodplain lagoons (Bishop et al., 1990; Pusey et al., 2004a). However, little is known about the larval development of this species. *A. percoides* have been found to have a wide temperature and salinity tolerance, and would be regarded as habitat generalists.

All three key medium-bodied species have been found to occur in the coastal wetlands of the Northern Territory (Allsop and De Lestang, 2000). The medium-bodied species display similar preferences for low-flow conditions and macrophyte cover, and would be regarded as habitat generalists. However, it is obvious that off-channel habitats, such as floodplain lagoons, are important throughout different life-history stages.
Small-bodied, stable flow spawners

**Key species:** *Craterocephalus spp.*, *Melanotaenia spp.*

The small-bodied *Craterocephalus* spp. (including *C. s. stercusmuscarum* and *C. lentiginosus*) have been found to be abundant in habitats with fine to medium-size substrates and submerged aquatic macrophytes, filamentous algae and submerged marginal vegetation (Pusey et al., 2004a). *Craterocephalus* spp. show preference for slow-flow habitats and have been found to occur in slow-flowing rivers, small streams, billabongs, swamps, ponds and reservoirs, but are also found in fast-flowing creeks (Allen and Leggett, 1990; Allen et al., 2002). *Craterocephalus* spp. occur in a range of water velocities up to 0.6 m/sec, with the majority found in less than 0.3 m/sec velocity waters (Pusey et al., 2004a). Individuals found in higher velocities were found low in the water column or in association with some form of cover. *Craterocephalus* spp. show a strong preference for slow-velocity waters. Pusey et al. (2002) examined habitat use of larval *C. s. stercusmuscarum* in the Johnstone River, Queensland and found no preference for water depth, but found the species strongly restricted to water velocities less than 0.1 m/sec. *Craterocephalus* spp. are abundant in streams that periodically become disconnected after extended periods of low flow (Pusey et al., 2004a). *Craterocephalus* spp. rapidly recolonise previously dry river stretches after wet-season flows resume and longitudinal connectivity is re-established (Pusey et al., 2004a).

*M. splendida* (*M. s. splendida and M. s. inornata*) are widespread and locally abundant species in catchments of the Northern Territory and Queensland (Allen et al., 2002; Morgan et al., 2011b). An ecomorphologically similar species, *Melanotaenia australis*, is common throughout the Fitzroy River (Allen et al., 2002; Morgan et al., 2011b). *Melanotaenia* spp. have been found to be abundant where they occur and electrofishing surveys are likely to under estimate their population sizes (Pusey et al., 2004a). The suggested preferred habitat of *Melanotaenia* spp. are small streams (Allen et al., 2002), however *Melanotaenia* spp. have been found to occur in a variety of habitats including large, lowland rivers, wetlands and floodplain lagoons (Kennard, 1995; Pusey et al., 2004a; Pusey and Kennard, 1996). They show a preference for habitats with low flow (Pusey et al., 1998) and exhibit generalist habitat associations. *Melanotaenia* spp. are predominately an open-water species, but will retreat to cover when threatened (Pusey et al., 2004a). Their eggs have adhesive filaments and are deposited on leaves of aquatic vegetation (Pusey et al., 2004a). Larvae remain in low-flow environments until metamorphosis is completed, and flow refuges (eddies and backwaters) are important in reducing flow-related mortality of larvae (Pusey et al., 2002).

**Conceptual model**

Figure 3-69 shows the conceptual model for stable flow spawning fish species in northern Australia, with the accompanying narrative given in Table 3-16.
Figure 3-69 Conceptual model illustrating the relationship between threats, drivers, effects and outcomes for stable flow spawning fish species in northern Australia
### Table 3-16 Knowledge underpinning the conceptual model of stable flow spawning fish species in northern Australia

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<th>1. HABITAT AND THREATS</th>
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<td>Floodplain wetlands - Aquatic macrophytes Climate change and Change in intensification in land use and management</td>
<td>Saltwater intrusion, greater variability in climate</td>
<td>Aquatic macrophytes are an important food source for some fish species (<em>Nematalosa erebi</em>). Provide important refuge habitat for small-bodied fish species during high flows by reducing focal velocities (<em>Craterocephalus</em> spp.). Habitats with abundant macrophyte cover are preferred among many stable flow spawners (<em>Craterocephalus</em> spp., <em>Glossamia aprion</em>). Aquatic macrophytes are also important for spawning (<em>Strongylura krefftii</em>) and feeding (<em>Strongylura krefftii</em>, <em>Glossamia aprion</em>). <em>Nematalosa erebi</em> is only stable flow spawning fish species where macrophytes have not been suggested to be important habitat (Pusey et al., 2004a). However, macrophytes can be an important food source for <em>Nematalosa erebi</em>.</td>
<td>Loss of aquatic macrophyte cover and ability of fish to persist in saline waters. The impacts of climate change include an increase in extreme events and sea-level rise (Catford et al., 2013). Water resource development is likely to reduce flows. Reduced flows coupled with sea-level rise and increased storm intensity is likely to increase the extent and influence of saltwater intrusion. Saltwater intrusion kills and slows the growth of many salt-intolerant plant species (Catford et al., 2013). Floodplains are unable to migrate inland as sea levels rise and become vulnerable to the impact of saltwater intrusion (Catford et al., 2013). Stable flow spawners occupy freshwater habitats and would be classified as salt-intolerant fish. Some species such as <em>Amnitaloba percoides</em> are likely to be salt-tolerant but have only been recorded in freshwater (Pusey, 2000). Floodplains are a crucial habitat occupied by all stable flow spawners at different life-history stages. Saline water and the loss of freshwater floodplain vegetation would adversely affect the ability of fish species to persist in these floodplains.</td>
<td>Winn et al. (2006) found that the tidal creeks of the East Alligator rivers Study area of the Northern Territory has already extended 4 km inland. Bare saline mudflats on the coastal plain in 2000 had undergone a nine-fold increase and 64% of the original salt-intolerant <em>Melaleuca</em> forest had been lost (Winn et al., 2006). It is likely that saline mudflats have continued to extend inland and the extent of the original <em>Melaleuca</em> forest has continued to decrease since 2000. In the Mary River wetlands, a transition from a salt-intolerant semi-aquatic vegetation community to a saline swamp and a salt-tolerant vegetation community has already been observed (Mulrennan and Woodroffe, 1998). Saline water makes floodplain soils more dispersive which will continue to escalate the extent of saltwater intrusion by extending seawater channels (Winn et al., 2006). It is likely that over half of the floodplain wetlands of northern Australia will be inundated with saltwater over the next 50 years (Bartolo et al., 2008; Catford et al., 2013). However, it is difficult to quantify the extent of saltwater intrusion in northern Australia.</td>
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| Habitat degradation and loss of riparian structure (proliferation of weeds). | Floodplain wetlands and lentic channel habitats are important habitats occupied by stable flow spawners. These habitats are characterised by zero or low flows and abundant macrophyte cover or flanked by marginal vegetation (riparian areas). The clearing of riparian vegetation reduces canopy cover and increases light intensity. Increased light intensity facilitates the proliferation of weed species and submerged grasses such as para grass (*Brachiaria mutica*) (Pusey and Arthington, 2003). | The proliferation of exotic pasture grasses leads to changes in habitat structure, water quality and food-web structure which all impact stable flow spawners (Bunn et al., 1998; Bunn et al., 1997; Pusey and Arthington, 2003). | The combination of climate change, water resource development infrastructure and land use change is likely to facilitate both the proliferation and persistence of exotic pasture grasses such as *Brachiaria mutica*. | Climate change entails elevated temperature and CO2 levels, which favour exotic, fire-tolerant, semi-aquatic C4 graminoids (Catford et al., 2013). Para grass (*Brachiaria mutica*) is an exotic species that is widespread in northern Australia. Climate change and changes in land use are likely to facilitate the proliferation of this species | The spread of fire-tolerant *Brachiaria mutica* increases dry-season fuel loads and alters fire regimes in northern Australia (Douglas and O’Connor, 2004). *Brachiaria mutica* threatens large areas of wetland in northern Australia. An altered fire regime facilitates the spread of *Brachiaria mutica* by contributing to a positive feedback loop of fire and grass invasion (Douglas and O’Connor, 2004). The spread of *Brachiaria mutica* actively displaces native plant species and can reduce channel capacity and create anoxic waters. Despite stable flow spawners favouring habitats with abundant macrophyte cover, submerged C4 grasses (i.e. *Brachiaria mutica*) do not provide the same ecosystem function that native macrophytes provide. The living tissue of submerged C4 plants contribute little to aquatic food webs (Bunn et al., 1997; Pusey and Arthington, 2003). Few stable...
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<td><em>Amniotaba percoides</em> have all been found to be infrequently associated with reaches dominated by <em>Brachiaria mutica</em> (Pusey et al., 2004a). An increase in the proliferation and persistence of exotic pasture grasses is likely to reduce and degrade available habitat which would adversely affect stable flow spawners. Flow spawning fish species consume large amounts of detritus. <em>Nematolosa erebi</em> is the only exception where detritus is a key part of their diet. However, <em>Nematolosa erebi</em> do not actually consume detritus material but rather the microalgae deposited on detritus material (Pusey et al., 2004a). <em>Brachiaria mutica</em> leaf material collects on the streambed and remains largely unprocessed by secondary consumers (Pusey and Arthington, 2003). The leaf material forms a thick anoxic layer and alters water quality parameters and reduces available habitat for periphyton and macroinvertebrates (Bunn et al., 1997; Pusey and Arthington, 2003). The combination of reduced canopy cover by means of riparian clearing for development (i.e. replacement of rainforest trees and shrub species with pasture grasses and sugarcane croplands) and reduced flows is likely to favour the spread and persistence of exotic pasture grasses such as <em>Brachiaria mutica</em>. The proliferation of pasture grasses such as <em>Brachiaria mutica</em> degrades instream habitat by trapping sediment and channelising flows (Pusey and Arthington, 2003). Once a channel becomes contracted by the proliferation of pasture grasses, low-frequency, high-intensity flood events are required to re-establish normal channel dimensions (Bunn et al., 1998; Pusey and Arthington, 2003). Impoundment infrastructure (i.e. dams, weirs) is likely to reduce the intensity of natural high-intensity flood events. These flood events are required to re-establish channel dimensions after the onset of pasture grass invasion. The proliferation of exotic grasses can favour other exotic fish species and alter water quality parameters (Arthington et al., 1983; Catford et al., 2013). Invasive species such as <em>Gambusia holbrooki</em> and <em>Oreochromis mossambicus</em> have been found to favour the proliferation of littoral grasses and reduced canopy cover (Arthington et al., 1983).</td>
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<tr>
<td>Habitat degradation and loss of riparian structure (sedimentation of rivers).</td>
<td>Direct effects on preferred habitats and water quality and indirect effects on preferred food sources.</td>
<td>Increased sedimentation can directly and indirectly affect stream fish. Increased turbidity and sedimentation impact on fish by altering food availability, reducing foraging behaviour and efficiency and altering intraspecific interactions (Pusey and Arthington, 2003). Sedimentation can also degrade available habitat for spawning, foraging and refuge.</td>
<td>The effects of sedimentation are likely to be felt gradually by stable flow spawners as a press disturbance. Despite stable flow spawning fish species being more tolerant of high turbidity levels, the indirect effects on food sources (micro/macroinvertebrates) is likely to have adverse impacts (Douglas et al., 2005).</td>
<td>Increased sedimentation can directly affect the growth of macrophytes (Douglas et al., 2005). Increased sedimentation can reduce available habitats for macroinvertebrates by clogging interstitial spaces between substrate particles (Douglas et al., 2005). Reid et al. (2017) found that increased sediment rates associated with human settlement reduced the persistence times of refugia waterholes in the Lower Balonne River system during low and no-flow periods by 2–4 months.</td>
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| Habitat degradation and loss of riparian structure (increased nutrient input). | Change in nutrient input into ecosystem and change in primary productivity. | A change or intensification of land use is likely to alter the exchange of organic material between terrestrial and aquatic ecosystems. | Increased nutrient input into freshwater systems can have varying impacts on stable flow spawners. For example, increased nutrient input coupled with an increase in light intensity (through riparian clearing) could result in the proliferation of exotic pasture grasses such as Brachiaria mutica. However, an increase in primary productivity could also facilitate the growth of aquatic macrophytes which is the preferred habitat of many stable flow spawners. The impact on stable flow spawning species would be dependent upon whether this increase in primary productivity facilitates macrophyte growth (and the Tropical streams often have low nutrient status and northern Australia river systems are likely to be sensitive to even small changes in nutrient load from urban/agricultural development (i.e. fertiliser runoff and manure) (Douglas et al., 2005). High light environments (riparian clearing) and elevated water temperatures (climate change) are likely to significantly increase primary production with nutrient addition (Douglas et al., 2005). Effects on aquatic systems would be dependent upon whether this increase production contributes to palatable (i.e. microalgae) or unpalatable forms (i.e. vascular plants or toxic algae) of biomass. The effects of increased nutrient load would be most apparent in the dry season because of low discharge, concentrations of nutrients and turbidity (Douglas et al., 2005). When nutrients are not limiting and light intensity is high, aquatic macrophytes can proliferate and have similar effects to invasive weeds (Pusey and Arthington, 2003). Therefore, despite aquatic macrophytes being important habitat for stable flow spawning species, an intensive proliferation of macrophytes can negatively impact fish by reducing habitat diversity and depressing secondary production (Douglas et al., 2005). However, negative impacts are likely to be more pronounced on fish species who are not as reliant on... |
Offchannel, floodplain and ephemeral habitats (dry-season refuges).

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<tr>
<td>Changes in wet-season inundation and flooding. Change in nutrient input into ecosystem and change in primary productivity. Restricted movement and dispersal of fish.</td>
<td>Changes in the extent of inundation of offchannel and wetland habitats.</td>
<td>Water resource development in likely to change the extent of floodplain and offchannel habitat inundation.</td>
<td>Despite stable flow spawner’s not requiring rising water to initiate spawning, recruits do benefit from exploiting the nutrient-rich floodplains during the wet-season. A reduction in the extent of wetland inundation would increase competition between species due to the decrease in available resources and space.</td>
<td>Extent of growth) or the proliferation of weeds.</td>
<td>High light and nutrient availability favour the growth of filamentous algae (Bunn et al., 1999; Pusey and Arthington, 2003). Filamentous algae is an important component of the diets of some stable flow spawning species such as M. s. splendida, Nematalosa erebi and Amniataba percoides (Pusey et al., 2004a). However, most fish species display a flexible feeding strategy that differs depending on environmental conditions. All three species will consume filamentous algae during adult stages and times of low flow. Juvenile life-history stages are characterised by a dependence on insect prey and are likely to be disadvantaged by conditions of high algal abundance (Pusey et al., 2004a; Pusey and Arthington, 2003).</td>
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<td>Change to dry-season water persistence (connection). Change in system connectivity.</td>
<td>Change in connection and disconnection of off-channel habitats to riverine systems.</td>
<td>The dry season is a period of naturally low flows, water extraction will further reduce discharge and extent of longitudinal, lateral and vertical connectivity.</td>
<td>Decreased dispersal and range of stable flow spawning species. Increase in competition between species due to the decrease in available resources and habitat.</td>
<td>Lower dry-season flows may limit the longitudinal movement of aquatic biota (King et al., 2015). Low flows may reduce the ability of fish species to disperse and seek suitable dry-season refugia with the onset of the annual contraction of intermittent and ephemeral rivers in the wet–dry transition period (Bishop et al., 1995; King et al., 2015).</td>
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<td>Change to dry-season water persistence (dry-season refugia)</td>
<td>During the transition from wet to dry-season, floodplain water contracts. Remaining surface water provides important refuge habitat for many stable flow spawning species such as Nematalosa erebi, Craterocephalus spp. and Melanotaenia spp.</td>
<td>Extraction of water results in reduced size, number, quality and persistence of remaining surface water (Warfe et al., 2011).</td>
<td>Decreased ability of stable flow spawning species to persist in dry-season refugia habitats. Increase in competition between species due to the decrease in available resources and habitat.</td>
<td>Pusey et al. (2004b) found differences in fish assemblage structure between two rivers of similar flow dynamics in the same region. The river that contained extensive floodplains and permanent water bodies that functioned as dry-season refugia supported more fish species. Pusey et al. (2004b) suggested that the presence of dry-season refugia is likely to be an important determinant of the extent to which pronounced seasonal reductions in discharge affect the fish fauna.</td>
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<tr>
<td>All habitats</td>
<td>Climate change and invasive species and Change/intensification</td>
<td>Greater variability in climate. Competition for resources. Change in fish assemblage structure. Changes in predation pressure and competition between species.</td>
<td>Increase in competition between species due to the decrease in available resources and habitat.</td>
<td>Increase in competition between species due to the decrease in available resources and habitat.</td>
<td>Higher temperatures usually favour traits possessed by non-Indigenous species (Catford et al., 2013). In a study by Burrows and Butler (2012) the critical thermal maximum (CTmax) of stable flow spawning fish species such as the Amniataba percoides, M. s. splendida and C. s. stercusmuscarum were examined. Test species were found to be tolerant of water temperatures greater than the temperatures usually found in northern Australian waters (Burrows and Butler, 2012). However, barramundi (Lates...</td>
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in land use/management

<p>| All habitats | Change in seasonality of flows. | Flow-related mortality of larvae – reduced recruitment strength. | Stable flow spawning fish species will spawn during the dry season or in times of predictably low flow. Variability in discharge in northern Australia is among the lowest in the country (Douglas et al., 2005). Out of season flows (i.e. to deliver water to an irrigator) would adversely affect stable flow spawning species during spawning periods (Pusey et al., 2004a). | Reduced recruitment strength. The level of effect would be dependent on the life-history strategy utilised by the stable flow spawning fish species. | In northern Australia, wet-season rains arrive almost unfailing and the time period of the wet season is well defined. Variability in annual stream discharge in northern Australia is among the lowest in the country (Douglas et al., 2005). Despite the reliability of annual flood pulses in north Australian systems, there is still significant inter-annual variation in hydrology (Douglas et al., 2005). Stable flow spawning fish species utilise different life-history strategies defined by Winemiller and Rose (1992) including, periodic, equilibrium and opportunistic life-history strategies. While fish are capable of producing hundreds to millions of young, even in a single breeding event, mortality rates generally exceed 95% for most species in these early life stages (McCasker et al., 2014). This high mortality rate means that even subtle changes in early life stage mortality rates is likely to have significant consequences to future year-class strength and the population dynamics of fishes. Small-bodied stable flow spawners are the most susceptible to flow-related mortality (Pusey et al., 2002). Pusey et al. (2002) examined habitat use of larval Craterocephalus s. stercusmuscarum in the Johnstone River, Queensland and found no preference for water depth but found Craterocephalus s. stercusmuscarum strongly restricted to water velocities less than 10 cm/sec. Out of season high flows during spawning periods would adversely affect the survival of Craterocephalus spp. and Melanotaenia spp. larvae and |
| River regulation and Dry-season water extraction | | | | | |
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<td>Change to system connectivity (connection and disconnection of offchannel habitats to riverine systems).</td>
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<td>Change in fish assemblage. Changes in predation pressure and competition between species.</td>
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<td>Changes in the connection and disconnection periods of offchannel and main channel habitats can alter fish assemblages occurring in these habitats. A reduced connection period (through reduced inundation and flows) could hinder the ability of stable flow spawning species to migrate from main channel to offchannel habitats. An increased connection period (through river regulation and introducing permanency of water in a previously ephemeral reach) could facilitate the dispersal of species who would not previously been able to migrate into these habitats. For example, <em>Lates calcarifer</em> exerts a major structuring force on fish community dynamics and behaviour. The introduction of <em>Lates calcarifer</em> (through increased connection) into offchannel habitats</td>
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<td>Changes in the connection and disconnection periods of offchannel and main channel habitats can affect stable flow spawners in two ways. First, a reduced connection time would limit the ability of stable flow spawners to migrate to suitable dry-season refugia habitat and inundated floodplains in the wet season. This could have detrimental effects on stable flow spawners as they would be forced to remain in main channel habitats where competition and predation effects would be greater. In contrast, an increase in the connection time between riverine and offchannel habitats could facilitate the dispersal of species who would normally not occupy offchannel habitats. This could alter fish assemblage structures and increase competition and predation.</td>
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<td>Translocated native species can have the same impacts on resident native species as exotic species, especially when the receiving streams contain communities that have evolved in the absence of predators (Pusey et al., 2006; Pusey et al., 2009). Kennard (1995) found <em>Lates calcarifer</em> to be a major structuring force in fish communities. Kennard (1995) found the presence of <em>Lates calcarifer</em> significantly altered both the foraging behaviour of prey species (i.e. <em>Glossamia aprion</em>) and hence their choice of prey, but also greatly influenced microhabitat use by potential prey species.</td>
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juveniles. However, *Craterocephalus* spp. and *Melanotaenia* spp. are capable of multiple spawning events within a given year. Limited out of season flows are likely to not have a great effect on long-term populations. Out of season flows during spawning periods of periodic and equilibrium life-history strategists would have a greater effect on long-term populations.
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<tr>
<td>All habitats</td>
<td>Competition for resources (invasive species)</td>
<td>Change in fish assemblage structure. Changes in predation pressure and competition between species.</td>
<td>Human development and settlement increases the chances of exotic species introductions (Olden et al., 2008). Changes in the natural environment can also promote the establishment of exotic species by disrupting environmental conditions.</td>
<td>Exotic species introductions, establishments and increased abundance can increase predation pressure and competition between species. Exotic species introductions is likely to adversely affect native species.</td>
<td>Invasive species such as <em>Gambusia holbrooki</em> and <em>Oreochromis mossambicus</em> have been found to favour the proliferation of littoral grasses and reduced canopy cover (Arthington et al., 1983). In a study conducted by Olden et al. (2008) patterns of drainage homogenisation of fish species were highly concordant with levels of disturbance in response to human settlement, infrastructure and land use change. Anthropogenic influences also increase the likelihood of exotic species introductions (Olden et al., 2008). Disturbance associated with land use change and physical infrastructure can also promote the establishment of exotic species by disrupting environmental conditions (Olden et al., 2008).</td>
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<td>Invasive species and River regulation and Climate change and Change in intensification in land use and management</td>
<td>Change to wet-season inundation and flooding</td>
<td>Change to dry-season water persistence</td>
<td>Change in fish assemblage structure (loss of rheophilic species). Changes in predation pressure and competition between species. Shallow, fast-flowing habitat (e.g. riffles) are the most vulnerable to reduced discharge in the low-flow period (King et al., 2015).</td>
<td>Reduced densities of rheophilic taxa and life stages can occur with prolonged low flows that extend beyond their normal life span (King et al., 2015). Most stable flow spawning species show a preference for low-flow, lentic conditions. The loss of rheophilic fish species would benefit stable flow spawning fish species by reducing competition and predation. However, riffle habitats are important for benthic algae and invertebrate production which are a key dietary component for many stable flow spawning species.</td>
<td>Lower flows reduce the volume, area, and depth of available aquatic habitat and change instantaneous velocity of rivers (King et al., 2015; Rolls et al., 2012). A number of studies have reported the loss of rheophilic species with prolonged low flows (Miller et al., 2007) but few examples exist that test rheophilic species’ resistance and resilience to water extraction in tropical environments (King et al., 2015). Riffle habitats are also important for benthic algae and invertebrate production (Douglas et al., 2005).</td>
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<tr>
<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
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<tr>
<td>All habitats</td>
<td>Change to wet-season inundation and flooding</td>
<td>Change in fish assemblage structure (increase in species with an opportunistic life-history strategy).</td>
<td>Many stable flow spawning species benefit from drying conditions however, small-bodied stable flow spawners are particularly favoured.</td>
<td>Low-flow conditions favour stable flow spawning species. Species with an opportunistic life-history strategy (Winemiller and Rose, 1992) are particularly favoured. The stable flow spawning fish community structure is likely to be skewed towards opportunistic species with continued low flows.</td>
<td>Doidge (2014) found that low-flow conditions favoured larvae of species that had opportunistic life-history strategies. Opportunistic species were the most abundant in the Daly River, Northern Territory during the dry season and equilibrium species were well represented. It is clear that specific flow conditions (in this case low flows) favour some life-history strategies over others (King et al., 2013a).</td>
</tr>
<tr>
<td>River regulation and Dry-season water extraction</td>
<td>Change to dry-season water persistence</td>
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*Amniatoba percoides* prefers shallow stream reaches with coarse substrate and flows greater than 0.3 m/sec (i.e. runs) (Pusey et al., 2004a). A reduction in preferred habitat would adversely affect *A. percoides*. *
### 3.3.3 FRESHWATER TURTLES

**Asset description**

Australia contains two families of freshwater turtles, with about 23 species inhabiting the Australian continent, all but one belonging to the Chelidae family. The Chelidae family contains the side-necked turtles, has a Gondwanan origin and is currently endemic to Australia, New Guinea and South America. The sole member of the Carettochelyidae family, *Carettochelys insculpta*, known as the pig-nosed turtle, is geographically restricted to northern Australia and southern New Guinea (CSIRO, 2004; NSW OEH, 2015). In Australia, *C. insculpta* is largely restricted to within the Daly, Flora and Alligator river systems and therefore will not be included in the freshwater turtle impact assessment, but is discussed as part of the freshwater turtle asset description.

All Australian species of freshwater turtles depend upon water for survival. However, some species have a flow dependency to support critical phases in their life history, or have habitat function associated with aspects of the flow regime. The turtle functional group includes species that occur across the three study areas and have in general a wet-season flow requirement. Members of this functional group include species of the genus *Chelodina* and *Carettochelys*. Members of the turtle functional group are of both ecological and cultural significance in Australia. Some species are important culturally, including forming an important harvest by Indigenous people as a seasonal source of protein (Jackson, 2015).

As dispersal by freshwater turtles is limited during dry periods, they are vulnerable to water quality and habitat degradation and loss. *C. insculpta* is listed on the IUCN’s Red List as a vulnerable species in need of specific conservation measures (IUCN, 2016), but populations in Australia are not under the same level of threatening processes.

**Distribution in Assessment study areas**

Figure 3-70, Figure 3-71 and Figure 3-72 show the distribution of *Chelodina* species in the three Assessment study areas.
Figure 3-70 Distribution of *Chelodina* species in the Fitzroy catchment
Figure 3-71 Distribution of *Chelodina* species in the Darwin catchments
In northern Australia turtles occupy a range of habitats, including both river and floodplain wetland habitats. Northern freshwater turtle species live by the rhythm of the wet–dry seasons. The wet season (November to April) sees a dramatic rise in water levels and monsoonal rainfall, while the following dry-season droughts limit the overall habitat availability. This pattern is highly variable between years and flood frequency is often unpredictable. Thus, species inhabiting the tropics have often developed adaptive traits to survive in this highly variable environment. Some species, including Cann’s snake-necked turtle (*Chelodina canni*) and the eastern long-necked turtle (*C. longicollis*) can survive for months in aestivation buried in soil. The northern snake-necked turtle (*C. rugosa* (*C. oblonga*)) has the capacity to lay eggs underwater or mud during the late wet season. *C. insculpta* embryos, once fully developed, enter aestivation and wait for the following wet-season rains and inundation, which act as a trigger for hatching. This way hatchlings can immediately seek shelter in waters where they will be able to feed and be safe from predators.

Changes in flow timing, velocity, persistence and extent can have important consequences on freshwater turtles’ ecological processes and habitats, including disrupted breeding cues, loss of feeding or nesting grounds, hampered migration to shelter habitats. Turtle species
and populations from eastern Australia and elsewhere are often more understood than those of northern Australia, and have often informed the bases of inferred understanding of turtle flow requirements and responses to flow for turtle species in northern Australia. The limitations and assumptions associated with this will be considered throughout the assessment process.

**Conceptual model**

Figure 3-73 illustrates the conceptual model for freshwater turtles in northern Australia, with the accompanying narrative given in Table 3-17.
### Conceptual model narrative

#### Table 3-17 Knowledge underpinning the conceptual model for freshwater turtles in northern Australia

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<th>1. HABITAT AND THREATS</th>
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<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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<tbody>
<tr>
<td>Floodplain wetlands</td>
<td>Changes to wet season: inundation/flooding regime</td>
<td>Timing of monsoonal rainfall and flooding important in maintaining ecological processes.</td>
<td>Alteration of flood dynamics can disrupt cues for nesting and hatching especially for <em>C. insculpta</em> (Warfe et al., 2011). Nesting dates depend on weather and temperatures, which influence the timing of eggs’ maturity. The trigger for hatching for <em>C. insculpta</em> is inundation of the eggs or first torrential rains of the wet season (Jolly et al., 2002).</td>
<td>Reduction in either rainfall or flood peaks, or changes in flood timing leading to reduced breeding success and population size of <em>C. insculpta</em> (Warfe et al., 2011). Early nesting and egg maturity result in the eggs perishing in the ground, while late nesting risks the eggs to be prematurely flooded by rising waters (Jolly et al., 2002).</td>
<td>Investigation of <em>C. insculpta</em> ’s nesting timing and site during the nesting seasons of 1996-1998 on the Daly River revealed that turtles nested earlier following big wet seasons because of better food uptakes (Doody et al., 2003). This same investigation revealed that nests survival to flood episodes was explained by lay date and height of the nest site (Doody, 2002).</td>
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<tr>
<td>Extraction/ regulation and Climate Change</td>
<td>Instream water storage and barriers can result in reductions in floodplain inundation. This can lead to disconnections, hampering movements on and off the floodplain during the wet season to dry-season transition (Warfe et al., 2011). Reduced chances of reaching freshwater shelter for the dry season (Warfe et al., 2011).</td>
<td>Reduction in either rainfall or flood peaks, or changes in flood timing leading to reduced breeding success and population size of <em>C. insculpta</em> (Warfe et al., 2011). Early nesting and egg maturity result in the eggs perishing in the ground, while late nesting risks the eggs to be prematurely flooded by rising waters (Jolly et al., 2002).</td>
<td>Reduced survival and population size of freshwater turtles.</td>
<td>Study about the role of connectivity and water level in tropical lagoons showed that disconnected lagoons had relatively stable depths, independent of water level oscillations in the main rivers (Dos Santos and Thomaz, 2007). Study of <em>C. longicollis</em> from 2004 to 2006, in Booderee National Park revealed that some <em>C. longicollis</em> survived extended terrestrial aestivation (up to 480 days) while awaiting re-flooding. But a large proportion (45%; irrespective of adult size or sex) died, compared to lower mortality rates (18%) of those that moved to permanent wetlands (Roe and Georges, 2008). This shows how important dry-season refuges are.</td>
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Changes to dry-season During the transition from wet to dry season, floodplain water contracts. Extraction of water results in reduced size, number and Reduced survival and population size of *C. canni* and Sampling of turtles from now regulated Lake Boga revealed about 500 *E. macquarii*
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<tr>
<td>Remaining surface water provides important refuge habitat for freshwater turtles (Warfe et al., 2011).</td>
<td>Water persistence</td>
<td>Chelodina (novaeguineae) canni and C. longicollis are both well-adapted to remote, shallow waters and drought events because of their ability to aestivate into soil and litter or low rate of evaporative water loss (DSITIA, 2014; Chessman, 1988).</td>
<td>Persistence of remaining surface water (Warfe et al., 2011). Reduced inundation of floodplains will increase stress upon C. canni and C longicollis which could be forced to migrate to permanent and stable water bodies with tough competition for resources (DSITIA, 2014; Chessman, 1988).</td>
<td>C. longicollis. (DSITIA, 2014; Chessman, 1988)</td>
<td>Reduced recruitment, survival and population size of freshwater turtles.</td>
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<tr>
<td>Reduced recruitment, survival and population size of freshwater turtles.</td>
<td>Saltwater intrusion</td>
<td>Chelodina rugosa has developed a capacity to lay eggs under shallow flooded wetlands at the end of the wet season in order to use wet-season energy reserves into reproduction and potentially produce more clutches; this chance can be lost if nesting is delayed until the end of wet season (Kennett et al., 1993a, b).</td>
<td>Prolonged flooding prevents the development of C. rugosa’s underwater laid eggs. Embryos survive immersion in a state of arrested development, waiting for the ground to dry with the dry-season to commence its development. However, they will die if immersed after embryonic development started (Georges et al., 1993).</td>
<td>Prolonged flooding events reduce breeding success and population size of C. rugosa.</td>
<td>Radio-tracking of gravid females of C. rugosa confirmed underwater nesting (Kennett et al., 1993 a, b).</td>
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<tr>
<td>Radio-tracking of gravid females of C. rugosa confirmed underwater nesting (Kennett et al., 1993 a, b).</td>
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<td>A series of immersion experiments on C. rugosa in laboratory showed that embryos can survive immersion for 12 week in experimental conditions. One experiment resulted in all eggs dying when they were immersed after the development commenced (Georges et al., 1993).</td>
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<td>Seawater intrusion has already occurred in some areas of the Northern Territory (e.g. Mary River). The possible effects on floodplain habitat are based on these experiences of seawater intrusion (Catford et al., 2013).</td>
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<td>Rogers noted that after 3 to 4 days of dehydration of C. longicollis individuals, large tears appeared leaving a white crust when dry, suggesting the presence of lachrymal salt glands which was confirmed by dissection (Rogers, 1966).</td>
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<tr>
<td>Rogers noted that after 3 to 4 days of dehydration of C. longicollis individuals, large tears appeared leaving a white crust when dry, suggesting the presence of lachrymal salt glands which was confirmed by dissection (Rogers, 1966).</td>
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<td>Emyruda macquarrii individuals were sampled in the confluence of the Murray</td>
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<tr>
<td>Floodplain wetlands</td>
<td>Changes in air temperatures</td>
<td>Temperature-dependent sex determination, whereby the sex ratio of the <em>C. insculpta</em> hatchlings is determined by nest temperatures during incubation</td>
<td>however persistence is unlikely in such environments (Rogers, 1966; Bower, 2011). Salinisation of river banks has the potential to reduce hatching success and morphology of freshwater turtles (Bower, 2011). Invasive marine worms can parasite freshwater turtles’ carapace, inhibiting limb movements (Bower, 2011).</td>
<td>and Darling Rivers then were tested in laboratory for two treatments: freshwater and saline treatments. Turtles successfully survived extended exposure to 15‰ salinity without change in body mass. These results are explained by behavioural strategies: reduced feeding and drinking, retained urine and faeces, or exit from water; which suggests that behavioural constraints may limit their persistence in brackish water as they need feeding. Freshwater turtles cannot survive indefinitely without freshwater (Bower, 2011).</td>
<td>Laboratory quantification of the influence of salt on the development of <em>C. expansa</em> hatchlings for four different salinities showed that embryos incubated in higher salinities had 39% less survival than those incubated in substrates with freshwater. Hatchlings were also smaller and lighter (Bower, 2011). Sampling of turtles in the lower Murray River showed <em>C. longicollis</em> as the only species encountered in the saline site during the study. All individuals were parasitised by the marine worm <em>Ficopomatus enigmaticus</em>, which encased 100% of the carapace. Apparently, <em>Emydura macquarii</em> were also found encased with the marine worm prior to the study (Bower, 2011).</td>
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<tr>
<td>Climate change</td>
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<td>Investigation on the Daly River of <em>C. insculpta</em>’s ecology and sex determination proves TSD for this turtle. 102 nests’ temperatures were monitored and eggs near-term were taken from the clutch to laboratory to proceed to sex</td>
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Climate change

Temperature-dependent sex determination, whereby the sex ratio of the *C. insculpta* hatchlings is determined by nest temperatures during incubation

Climate warming could bias offspring sex ratio (Schwanz and Janzen, 2008). Proximate shifts in climate temperature are expected to be rapid, which may preclude successful gradual responses that

Altered sex ratios could affect population demographics and persistence. This species could suffer local extinction (Schwanz and Janzen, 2008).
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<tr>
<td>Floodplain wetlands</td>
<td>Feral pigs</td>
<td>During the dry season, turtles often move to the shallows to aestivate. Weeks immediately before drying are the riskiest in terms of predation on turtles. Feral pigs will also degrade aquatic ecosystems (Doupé et al., 2009).</td>
<td>functioned historically, like active modification of geographic range.</td>
<td>Reduced recruitment, survival and population size of freshwater turtles.</td>
<td>In areas of high pig abundance, pig predation accounts for 96% of <em>C. rugosa</em> fatalities in dry years (carapace length &gt;100 mm; Fordham et al., 2008). The influence of pig predation on <em>C. rugosa</em> was investigated using a stage-based matrix population model. Turtle predation by pigs at levels higher than 40% of the total population of turtles with carapace length larger than 100 mm is predicted to cause extirpation or substantial population within 50 years. (Fordham et al., 2008). Four successive sampling of the Laura lagoons in Cape York Peninsula revealed the different effects on the ecosystem (sediment upheaval, vegetation destruction, creation of anaerobic and acidic conditions). But the effects of feral pig foraging on freshwater turtle habitat have not yet been quantified. It is still</td>
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<td>Introduced species</td>
<td><em>Sus Scrofa</em></td>
<td>Presence of introduced feral pigs represents a high risk of predation on turtles and eggs (Pusey and Kennard, 2009; Fordham et al., 2006). Feral pigs also have an undesirable effect on turtles' habitat by upheaval of sediments, the destruction of aquatic vegetation, creation of anaerobic and acidic conditions, and enrichment wetlands with nutrients: Turbid conditions would limit visibility compromising hunting opportunities. Destruction of vegetation significantly alters production and respiration regimes, causing anoxic conditions and pH imbalances (Doupé et al., 2009).</td>
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(Doody et al., 2002).

A long-term study on the phenotypic plasticity of a population of painted turtles (*Chrysemys picta*) with TSD in the United States concluded that phenotypic plasticity observed in female painted turtles seemed unlikely to prevent future biased sex ratios caused by climate change. They hypothesise that this conclusion applies to other populations with TSD (Schwanz and Janzen, 2008).

Determination: cooler temperatures produced males, intermediate temperatures (around 32°C) produced mixed sexes, and hotter temperatures produced females (Doody et al., 2002).
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<th>EFFECT</th>
<th>OUTCOME</th>
<th>EVIDENCE AND PATHWAYS</th>
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<tr>
<td>Floodplain wetlands</td>
<td>Cane Toads (Rhinella marina)</td>
<td>Glands on Can toads’ shoulders exude a very toxic poison which can kill most creatures that consume it. Eggs and tadpoles are also toxic if ingested (Thomson et al., 2011; Phillips and Shine, 2005).</td>
<td>Tadpoles, eggs, juveniles and metamorphs are likely prey items for many species, particularly those frequenting still water where toads are likely to breed. Turtles of the genus <em>Chelodina</em> are the best equipped with their long neck to hunt animal prey which put them at higher risk (Smith and Phillips, 2006).</td>
<td>Reduced survival and population size of freshwater turtles.</td>
<td>Experimental study by Smith and Phillips (2005) indicates that <em>C. rugosa</em> is reasonably susceptible to cane toad’s toxin. But data addressing the effect on wild populations are lacking for most species (Smith and Phillips, 2006). Hamley and Georges (1985) would have succeeded in maintaining several Saw-shelled turtles (<em>Elseya latisternum</em>) on a toad’s diet for several months with no apparent ill effects. This suggests that this species could have a much higher resistance to toads’ toxin (Smith and Phillips, 2006).</td>
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<td>Overharvesting</td>
<td>Unsustainable use</td>
<td>Freshwater turtles like <em>C. rugosa</em> and their eggs are harvested by Indigenous people in the tropics as a seasonal source of protein. Harvest only occurs during dry years and is influenced by the frequency and timing of ceremonies/ cultural activities that interfere with harvests. Thus, specific wetlands would be harvested in no more than 50% of dry years (Fordham et al., 2008).</td>
<td>Harvesting of freshwater turtles results in low sub-adult and adult turtles survival (Fordham et al., 2007a, b). Although harvesting rates in Australia are not currently considered as threatening to turtles’ survival, the situation is different elsewhere. In Papua New Guinea, there is a substantive decline in populations of <em>C. insculpta</em> (Eisemberg et al., 2011).</td>
<td>Reduced recruitment and survival and reduced population size of harvested freshwater turtle.</td>
<td>Study of variations in demographic parameters in <em>C. rugosa</em> across waterholes subject to varying levels of harvest demonstrate clearly that there is a compensatory increase in hatching survival. It is sufficient to allow annual stage-specific harvests of up to 20% of sub-adult and adult <em>Chelodina rugosa</em>, without causing extirpation or substantial population suppression. However, this species is a fast-growing, early maturing and highly fecund species, with density-dependent compensatory growth and survival of juveniles. Turtles which do not possess these traits would have slower responses to increased mortality rates (Fordham et al., 2007a, b). In Papua New Guinea, a study combined matched village and market surveys separated by 30 years, trends in nesting</td>
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<td>River channels</td>
<td>Extraction/ regulation</td>
<td>During the dry season, perennial rivers baseflow is maintained by groundwater discharge (Warfe et al., 2011). During the transition from the wet to dry seasons, intermittent rivers contract to waterholes. These provide important refuge habitats for freshwater turtles (Warfe et al., 2011).</td>
<td>Reduced dry-season baseflows (by extraction) in perennial rivers could reduce the availability of flow sensitive habitats for turtles. It could even shift the rivers from perennial to intermittent status (Warfe et al., 2011). Permanent water specialist <em>C. insculpta</em> requires movement to feed and access suitable sandbanks for nesting (because unadapted to movements on land). Thus reduction in perennial baseflow can create disconnections hampering movements and nest choices (Jolly et al., 2002).</td>
<td>Reduced nesting success and population size of <em>C. insculpta</em> (Erskine et al., 2003).</td>
<td>Modelling of <em>C. insculpta</em> passage through the study reach of the Daly River (from Claravale Crossing to the confluence with Cattle Creek) defined breakpoints as points which would restrict the movement of turtles along the Daly River longitudinal continuum: At a discharge greater than 16.1 m³ s⁻¹, and maximum flow depth always exceeding 0.5 m at Dorisvale gauge, there would be no barriers to the free movement and nesting of turtles. Minimum streamflow less than 6.2 m³s⁻¹ were classified as ‘bust’ conditions because of restricted home range, restricted access of nesting banks and feeding grounds... (Erskine et al., 2003).</td>
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<tr>
<td>Extraction/ regulation</td>
<td>Changes to dry-season water persistence</td>
<td>Extraction of water and reduction in flows will reduce the standing crop of benthic macroalgae like <em>Spirogyra</em> and <em>Vaucheria</em> which represents 30% of the dry-season diet of <em>Elseya dentata</em> (Kennett and Tory, 1996). Macroalgae also have a role in rivers’ nutrients dynamics through mineralisation, storage, and uptake; thus they have a</td>
<td>Reduced survival and population size of freshwater turtles. (Townsend and Padovan, 2009).</td>
<td>Modelling of impact of reduced dry-season baseflow, based on two extraction scenarios, revealed a maximum potential standing crop (MPSC) under the historical minimum in both cases. The fixed-extraction scenario had the greatest relative impact on the MPSC of naturally low flows, while the proportional-extraction scenario had a greater relative impact on</td>
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<td>significant importance in rivers ecology (Townstend and Padovan, 2009).</td>
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<td>the MPSC of high flows (Townstend and Padovan, 2009).</td>
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<td>Comparison of two lakes’ summer standing crop per unit area (Lake McKerlzie and Lake Windermere) confirms that growth and body condition are related to food availability; growth was slower, and the turtles were in poorer condition in the lake with less food availability (Kennett and Georges, 1990).</td>
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<td>Flushing and collecting of stomach contents of 47 individuals of <em>Emydura macquarii</em> in the Murray River showed that <em>Cladophora</em> and <em>Spirogyra</em>, were present in 66% of the turtles’ stomachs with a volume of 61% of their diet (Spencer et al., 1998).</td>
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<td>The same method was used to determine that about 60% of <em>Elseya dentata</em>’s stomachs contained <em>Spirogyra</em> and <em>Vaucheria</em> during the dry season with a volume of 30% of their dry-season diet (Kennett and Tory, 1996).</td>
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<td>Extraction of water results in reduced size, number and persistence of waterholes; and delayed waterholes reconnection (Warfe et al., 2011). This goes hand in hand with high turtle density which causes competition for decreasing resources, resulting in turtles no longer directing their energy into reproduction and growth. High abundance of turtles can constraint the survival of hatchlings either due to direct predation or resource Reduced recruitment, survival and population size of freshwater turtles. Sampling of <em>Emydura macquarii kreffitii</em> in the Ross River with high density of population was compared with the Townsville Creek with low density. The Townsville Creek’s populations had a large number of juveniles whereas Ross River had few of them (Trembath, 2005). Mark-recapture program of <em>C. longicollis</em> (between 1983 and 1987) in Lake McKenzie, Lake Windermere, and in Ryan’s Swamp showed a relationship between food availability and population dynamics. Turtles congregating in permanent water bodies during dry periods could become...</td>
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<tr>
<td>Competition between adults and juveniles (Trembath, 2005).</td>
<td>Dams, weirs, locks can increase dry-season flow to maintain a permanent low flow throughout the dry season (Kingsford, 2000).</td>
<td>Prolonged increases in water levels can reduce light availability for some macrophyte populations like Vallisneria nana that favours shallow waters. Lack of light availability results in decomposition and death of those macrophytes (Marshall et al., 2015) and destruction of a key habitat for C. insculpta (feeding and refuge area) (Rea et al., 2002).</td>
<td>Reduced survival and population size of C. insculpta.</td>
<td>Welsh et al. (2017) found that 74% of the total mass of adult C. insculpta’s diet is comprised of Vallisneria during the dry season in the Daly River (Rea et al., 2002; Doody et al., 2002). Studies were conducted prior to, during and following the construction of a new weir on the Burnett River showing that V. nana dies within 6–9 weeks if it is held below its light compensation depth following inundation; Corresponding to a water level increase of less than 1m above the plants in the Burnett River (Duivenvoorden, 2008).</td>
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Changes in flow regime/velocity | Freshwater turtles (either omnivorous, herbivorous or carnivorous) rely on the aquatic ecosystem and its surroundings for foraging (Tucker et al., 2012) | Impoundments by dams or weirs regulate river flows and replace currents by still-water conditions. This participates in declines in aquatic complexity and biodiversity (especially declines in aquatic macrophytes and invertebrates). Long-lived freshwater turtles respond slowly to changes in its environment. Demographic effects may not be significant until many years after creation of impoundments (Tucker et al., 2012; Waltham et al., 2013). | Reduced survival and population size of freshwater turtles (Tucker et al., 2012). | Diets of three turtles (Elseya albagula-herbivorous, Myuchelys latisternum-carnivorous, Emydura krefftii-omnivorous) in Queensland were studied from free-flowing or impounded rivers to evaluate the effects of flow regulation. Results demonstrate that turtle food webs are altered by river regulation. Compared to river samples, diets in impoundments were substantially reduced in prey abundance, species richness, and dietary breadth (Tucker et al., 2012). |
<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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<tbody>
<tr>
<td>Sandbars, river banks and beaches</td>
<td>Sandbars, river banks and beaches habitat reduction</td>
<td>Necessary nesting habitat for most species during the dry season.</td>
<td>High dry-season flow reduces/eliminates emergent sandbars, (Tracy-Smith, 2006) affecting the availability of favoured nesting habitat, probably leading female turtles to seek alternate, less suitable habitat (e.g. levees) (Bodie, 2001).</td>
<td>Reduced breeding success, survival and population size of freshwater turtles.</td>
<td>In the US, sandbars and beaches along the Missouri River have almost entirely been eliminated. The reduced population size of few local freshwater turtles were attributed to this loss of habitat (Bodie, 2001). Nineteen sandbars of the lower Missouri River were analysed, mapped, measured with morphometric variables to evaluate the potential effect of sandbars availability on softshell turtles (Tracy-Smith, 2006). It was concluded that High dry-season flow reduced breeding success.</td>
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<tr>
<td>Impoundment, regulation, channelisation</td>
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<td>Changes to dry-season water persistence</td>
<td>Eggs are laid at different distances from the shore according to species, making them more or less vulnerable to fluctuating water levels.</td>
<td>Fluctuating water levels as a result of water management can inundate freshwater turtle’s nests resulting in egg mortality (Waltham et al., 2013).</td>
<td>Reduced breeding success and population size of freshwater turtles (Georges et al., 1993).</td>
<td>Sampling of eggs of <em>C. rugosa</em> and <em>C. longicollis</em> and laboratory tests of incubation for different periods of times showed the incapacity of <em>C. longicollis</em> eggs to withstand immersion. <em>Chelodina rugosa</em> is the only known species which eggs can survive immersion. Freshly laid eggs of <em>C. longicollis</em> died when immersed for longer than one week and eggs of both species died when immersed after post-laying embryonic development had commenced (Georges et al., 1993).</td>
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3.4 Category: Processes

3.4.1 FLOODPLAIN AND INCHANNEL PRODUCTIVITY

Asset description

The production and supply of basal carbon resources (i.e. the organic matter at the base of the food web) and the transfer of carbon and nutrients from basal resources up through the food web to animals, such as fish and birds, is of fundamental importance in determining the viable populations of these species. It is also important for understanding the resilience and biogeochemical function of ecosystems.

In river–floodplain systems, basal resources come from imported terrestrial organic matter, such as leaf litter, soil and dissolved organic carbon (allochthonous carbon), and from primary production by plants and algae within the river system itself (autochthonous production).

The supply of terrestrial carbon to river systems is influenced by:

- land use and land management (which affects anthropogenic carbon sources, as well as erosion and delivery of soil to streams)
- riparian and floodplain vegetation (which affects the supply and nature of leaf litter and detrital plant material, as well as influencing soil carbon content and erosion rates)
- groundwater flow and surface runoff (which transport dissolved carbon and particulate material, such as leaf litter, into rivers and wetlands)
- hydrological connectivity (which affects the extent to which water-dependent animals can access floodplain and wetland resources).

Flying insects can also be a source of terrestrial carbon to aquatic food webs. Insects and other animal carbon can be a lower volume, but higher quality, food source than terrestrial plant material.

Douglas et al. (2005) presented evidence that river and wetland food webs in Australia’s tropical rivers depend strongly on algal production (not terrestrial carbon or carbon from aquatic macrophytes) and that seasonal hydrological features strongly influence productivity. The production of autochthonous carbon is influenced by water quality and water clarity as well as substrate type, wetted area, water velocity and depth. River discharge controls how much of this material is carried out to sea, and how far flood plumes extend into marine waters. This in turn can strongly influence coastal marine productivity, including fishery productivity.

Basal carbon can enter the food web to drive higher-level production (such as production of fish, waterbirds and iconic species), or enter the microbial loop and release carbon from the
system in the form of greenhouse gases, or be buried in sediment or otherwise lost from the system. The balance between these pathways is determined by:

- food quality (e.g. C:N:P, fatty acid content and cellulose content)
- environmental conditions (e.g. light, temperature, pH, oxygen concentration and water velocity)
- timing (e.g. availability in synchrony with fish and bird breeding events versus out of season)
- food-web structure.

All of the above factors are influenced by hydrology.

**Distribution in Assessment study areas**

Ecosystem productivity is relevant across all freshwater and marine aquatic habitats, in all systems (Table 3-18).

**Table 3-18 Current status of ecosystem productivity in northern Australia**

<table>
<thead>
<tr>
<th>RIVER</th>
<th>FRESHWATER</th>
<th>MARINE</th>
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<tr>
<td>Fitzroy</td>
<td>Heavy grazing in the upper Fitzroy catchment has increased erosion and hence soil delivery to the river, however a diverse fish (Morgan et al., 2004) and macroinvertebrate (Kay et al., 1999) assemblage suggests a healthy level of productivity.</td>
<td>Waters of the northwest shelf (south of the Fitzroy) have been found to be highly productive despite very low ambient nutrient concentrations. This is attributed to active tidal turnover of particulate organic matter stocks, which drives nutrient turnover. Bacterial production is less than half of measured phytoplankton production, implying that most of the autochthonous primary production is going into the food web (Furnas and Mitchell, 1999). The region supports a highly productive prawn fishery (Newman et al., 2011). Mangroves line King Sound, at the mouth of the Fitzroy River (Cresswell and Semeniuk, 2011).</td>
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<tr>
<td>Adelaide</td>
<td>The Adelaide River is mesotrophic to eutrophic, with low ammonium concentrations, but with elevated dissolved and total phosphorus concentrations (Short, 2016), which frequently exceed ANZECC trigger values for rivers of the wet tropics and south-east Queensland (Department of Environment and Heritage Protection, 2009). This suggests a likelihood of elevated phytoplankton and bacterial production and reduced food-web production, compared with pre-European conditions.</td>
<td>Van Diemen Gulf, at the mouths of the Adelaide, Wildman and Mary Rivers, is turbid all year round due to strong tidal mixing. Phytoplankton concentrations are thus relatively low (Blondeau-Patissier et al., 2017) and benthic productivity in this turbid area will also be light-limited. Productivity is probably higher in clearer waters beyond the Gulf. The estuaries and tidal creeks associated with Darwin catchments rivers support significant areas of mangrove production, including around 1000 ha associated with the Wildman River (Mitchell et al., 2007). Mangroves of nearby Darwin Harbour produce leaf litter biomass of up to 1400 g m⁻² yr⁻¹ (Woodroffe et al., 1988). Near-shore coastal areas in the Darwin catchments also support extensive seagrass beds (see seagrass asset description). It may be difficult to disentangle the impacts of the Darwin catchments rivers on marine productivity, due to the extent of flood plumes and mobility of key fisheries species.</td>
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<tr>
<td>Finniss</td>
<td>Heavy-metal contamination in the Finniss River from mining activities in its catchment likely reduced riparian vegetation coverage (Taylor, 2007), altered food-web structure, and reduced at least some fish populations at its peak (Jeffree and Williams, 1980), though fish communities appear to have now recovered (Jeffree et al., 2001). Floating mats of aquatic plants and vegetation have been observed on floodplain billabongs of the Finniss River, supporting primary producers and diverse faunal communities (Hill et al., 1987).</td>
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<td>RIVER</td>
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<tr>
<td>Mary</td>
<td>Townsend (2006) studied the water quality and phytoplankton of Mary River waterholes during the dry season. Phytoplankton biomass was found to increase during the dry season, although this does not necessarily indicate higher productivity. The study did not consider benthic productivity or terrestrial organic matter inputs. Primary productivity in the Mary River is probably nitrogen limited (Townsend and Edwards, 2003). In at least one instance, a large input of terrestrial organic matter with the onset of flow conditions has contributed to a fish-kill event in the Mary River channel, (Townsend and Edwards, 2003), so timing and return intervals of flow events are likely to be important. Whitehead et al. (1990) describes the floodplain vegetation and mangrove communities of the Mary River and as supporting a diverse ecosystem, large magpie goose population, and productive recreational fishery. They suggest that this is threatened by invasive species and salinisation of wetlands. Growth of barramundi in the Mary River has been found to be considerably higher during the wet season than during the dry season (Davis and Kirkwood, 1984), reflecting the high overall system productivity associated with floodplain inundation.</td>
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<tr>
<td>Wildman</td>
<td>Located in Kakadu National Park, the Wildman River is relatively inaccessible for research in comparison with other Darwin River study areas, and hence little information is available about ecosystem productivity in this river, beyond remote-sensing estimates of vegetation cover and related carbon stores (Collins et al., 2009; Mitchell et al., 2007). The estuarine fish assemblage of the Wildman River is reported as being almost identical to that of the Roper River (Larson, 1999), so ecosystem function including productivity may be similar. Floodplain wetlands in this study area are likely to be very productive, supporting extensive coverage of aquatic plants (Cowie, 2003).</td>
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<tr>
<td>Mitchell</td>
<td>Net primary production in the Mitchell River is nitrogen limited while gross primary production is regulated by light and is considerably higher (Hunt et al., 2012). Dry-season food-web production is driven primarily by benthic algae, while terrestrial carbon sources are broken down bacterially. Fish productivity is probably dependent on wet-season floodplain and wetland production (Hunt et al., 2012). Marine productivity in the deeper waters of the Gulf is driven by marine nitrogen fixation rather than nitrogen inputs from rivers (Burford et al., 2009; Moriarty and Odonohue, 1993). Riverine nitrogen loads may be important in the shallow coastal areas, where river input is largely trapped by the coastal boundary layer (Burford and Rothlisberg, 1999; Burford et al., 2009). The coastal area of the Gulf is dominated by vast, highly productive seagrass beds (Burford et al., 2009).</td>
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Net algal production is likely to respond immediately to any increase in catchment nutrient loads associated with water resource development, with a concomitant change in food-web structure.

Juvenile tiger prawn production in the north-east Gulf seems to be mostly based on seagrass organic matter, with little uptake of terrestrial or mangrove organic matter (Loneragan et al., 1997a). Juvenile banana prawns live in mangroves in estuaries and feed on small crustaceans and molluscs in the mangrove and intertidal muds (Wassenberg and Hill, 1993).

Bacterial productivity in the Gulf accounts for about half the seagrass production. Primary production higher in the Bay and reef flat than at the river mouth (Moriarty et al., 1990).

In the short term, floods reduced primary and secondary productivity in a typical Gulf estuary due to the drop in salinity (Burford et al., 2012; Duggan et al., 2014). There was a substantial increase in nutrient loading to the coastal waters as a result of the high water volumes discharged during flood years (Burford et al., 2012).

Coastal salt flats are also important areas of primary production in flood years. Salt flats occupy large areas of land throughout the southern Gulf (Burford et al., 2016).

### Habitat use

Several prominent conceptual models consider drivers of riverine productivity and the roles of different parts of river–floodplain systems. The River Continuum Concept (Vannote et al., 1980) emphasises differing drivers of production along the length of a river channel, with terrestrial inputs dominating in forested headwaters and autochthonous production more important in open waters further downstream. This concept may have some relevance during the dry season for wet tropical rivers of the Darwin catchments, which have their headwaters in forested regions of Litchfield and Kakadu national parks, but has very limited relevance to the Fitzroy or Mitchell rivers.

Generally considered more appropriate for tropical river systems is the flood pulse concept (Junk et al., 1989), which emphasises the importance of seasonal floodplain inundation for river productivity. According to this conceptual model, the aquatic/terrestrial transition zone is the most productive part of a river system and floods, and extending this zone over a large and moving area greatly enhances riverine productivity. Hence, the floodplain and its associated wetlands are the most important habitats for river productivity. Jardine et al. (2012b) found that floodplain productivity is brought back to river channels through fish, which access inundated floodplain carbon resources and then retreat to the river channel and waterholes as flood waters recede.

A lower level of ecosystem productivity is maintained in perennial river channel reaches, waterholes and estuarine areas throughout the dry season. This is essential to maintaining aquatic animal populations between floods.

When flood pulses reach the mouth of a river, river discharge produces a flood plume in coastal waters. Flood plumes provide a dynamic and complex environment for marine productivity. Productivity is temporarily depressed in the turbid water of the inner plume,
but enhanced at the edge of the plume, where terrestrially sourced nutrients drive phytoplankton production, and terrestrial organic matter is available to marine crustaceans and other grazers. Fisheries production is usually elevated following years of high annual river discharge, which is believed to be at least partly due to increased marine productivity, though it may also be attributable to habitat connectivity (Gillson, 2011). Studies of banana prawns show that lack of food availability and osmoregulatory stress due to the low salinity in estuaries drives animals into the deeper waters where fishing occurs (Duggan et al. 2014).

Beyond the flood plume, seagrass (Gullstrom et al., 2002), macroalgae beds and coral reefs are hotspots for marine productivity. Seagrass itself has relatively low palatability, and much of the carbon fixed by seagrasses is ultimately stored in sediments. This may be important in regional carbon balances. Seagrass also provides a substrate for epiphytic algae, which is more readily available to grazing animals and drives food web production. The Gulf of Carpentaria (Poiner et al., 1987) has extensive seagrass beds. Coastal habitats near the mouth of the Fitzroy River have been less well mapped; however, seagrasses, macroalgae and coral habitats have all been recorded in Kimberley coastal waters (Keesing et al., 2011).

More broadly, phytoplankton, especially nitrogen-fixing taxa such as *Trichodesmium*, are important to coastal productivity (Burford et al., 1995). In the Gulf of Carpentaria, nutrients from rivers, and hence enhanced primary production, are largely confined by ocean currents to the immediate coastal zone (Burford and Rothlisberg, 1999; Burford et al., 2009). Coastal productivity has been less closely studied near the mouths of the Fitzroy and Mitchell rivers, but the recently funded National Environmental Science Program Northern Environmental Resources Hub is addressing this in the Mitchell River.

On coastal margins, mangroves and salt flats produce carbon that has relatively low direct food-web uptake. However, salt flats and vegetated wetlands, like seagrasses, provide substrate for epiphytic algae. When physically accessible, epiphytic algae can be a high-quality food source during periods of inundation (Burford et al., 2016).

Ecosystem productivity is an emergent property of net primary production (carbon fixation by plants and archaeophytes minus gross community respiration), net import of organic material (terrestrial carbon subsidies minus exports) and transfers through the food web. Hydrological change will affect freshwater and marine ecosystem productivity in several ways. Figure 3-74 summarises some of the key pathways to change for productivity and ecosystem function.

**Conceptual model**

Figure 3-74 shows the conceptual model for productivity and ecosystem function in northern Australia.
Evidence and narrative for possible pathways of change for ecosystem productivity are presented in Table 3-19. Note that some possible paths of change show effects in opposite directions, depending on different drivers or different flow management options. Effects are grouped by habitat and then by driver, so contradictory effects involving the same supporting system processes may not be grouped together in the table.
<table>
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<tr>
<th>1. HABITAT</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>Mangroves and salt marshes</td>
<td>Climate change and sea-level rise</td>
<td>Submersion of current coastal habitats</td>
<td>Sea-level rise, in combination with competing land uses (development of adjoining land) and/or river regulation, may result in loss of mangrove and salt marsh habitat.</td>
<td>Loss of mangrove and salt marsh production, and loss of associated periphyton production. Altered food-web structures.</td>
<td>Mangroves can keep pace with sea-level rise by shifting inland, but only if sufficient sediment supply is maintained (Lovelock et al., 2015). Salt marshes, similarly, can keep pace with sea-level rise given sufficient sediment supply (Simas et al., 2001). Sediment supply is often reduced by river regulation (Lovelock et al., 2015). Sediment trapping by dams has been extensively documented worldwide, including in northern Australia. The Burdekin Falls Dam in Queensland, for instance, traps 50-85% of the river sediment load (Lewis et al., 2013). Damming of the Ord River is estimated to have reduced the load of sediment reaching Cambridge Gulf from 100,000 ton d⁻¹ to almost zero (Wolanski et al., 2004). Landward migration of mangroves is in some cases limited by local topography and land use (Di Nitto et al., 2014; Wong et al., 2014).</td>
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<tr>
<td>Seagrass and macroalgae beds and coral reefs</td>
<td>Increased marine water depth</td>
<td>Deeper areas of current benthic habitat for corals, sea grasses and some macroalgae receive too little light due to increased water depth. May be ameliorated by migration to newly inundated areas in the case of seagrasses and macroalgae. Corals likely to be replaced with algae or seagrasses as coral reef migration is too slow.</td>
<td>Loss of coral habitats. Possible loss of sea grass beds and some macroalgae beds, depending on details of local morphometry.</td>
<td></td>
<td>Relationships between water depth and seagrass and coral coverage are very well established in the literature. Short and Neckles (1999) and Duarte (2002) review impacts of climate change, including sea-level rise, on seagrasses. Saunders et al. (2013) have implemented models to predict the impacts of sea-level rise on seagrasses in Moreton Bay (S-E Queensland), and show that two factors (benthic irradiance, which is closely correlated with depth, and wave height) are sufficient to map seagrass extent with 83% accuracy. Similarly, Beaman et al. (2016) demonstrated that water depth is a key determinant of benthic habitat in the Coral Sea. Similar work has demonstrated the relationship in marine habitats around the world.</td>
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<tr>
<td>Corals</td>
<td>Changes in ocean circulation and upwelling</td>
<td>Changes in nutrient delivery to marine ecosystems, changes in temperature and changes in transport of reproductive spores and juveniles.</td>
<td>Increased pressure from invasive species; change in habitat suitability for existing primary producers and habitat-forming species. Medium-term reduction in ecosystem productivity.</td>
<td>Hoegh-Guldberg and Bruno (2010) present evidence that climate change has altered marine ecosystem function and structure in various ways, with a net result of reduced marine ecosystem productivity.</td>
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<td>Phytoplankton and benthic algae</td>
<td>Ocean acidification due to increased atmospheric CO₂ concentrations</td>
<td>Calcification is inhibited (marine animals are unable to form hard shells)</td>
<td>Loss of rocky corals and some shellfish. Loss of ecosystem production hotspots.</td>
<td>A review of experimental evidence shows that ocean acidification can shift coral reefs from a state of net accretion (growth) to net erosion (Andersson and Gledhill, 2013). Field experiments and observations provide strong evidence that this is already happening in many reefs around the world (Albright et al., 2016). A meta-analysis of literature shows that responses of shellfish to ocean acidification are variable. Some populations are more sensitive than others, particularly when subject to other stressors such as elevated temperatures (Kroeker et al., 2013).</td>
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<td>Tidal estuary channels</td>
<td>Sea-level rise</td>
<td>Estuarine turbidity maximum and salinity gradients move upstream</td>
<td>Shift in balance of estuarine and freshwater productivity</td>
<td>Many modelling studies (e.g. Chen et al., 2015) predict upstream propagation of salt and (in macrotidal estuaries) turbidity maxima with sea-level rise. The extent of upstream propagation shows a nonlinear response to sea-level rise and depends on local topography (e.g. Aubrey and Speer, 1985).</td>
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<tr>
<td>GDEs</td>
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<td>Salt intrusion into groundwater</td>
<td>Decline in health of groundwater-dependent vegetation</td>
<td>Modelling studies (e.g. Carretero et al., 2013) and some experimental work (e.g. Morgan et al., 2013) show increased saline groundwater intrusion should be expected with sea-level rise, though the effect may be smaller than the effect of reduced freshwater recharge. Evidence from around the world has been comprehensively reviewed by Ketabchi et al. (2016). Observational evidence has demonstrated increased stressing of forest vegetation associated with saline groundwater intrusion (Barbarella et al., 2015).</td>
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<td>Estuarine habitats</td>
<td>Altered estuarine morphology</td>
<td>Shifts in coastal habitat</td>
<td>Unpredictable change in extent and distribution of mangrove, salt marsh and estuarine habitats.</td>
<td>Effects of sea-level rise on groundwater tables may cause more inundation than direct effects of sea-level rise (Rotzoll and Fletcher, 2013). Observed shifts in mangrove distribution in the Gulf of Carpentaria have been attributed to sea-level rise and climate change (Asbridge et al., 2016).</td>
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<tr>
<td>Coral reefs, soft corals, seagrasses and macroalgal beds</td>
<td>Increased severity of tropical storms</td>
<td>Loss of coral habitat. Uprooting of seagrass and macroalgae. Increased turbidity of marine and estuarine waters.</td>
<td>Reduced marine ecosystem productivity. Long-term change in marine food-web structure.</td>
<td>Statistical analysis of observational data suggests that there has been a major decline in coral cover in the Great Barrier Reef over the past 30 years, and that tropical storms are one of the key contributors to coral loss (De'ath et al., 2012). Recent global climate change modelling predicts a reduction in frequency of tropical cyclones (Tory et al., 2013) and observational evidence supports this (Haig et al., 2014), however some modelling and data analysis suggests that storms may become more severe (Lin and Emanuel, 2016; Wasko and Sharma, 2015).</td>
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<tr>
<td>All</td>
<td>Altered hydrology</td>
<td>Potential effects similar to effects of direct alteration of hydrology by humans (e.g. river regulation).</td>
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<td>All</td>
<td>Increased temperature</td>
<td>Increased metabolic rates with stress conditions for some species</td>
<td>Likely increased gross primary production in combination with increased bacterial and animal respiration.</td>
<td>Taniwaki et al. (2017) review evidence and knowledge gaps relating to the effects of climate change on tropical streams, including direct and indirect effects of temperature change on algal, macroinvertebrate and fish communities and cumulative impacts of temperature change with other stressors. There is evidence for increased ecosystem respiration along with changes in community structure.</td>
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<tr>
<td>Coastal habitats</td>
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<td>Likely increase in disease and metabolic stress affecting both primary producers and animals. Unpredictable net ecosystem productivity effects.</td>
<td>and food-web structure, but the authors conclude that more study is needed to fully understand the effects of multiple stressors including temperature. Burge et al. (2014) review evidence for effects of climate change on marine infectious diseases, showing several lines of evidence for increased risk and concluding that disease spread that is likely to be associated with climate change is difficult to predict and difficult to manage. Recent marine heatwaves have been associated with massive mangrove, seagrass and coral die-off across northern Australia (ABC News, 2016) (Ainsworth et al., 2016; Thomson et al., 2015). Paerl and Paul (2012) review the broad range of evidence suggesting that climate change, including temperature responses, are likely to lead to increased incidence of harmful algal blooms in fresh and coastal waters. Much of this work, however, has been conducted in subtropical and temperate areas, and these conclusions may not be valid for tropical systems, where temperatures are already close to or above those optimal for cyanobacterial bloom formation.</td>
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<tr>
<td>Lacustrine, estuarine and coastal waters</td>
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<td></td>
<td>Death of mangroves, corals and seagrasses</td>
<td>Loss of productive habitat; loss of overall ecosystem productivity.</td>
<td>Recent marine heatwaves have been associated with massive mangrove, seagrass and coral die-off across northern Australia (ABC News, 2016) (Ainsworth et al., 2016; Thomson et al., 2015). In the case of corals, there is a long record of increasing frequency and severity of bleaching events associated with exposure to high water temperatures. In the case of mangroves, the causal connection has not yet been definitively proven but is the subject of active research.</td>
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<td>Change in dominant primary producer species (some species thrive at higher temperatures, others cannot survive with elevated temperatures)</td>
<td>Increased potential for harmful algal blooms</td>
<td>Current evidence is relatively speculative. What is known and not known is discussed by Wells et al. (2015).</td>
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## HABITAT

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<th>1. HABITAT</th>
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| All habitats   | Introduced species| Displacement of native species by weeds and feral animals | Altered food-web structures  
Possibility of shortened food chains (reduced number of trophic levels)  
Changes in water clarity and water quality | Changes in food-web structure may result in increased or unchanged ecosystem production but reduced biodiversity and reduced production of desirable species such as commercial fishery species  
Some invasive species adversely impact water clarity and water quality, increasing algal production but reducing higher-level ecosystem production. | Several examples in the literature demonstrate changes in food-web structure and ecosystem productivity associated with aquatic invasive animals, though the effects are complex and vary depending on local specifics. For example, Kreps et al. (2016) presents evidence that invasive crayfish reduced biodiversity and higher trophic level productivity in north American lake ecosystems. Walsh et al. (2016) present evidence that an invasive water flea disrupted food webs in a north American Lake and resulted in a substantial reduction in water clarity and ecosystem services associated with reducing grazing and increased algal biomass. Baxter et al. (2004) showed that invasive fish had a dramatic effect on food-web structures in a stream ecosystem, increasing algal production and reducing the biomass of aquatic insects and spiders. Invasive cane toads in northern Australia have resulted in reduced predator populations (e.g. Letnic et al., 2008) and the spread of cane toads is expected to accelerate with increased development and road traffic. |
<p>| River channel and coastal waters | Clearing and farming | Increased erosion and channel disturbance by domesticated animals | Increased sediments enter river channels, increasing turbidity and reducing water clarity | Reduced productivity. Loss of benthic plants. | Increased erosion and resulting increased sediment loads due to clearing have been extensively documented in Great Barrier Reef catchments. Sediment loads from Great Barrier Reef rivers has increased by a factor of four or more over pre-development loads, which has also increased the turbidity of near-shore coastal waters (Neil et al., 2002). This is primarily due to intensive cropping and grazing (Haynes and Michalek-Wagner, 2000), so water resource development in northern Australia is likely to increase impacts on sediment and nutrient loads above existing impacts from grazing. Increased turbidity associated with elevated sediment loads has been associated by some with declining seagrass coverage in Great Barrier Reef waters (Brodie et al., 2012). Others have concluded that there is no clear evidence for an overall decline in seagrass coverage (Schaffelke et al., 2005). Interestingly, there is overlapping authorship between these literature sources. There is extensive evidence of negative impacts of high sedimentation rates on corals (reviewed by Bartley et al., 2014). |</p>
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<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
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<th>6. EVIDENCE AND PATHWAYS</th>
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<tr>
<td>Waterholes and other lacustrine habitats</td>
<td>Increased sediment load or trapping of sediments by weirs leads to siltation of waterholes and smothering of aquatic angiosperms (benthic plants).</td>
<td>Switch from benthic to pelagic (phytoplankton) production. Reduced retention of production within freshwater food webs.</td>
<td>The degree to which sediments from rivers directly smother corals in Great Barrier Reef coastal waters is contested (Orpin and Ridd, 2012). Aquatic macrophytes can adjust to settling sediments by growing new leaf surfaces, but production is reduced and slower-growing species are smothered (Jones et al., 2012).</td>
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<td>Freshwater and marine habitats</td>
<td>Increase in nutrient loads associated with sediments</td>
<td>Increased primary production in freshwater and marine habitats but potentially simplified food webs and reduced biodiversity</td>
<td>Release of nutrients from fine sediments has been documented and shown to have increase primary production and drive loss of macrophytes in both freshwater and marine systems (Devlin and Brodie, 2005; Jones et al., 2012). Nutrients released from sediments disperse further into marine waters than the sediments themselves, and the effect of this on phytoplankton production (increased production) and crown-of-thorns reproductive success is hypothesised to have contributed greatly to the decline of coral cover in the Great Barrier Reef (Brodie et al., 2008).</td>
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<tr>
<td>Freshwater and marine habitats</td>
<td>Increased nutrient loads associated with use of fertilisers and production of animal manure</td>
<td>Increased aquatic primary production; shift in production from benthic plant-dominated to phytoplankton-dominated; reduced water quality.</td>
<td>Initial increase in productivity of low-nutrient waters; ultimately reduced productivity of desirable fish and bird species if nutrient loads continue to increase. Possibility of harmful algal blooms and deoxygenation leading to fish kills and loss of biodiversity.</td>
<td>There is a very extensive literature on the effects of increasing nutrient loads (eutrophication) on fresh and coastal primary production and ecosystem productivity (Ryther and Dunstan, 1971; Smith and Schindler, 2009). In general, primary production increases with increasing nutrient loads, while effects on fishery production are variable (Micheli, 1999). In low nutrient fresh and marine waters, increased nutrient loads result in increased benthic and pelagic primary production (e.g. Townsend et al., 2012; Udy et al., 1999). Nutrient addition experiments in the low-nutrient Daly River, NT, showed that primary production (in the form of periphyton growth) increased when nutrients were added (Townsend et al., 2012). In the Flinders River, southern Gulf, nitrogen and phosphorus addition resulted in increased primary production (Faggetter et al. 2013). The early...</td>
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<tr>
<td>River channels and freshwater wetlands</td>
<td>Rising groundwater tables</td>
<td>Salinisation</td>
<td>Elevated salinity in freshwater systems can increase turbidity (through flocculation of fine particles), reducing primary production. High salinity can also kill riparian vegetation and reduce decomposition of terrestrial organic matter, also contributing to lower ecosystem productivity.</td>
<td>(Reice and Herbst, 1982 as cited by Bunn, 2006) found reduced leaf litter decomposition due to reduced fungal and bacterial activity in saline desert streams. Bunn et al. (2006) also cite evidence for increased turbidity and changes in riparian vegetation composition resulting from salinisation of desert streams.</td>
<td>stages of eutrophication can increase fishery productivity and fish populations (Willemsen, 1980) however a path to eutrophication is difficult to reverse. In higher nutrient conditions, phytoplankton and bacterial production increases while benthic production declines due to declining water clarity. Harmful algal blooms and hypoxia become a problem, fish catches are likely to decline and fish kills are likely to occur (e.g. Smith and Schindler, 2009; Tammi et al., 1999; Willemsen, 1980). Hyper-eutrophication of the Gulf of Mexico has created a “dead zone” of very low ecosystem productivity due to hypoxia (Rabalais et al., 2002). Nutrient concentrations in at least some rivers in the present study are already at moderate to high concentrations (Short, 2016), so the effects of increased nutrient loads on higher trophic level ecosystem productivity are likely to be negative. Brodie and Mitchell (2005) find that tropical Australian rivers in areas with developed catchments have consistently higher nitrogen concentrations than those with undeveloped catchments. Ganf and Rea (2007) found strong potential for algal blooms in Northern Territory rivers with land development.</td>
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<tr>
<td>Freshwater habitats</td>
<td>Increased spread of invasive species associated with roads, movement of people and domestic animals.</td>
<td>Effects as summarised in the “introduced species” section above.</td>
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<td>Floodplains</td>
<td>Works to reduce floodplain inundation</td>
<td>Reduced frequency and duration of floodplain inundation. Reduced habitat connectivity during flood events.</td>
<td>Greatly reduced river system and marine production</td>
<td>Junk et al. (1989) present a conceptual model and supporting evidence that loss of floodplain inundation greatly reduces river productivity. Davies et al. (2008) present evidence that floodplains are the most productive part of a tropical river system. Bayley (1991) shows that tropical river fishery production is higher than temperate river fishery production, and argues that this is due to the ‘flood pulse advantage’, which is lost if floodplains are disconnected. Winemiller and Jepsen (1998) show that fish migration to and from floodplains subsidises river and marine ecosystem productivity. Jardine et al. (2012b) showed that floodplains on the Mitchell River were a major contributor to fish productivity in the river/estuary system.</td>
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<td>Floodplains</td>
<td>Increased frequency of fires</td>
<td>Reduced accumulation of terrestrial organic matter. Carbon converted to less bioavailable forms. Reduced terrestrial subsidy to aquatic systems.</td>
<td>Reduced productivity boost associated with floodplain inundation.</td>
<td>Of immediate local relevance, Edwards et al. (2003) show that burning in Kakadu National Park substantially reduces terrestrial leaf litter and alters floodplain vegetation community structures. Pettit et al. (2011) show that leaf litter is an important source of carbon to a Darwin region tropical river–floodplain system. Mihuc and Minshall (2005) show that food-web production shifted from reliance on terrestrial subsidies to autochthonous (instream) primary production in the aftermath of a major fire in Yellowstone National Park.</td>
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<td>Floodplains</td>
<td>Loss of floodplain and riparian vegetation</td>
<td>Reduced accumulation of native terrestrial organic matter such as leaf litter. Terrestrial organic matter associated with crop production is harvested and removed.</td>
<td>Reduced productivity boost associated with floodplain inundation.</td>
<td>Bunn et al. (1997) present evidence that loss of riparian vegetation of a far north Queensland stream led to a reduction in supply of terrestrial organic matter usable by the aquatic food web, combined with excess primary production within the stream and choking of the stream with emergent macrophytes and aquatic plants. They suggest that this results in reduced production of high order consumers. Pettit et al. (2011) show that leaf litter is an</td>
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<td>River channels and offchannel wetlands</td>
<td>River regulation and weir construction</td>
<td>Increased trapping of sediments and nutrients in dams and weirs</td>
<td>Siltation of upstream river channel habitat; loss of riffle-pool sequences</td>
<td>Reduced ecosystem productivity</td>
<td>Bednarek (2001) review the effects of damming and dam removal on river system function, finding that dam removal resulted in restoration of riffle-pool habitat sequences as well as improvements in connectivity. Effects on river system productivity were not directly discussed.</td>
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<td>Reduced organic matter and nutrient loads to downstream waterholes and river channels</td>
<td>Reduced ecosystem productivity</td>
<td>Maeck et al. (2013) estimate that trapping of organic matter by dams globally has increased methane emissions from rivers by 7%, implying increased bacterial breakdown of these materials.</td>
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<td>Reduced export of nutrients to marine waters</td>
<td>Reduced estuarine and marine productivity</td>
<td>Ittekkot et al. (2000) notes that although dams trap sediments and nutrients, nitrogen and phosphorus loads from developed catchments are usually elevated overall. Trapping of silica, however, can affect marine productivity by altering phytoplankton community composition, reducing the production of diatoms which can be nutritionally important.</td>
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<tr>
<td>River channels</td>
<td>Reduced connectivity; fish no longer able to migrate upstream</td>
<td>Loss of mechanism for upstream and lateral transfer of carbon and nutrients. Changes in food-web structure; increased food-chain length.</td>
<td>Reduced river channel and wetland productivity; ultimately reduced marine productivity Reduced biomass of top-level predators</td>
<td>Fish migration can be an important component of river estuary carbon and nutrient budgets (Deegan, 1993). Friedl and Wuest (2002) briefly review evidence that damming can cause oligotrophication (reduction in nutrient loads) and declining fish populations. Ruhi et al. (2016) present evidence that, by reducing streamflow variability, river regulation increased intra-guild predation and thus increased food chain length in Mediterranean streams.</td>
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<td>Estuaries</td>
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<td>Estuaries</td>
<td>Reduction in peak flows</td>
<td>Reduced nutrient supply to tidal estuaries</td>
<td>Reduced estuarine primary production</td>
<td>Burford et al. (2011) found that regulation of the Ord River has probably reduced estuarine productivity. Flow regulation in Ord River appears to be linked with reduced densities of prawns in the estuary (Kenyon et al., 2004)</td>
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<td>Increased delivery of sediments due to catchment and bank erosion, and reduced removal of sediments with outflowing water</td>
<td>Siltation of estuaries</td>
<td>Reduced estuarine productivity</td>
<td>Wolanski et al. (2001) present evidence that this has occurred in the case of the Ord River estuary. Similar effects have been found in other dammed tropical rivers (Hoa et al., 2007)</td>
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<tr>
<td>Reduced delivery of nutrients due to trapping in dams</td>
<td>Reduced nutrient and organic matter loads to estuaries</td>
<td>Reduced ecosystem productivity</td>
<td>Burford et al. (2011) suggest that this has occurred in the Ord River estuary, however this effect may be overwhelmed by increased anthropogenic nutrient loads associated with agricultural intensification (Brodie and Mitchell, 2005; Mitchell et al., 2009).</td>
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<tr>
<td>Ephemeral river channels and wetlands</td>
<td>Loss of dry-season flow</td>
<td>Reduced extent and wetted duration of ephemeral river channels and ephemeral wetlands</td>
<td>Reduced primary production in these habitats.</td>
<td>Bunn et al. (2006) discusses this issue, in theoretical terms. Larned et al. (2010) presents conceptual models for ephemeral river system ecosystems. Regular drying and rewetting increase biogeochemical process rates and hence productivity. Gradual drying may allow algae to adapt and recover quickly upon rewetting, whereas rapid drying further reduces productivity, though the evidence is mixed (Stanley et al., 2004).</td>
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<td>River channels, floodplains and wetlands</td>
<td>Reduced flow variability</td>
<td>Reduced biogeochemical activity releasing nutrients from wetting and drying. Reduced removal of established biomass (aquatic and riparian plants) and hence reduced opportunity for new plants and algae to establish.</td>
<td>Reduce primary production in river channels and wetlands.</td>
<td>Baldwin et al. (2000) and Venterink et al. (2002) both found that wetting and drying leads to pulses of phosphorus and other nutrients into floodplain aquatic systems. Mitsch et al. (1991) found that the highest primary productivity in forested wetland communities in a North American river system was associated with variable flow conditions (vs constant flow or stagnant water).</td>
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<td>Estuaries and marine waters</td>
<td>Increased dry-season discharge due to controlled releases from dams to support irrigation and other downstream water uses.</td>
<td>Increased dry-season terrestrial subsidy to marine production</td>
<td>Increased near-shore marine production; altered marine community structure</td>
<td>Damming of the Ord River and associated hydro-electric production has been associated with a change from a seasonally-flowing river to continuous flow, with slightly elevated nutrient and chlorophyll concentrations in the lower river during the dry season associated with irrigation return flows (Robson et al., 2013).</td>
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<td>River channels and floodplain wetlands</td>
<td>Groundwater extraction</td>
<td>Reduced water levels in river channels and wetlands</td>
<td>Reduced freshwater area for production but potentially increased area of shallow water allowing increased benthic algal production.</td>
<td>Moderate reduction or increase in river channel and wetland productivity, depending on details of local conditions</td>
<td>Robson et al. (2008) observed (in-situ) an increase in primary production in the Ord River in response to a temporary reduction in flow due to dam operation matters. Vis et al. (2007) found reduced water levels in the St. Lawrence River (Canada) associated with increased algal production but reduced macrophyte production, and hence little change in overall primary production according to analysis of remote-sensing data.</td>
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<tr>
<td>Ephemeral freshwater habitats</td>
<td>Reduced duration of inundation of ephemeral river channel reaches, waterholes and wetlands</td>
<td>Reduced freshwater area for production. Reduced export of material from these environments to perennial river channels</td>
<td>Moderate reduction in river system productivity</td>
<td>NAWRA hydrological modelling shows that water resources development is likely to reduce the persistence of waterholes through the dry season. Balcombe and Arthington (2009) present evidence for the importance of such waterholes for fish productivity.</td>
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<tr>
<td>Coastal systems</td>
<td>Reduced dry-season river discharge to estuarine and marine waters; reduction in peak wet season flows.</td>
<td>Reduced terrestrial subsidy to marine production</td>
<td>Small to moderate reduction in marine productivity</td>
<td>Dry-season nutrient loads from Australia’s tropical rivers are typically a much smaller source of nutrients to marine waters that wet-season loads (Brodie and Mitchell, 2005; Devlin and Brodie, 2005; Robson et al., 2008; Robson et al., 2010), however could have a disproportionate effect due to the longer duration of the dry season.</td>
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<td>River channel benthic plants</td>
<td>Reduced dry-season water velocity in river channels</td>
<td>Reduced nutrient transfer to benthic plants in low-nutrient rivers</td>
<td>Reduced benthic primary production in river channels</td>
<td>Townsend et al. (2012) found that benthic algae in the Daly River during the dry season showed higher productivity at higher water velocities.</td>
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<td><strong>GDEs</strong></td>
<td>Vine forests unable to access groundwater.</td>
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<td>Loss of vine forests</td>
<td>Reduced overall system productivity and biodiversity</td>
<td>Speculative (but refer to vine forest asset description)</td>
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<td><strong>River channel pools</strong></td>
<td>Increased residence time</td>
<td>Increased phytoplankton accumulation, reduced water clarity in river channel pools and weir pools</td>
<td>Loss of benthic productivity. Potential for harmful algal blooms.</td>
<td>Residence time has been clearly related to the occurrence of harmful algal blooms in eutrophic (high nutrient) conditions (e.g. Paerl et al. (2001); and more directly, Romo et al. (2013)). The Adelaide River already has elevated nutrient concentrations (Short, 2016) and increased nutrient concentrations are a very common result of irrigated agricultural development (e.g. Matson et al., 1997; Monteagudo et al., 2012).</td>
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4 Asset descriptions: Marine

4.1 Category: Habitats

4.1.1 MANGROVE

Asset description

Mangroves are diverse group of trees and shrubs that have evolved to exploit the land–sea interface under conditions that would severely stress other plants. They exist in tropical and subtropical coastal ecosystems worldwide between latitudes 32 °N to 38 °S. Mangroves are phylogenetically disparate, coming from several families, but have converged on saline coastal habitats as their optimum location. The 33 species of mangrove in tropical Australia range from small shrubs to 14 m trees. Their diverse assemblages form communities that provide a profound array of ecosystem services to estuarine and coastal habitats. Some of the key services mangroves provide include:

- stabilisation of coastal foreshores and habitats; both from coastal erosion and by moderating discharge loads from the terrestrial environment
- moderation of saline intrusion and tidal inundation to supra-tidal and terrestrial habitats, respectively
- incorporation of carbon and primary biomass to the estuary
- provision of a substrate for epiphytic and periphytic flora and fauna
- provision of three-dimensional habitats and ecosystem services to a multitude of fauna, many of economic significance
- provision of roosting and nesting habitat for coastal and terrestrial birds.

Mangroves create a three-dimensional habitat both above and below the substrate in which they grow. Their prop roots, pneumatophores and buttress trunks create complex structures within the water column that form a refuge and food basket for a multitude of flora and fauna (Blaber, 2007; Nagelkerken et al., 2008). Below ground, their roots stabilise muddy soils and allow crabs, worms and burrowing crustaceans to construct a labyrinth of holes, tunnels and cavities that support part-subterranean fauna critical to the decomposition and nutrient release supporting ecosystem productivity (Andreetta et al., 2014; Nagelkerken et al., 2008; Stieglitz et al., 2000).

Mangroves create a land–ocean interface, fringing much of the coastline of mainland Australia, with their most extensive and diverse communities found along the northern coastline (e.g. Wightman et al., 2004). Mangroves act as a buffer between land and sea, filtering terrestrial discharge and decreasing sediment loading. They play a critical role in nutrient and carbon recycling and export of organic material to food webs in coastal waters (Jennerjahn and Ittekkot, 2002). Importantly, their physical structure provides habitat and
A latitudinal gradient of mangrove species is evident, with the tropics supporting the most diverse mangrove communities (Duke et al., 1998). The location of mangrove species on a coastline can be generally described by two factors: freshwater input and intertidal position (Duke, 2013). However, the diversity of geomorphology and depositional environments along the Australian coast create a heterogeneous geo-environment providing the opportunity for globally significant diversity in mangrove community types (Brocx and Semeniuk, 2011; Semeniuk, 1985a; Semeniuk, 1985b; Semeniuk, 2008).

Within an estuary, a sequence in mangrove species occurs both within the tidal profile (temporal inundation) and along the longstream extent of the estuary (salinity gradient) (Duke et al., 1998; Vance et al., 2002). Mangrove species respond to ecophysiological cues along salinity, tidal and other environmental gradients, and disturbance regimes. In the Embley River estuary in the north-east Gulf of Carpentaria, the mangrove forests are dominated by three main species. The red mangrove (*Rhizophora stylosa*) occurs lowest in the tidal profile, *Ceriops tagal* in the upper intertidal and the grey mangrove (*Avicennia marina*) in the uppermost tidal profile, often only being inundated on spring tides (Conacher et al., 1996; Vance et al., 2002). In the southern Gulf of Carpentaria, *A. marina* dominates most coastal and estuarine mangrove communities (Danaher and Stevens, 2000).

**Distribution in Assessment study areas**

Figure 4-1, Figure 4-2 and Figure 4-3 show the distribution of mangroves in the Northern Australia Water Resource Assessment (the Assessment) catchments and marine study areas.
Figure 4-1 Distribution of mangroves in the Fitzroy catchment and marine study area, displayed by genera. In the background, the general distribution of mangroves from the National Intertidal/Subtidal Benthic habitat classification using dominant vegetation class 2.0.2.2 (mangroves) is shown. For better visibility, a 1-point outline has been set, which may over-represent the distribution.
Figure 4-2 Distribution of mangroves in the Darwin catchments and marine study area, displayed by genera. In the background, the general distribution of mangroves from the National Intertidal/Subtidal Benthic habitat classification using dominant vegetation class 2.0.2.2 (mangroves) is shown. For better visibility, a 1-point outline has been set, which may over-represent the distribution.
Habitat use

Mangroves grow in exposed and sheltered habitats, such as along sand-beach open coasts, and in estuarine creek complexes of mud sediments, meandering creeks and supra-littoral salt flats. They colonise deposition sediments and stabilise erosion zones, responding quickly to natural disturbance. Mangroves exploit heterogeneous habitats, from the fringe of extensive salt flats to expansive mud habitats in estuaries of the humid wet tropics and pockets of sediment on dry escarpment coastlines (Semeniuk, 1985a; Semeniuk, 2008). Mangroves grow in sediments that are generally low in oxygen and highly saline. Different taxa exhibit diverse morphological and physiological mechanisms to deal with such environmental stresses. Some species, such as *Rhizophora*, have aerial prop roots that branch down from the trunk and provide structural support in the mud soils. Others, such as *Avicennia*, have pneumatophores: aerial roots for gas exchange that grow vertically above.
the mud–soil surface from a shallow extensive network of anchoring roots. A higher diversity of mangroves is found in wetter regions than along drier coastlines, for example the eastern coast of Queensland compared with the drier Gulf of Carpentaria (Bruinsma and Duncan, 2000). The volume of runoff from catchments is a contributing factor to this observation (Duke et al., 1998). Considerable variability is also observed in the patterns of occupation and number of mangrove species among estuaries across northern Australia (Bunt, 1996).

Mangroves grow in habitats where the temperatures of the coldest months are higher than 20 °C, and where the seasonal range does not exceed 10 °C (Duke et al., 1998). Sedimentation and deposition in the coastal environment creates and maintains substrates for mangrove colonisation and habitation (Asbridge et al., 2016). Floodflows deliver sediments to the coastal zone, and modification of flows disrupts the stability of mangrove habitats (Asbridge et al., 2016; Duke et al., 1998).

The hydrology of mangroves is complex. Tidal inundation, rainfall, soil water, groundwater seepage and evaporation all influence soil salinity and have a profound effect on mangrove growth and distribution. High coastal rainfall and high riverine freshwater inputs support more diverse mangrove communities than areas of low coastal rainfall and limited runoff (Duke et al., 1998). A change in river flow regime within an estuary will also affect mangrove communities. Mangrove forests require freshwater input; many mangroves live close to their salinity tolerance levels. The use of fresh water by mangroves increases as their access to it increases, and their growth rate increases as their use of fresh water rises (Siantini et al., 2014; Wei et al., 2013). The water sourced by mangroves varies between the dry and wet seasons, as rainwater is not available over the dry season, while groundwater remains accessible (Ewe et al., 2007; Wei et al., 2013). In the Adelaide River estuary and adjacent floodplain, as measured at the end of the approximately 8-month dry season, soil water salinity and soil water content were major determinants of mangrove distribution (Ball, 1998). The highest species richness was encountered in soils with high soil water percentage (>40%) and medium salinity (20 to 80 ppt). At the extreme of tolerance, soil salinity was very high (~180 ppt) in some hypersaline soils that supported one or two mangrove species.

Mangrove productivity and ecosystem function

Mangroves are inextricably linked to the productivity of estuaries and coastal ecosystems, though their exact contribution has been contentious (Blaber, 2007). Mangrove forests shed leaves, branches and roots daily, contributing from approximately 44 to 1022 g carbon/m²/year from leaves and 912.5 to 6862 g carbon/m²/year from roots. The path of incorporating this carbon into the coastal ecosystem continues to be explored (Robertson and Alongi, 2016; Robertson and Daniel, 1989a; 1989b).

Tidal export of leaves from the forest occurs (~107 g carbon/m²/year); while various microbes and fauna consume the leaf matter in-situ (litter floor ~6 g carbon/m²). Burrowing crabs have been shown to bury leaves and enhance the decomposition and consumption cycling of nutrients from the leaf matter within the forests (>70% of the litter, from 62 to 803 g carbon/m²/year) (Robertson, 1986; Robertson and Daniel, 1989b). The nutrients
released or consumed support a food web and carbon sink within the mangrove community and soils (Gleeson et al., 2013; Robertson and Alongi, 2016). They are consumed by both epibenthic and demersal species, as well as crustaceans, annelids and fungi that occupy a myriad of well-flushed subterranean burrows and chambers within the mangrove forest substrates (Andreetta et al., 2014; Stieglitz et al., 2000).

Top predators, such as large fish and crabs, eventually consume components of this food chain, yet even at this level, the direct contribution of mangrove productivity to the estuarine fish community is contentious. Despite being a major component of estuarine ecosystems, mangroves themselves do not necessarily contribute primary production directly to higher-level consumers. Resident crustaceans and epifauna do not always derive nutrients directly from mangroves; they may derive nutrients from epiphytes and microphytobenthos within the mangrove community (Bouillon et al., 2008; Loneragan et al., 1997). Jardine et al. (2012b) showed that despite estuarine fish residing in large, mangrove-lined estuaries, seasonally they derive much of their nutrients from inundated floodplain adjacent to the estuary. Thus, the perennial discussion of the exact nature of the ecological services provided by mangroves continues (see Blaber, 2007; Nagelkerken et al., 2008).

In an estuary or embayment, mangroves grow in close proximity to other communities, such as seagrasses, and they grow in forests dissected by creeks and gutters with a myriad of mudbanks and shallow waters. The complex matrix of community types enhances the faunal community structure living within the ecosystem, including enhancing the species diversity and abundance of epibenthic fauna within the both the mangrove forest and, for example, nearby seagrasses and coral reefs (Nagelkerken et al., 2008; Serafy et al., 2015; Skilleter et al., 2005).

Estimates of 412,000 ha and 460,000 ha of mangroves occur along the Northern Territory and Queensland coastlines, covering 42% and 18% of the coasts, respectively (http://www.mangrovewatch.org.au; accessed 23 June 2017). Some 77,700 ha of mangroves (dominated by *Avicennia* and *Rhizophora*) and saltpan (242,860 ha) occur along the west Cape York coastline from Torres Strait to the Flinders River – the most extensive vegetated habitat type in the coastal zone (Danaher and Stevens, 2000). Mangroves support numerous crustaceans, fish and other species, including commercially important species, such as barramundi and penaeid prawns (Danaher and Stevens, 2000; Laegdsgaard and Johnson, 2001; Manson et al., 2005; Vance et al., 2002). Mangroves in Queensland waters are protected under the *Fisheries Act 1994*, with additional protection for some mangrove communities as Fish Habitat Areas (reserves) with regard to the sustainability of dependent fisheries (Bruinsma and Duncan, 2000; Danaher and Stevens, 2000).

In Australian estuaries, commercially important fish and crustacean species are strongly linked to the area of mangroves and salt marsh (Lee, 2004; Manson et al., 2005) and habitat connectivity within coastal ecosystems (e.g. spatial pattern of mangrove patches and length of creek systems) (Loneragan et al., 2005; Meynecke et al., 2008; Pittman et al., 2004; Skilleter et al., 2005). Export of organic matter from mangroves is important in sustaining
the coastal food webs that support commercial fisheries (Jennerjahn and Ittekkot, 2002; Melville and Connolly, 2005). Thus, modification of mangrove extent and connectivity of patches, alteration of mangrove productivity, or mangrove loss will impact on the economic value of dependent fisheries and associated biodiversity (Lee, 2004; Loneragan et al., 2005; Manson et al., 2005; McMahon et al., 2012; Serafy et al., 2015). Water extraction or modification of flow characteristics from the upland sections of rivers will affect the ecology of mangrove assemblages within their estuaries.

Conceptual model

Figure 4-4 shows the conceptual model for mangroves, with the accompanying narrative given in Table 4-1.
### Table 4-1 Knowledge underpinning the conceptual model for tropical mangroves in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intertidal substrates at the land–ocean interface</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow Timing and amplitude of monsoonal rainfall and flooding important in maintaining brackish ecotone in estuaries and shallow near-shore waters</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus, the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of freshwater and brackish waters decreases or is lost to the estuary. Optimal brackish salinity for some mangrove species is lost, thus critical habitat is lost. Possible loss of optimal salinity conditions for benthic fauna (including burrowers) that are critical to detritus cycling within the mangrove forest. Weak/ out-of-sync monsoon low rainfall, low runoff and reduced groundwater maintenance. Reduction in early-wet-season flows that sustain baseflow and freshwater inputs to estuary. The horizontal extent of brackish waters decreases or is lost to the estuary.</td>
<td>The reduction of brackish water habitat reduces the ability of some mangroves to survive in an estuary. A shift in mangrove community species richness and longstream distribution. Mangroves suited to marine-salinity waters (or higher) may colonise and dominate the mid- to upper estuary where previously freshwater habitats or brackish water species were dominant. Impeded breakdown of organic matter within the community and disrupted nutrient cycling.</td>
<td>(Duke et al., 1998; Gilman et al., 2008).</td>
</tr>
<tr>
<td>Water extraction/regulation and Climate change</td>
<td>Reduction in flood flow inundation/flooding regime</td>
<td>Maintenance of amplitude, volume and duration of overbank peak flows Timing of monsoonal rainfall and flooding important in maintaining estuarine ecotone Maintenance of sediment transfer from catchment to coast</td>
<td>Reduction in flood flow volume and duration reduces transport of sediment and nutrients to the estuary and near-shore deposition zones. Reduction in freshwater availability to mangroves during the wet-season; Reduction in groundwater and soil water recharge during the wet season; reducing the ‘store’ of soil water available to mangroves over the dry season.</td>
<td>Disruption to the accretion of intertidal sediments in estuarine and near-shore deposition zones. Disruption to colonisation of intertidal substrates by mangroves; possible erosion of mangrove community due to loss of depositional sediments. Water stress during the dry-season which may lead to mangrove death under a combination of waters stress, acid sulfate soils, abnormally high air/water Freshwater requirements (Siantini et al., 2014; Wei et al., 2013). Mangrove soil water stress (Lovelock et al., 2017). Sediment deposition in coastal zones (Asbridge et al., 2016)</td>
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<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Reduction in flood flow volume</td>
<td>Maintenance of volume and duration of peak flows</td>
<td>Reduction in flood flow volume and duration reduces the export of nutrients to inshore coastal waters within the floodplume.</td>
<td>Mangrove communities about the floodplume in the near-shore zone; they may benefit from floodplume nutrient transport and deposition.</td>
<td>Acid sulfate soils form when mangrove sediments dry out due to interrupted inundation (Luke et al., 2017).</td>
<td></td>
</tr>
<tr>
<td>Change in inter-annual sequencing</td>
<td>No impact</td>
<td>No impact.</td>
<td>No impact.</td>
<td>No impact.</td>
<td></td>
</tr>
<tr>
<td>Saltwater intrusion</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion reduces the extent of presence of a brackish ecotone within the estuary.</td>
<td>Expansion of mangrove habitats due to expansion of saline habitats at the land/seas interface. Loss of habitat to some mangrove species that thrive in brackish waters</td>
<td>(Duke et al., 1998; Gilman et al., 2008).</td>
<td></td>
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<tr>
<td>Estuarine/saltpan wetland and floodplain</td>
<td>Loss of seasonal inundation that recharges soil water and reduces the availability of freshwater to mangroves</td>
<td>Annual flooding of salt flats and wetlands mark an inundation of mangrove habitat, recharging soil water. Desiccated algal crust on salt pans invigorates and begin to photosynthesize Carbon and nitrogen production occur with input to the estuarine ecosystem,</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation saltpan and wetland adjacent to tropical estuaries. Reduction in freshwater availability to mangroves during the wet-season; Reduction in groundwater and soil water recharge during the wet season; reducing the ‘store’ of soil water available to mangroves over the dry season.</td>
<td>Reduction in mangrove extent and shift in community structure (Duke et al., 2017; Lovelock et al., 2017). Reduction in nutrient cycling in tropical estuaries with subsequent reduced coastal productivity for coastal species (Burford et al., 2012; Burford et al., 2016)</td>
<td>Death of species within the mangrove community due to water and heat stress (Duke et al., 2017; Lovelock et al., 2017). Reduction in nutrient production from flooded salt pans during monsoon season large flood flows (Burford et al., 2016).</td>
</tr>
<tr>
<td>Change in inundation frequency, depth and duration</td>
<td>Change in land use and management</td>
<td>Habitat degradation/</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/STRESSOR</td>
<td>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Homogenisation</td>
<td>Including mangroves (Burford et al., 2016)</td>
<td>Reduction in salt flat productivity as they remain in a desiccated state. Reduction in brackish ecotone and the persistence of the ecotone into the early dry season.</td>
<td></td>
<td>Reduction in nutrients exported from tropical estuaries with subsequent reduced productivity for species in near-shore habitats (Burford et al., 2010; Burford et al., 2012).</td>
<td></td>
</tr>
<tr>
<td>Longstream habitat connectivity</td>
<td>Truncation of estuarine conditions due to the placement of barriers across the estuary; freshwater conditions above the barrier; marine conditions below the barrier</td>
<td>Truncation of estuarine conditions</td>
<td>Physical barriers block upstream tidal flows of marine waters causing a truncation of available habitat. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary.</td>
<td>Truncation of estuary. Reduced linear habitat extent; and estuarine ecotone habitat. Loss of mangrove diversity along the extent of the estuary; upstream invasion of marine-adapted mangrove species.</td>
<td>Expert opinion.</td>
</tr>
<tr>
<td>Loss of saline conditions suitable for mangrove habitat</td>
<td>Truncation of estuarine conditions</td>
<td>Truncation of estuarine conditions</td>
<td>Physical barriers block upstream tidal flows of marine waters causing a truncation of available habitat. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary.</td>
<td>Truncation of estuary. Reduced linear habitat extent; and estuarine ecotone habitat. Loss of mangrove diversity along the extent of the estuary; upstream invasion of marine-adapted mangrove species.</td>
<td>Expert opinion.</td>
</tr>
<tr>
<td>Water column habitats all waters</td>
<td>Loss of sediment deposition within the estuary and near-shore flood plume zone destabilise mangrove habitats Poor water clarity, light penetration which may impact epiphytes on mangrove trunks, branches and leaves Loss of water column productivity/epiphytic algal productivity</td>
<td>Loss of flood-driven sedimentation processes within the estuary; loss of deposition environment. Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on the substrate, the large-form flora, and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of primary production; disruption of nutrient cycling (Duke et al., 1998).</td>
<td>Erosion of coastal landforms; loss of mangrove habitat (Duke et al., 1998).</td>
<td>Synthesis of literature.</td>
</tr>
<tr>
<td>Sediment loads</td>
<td>Loss of sediment deposition within the estuary and near-shore flood plume zone destabilise mangrove habitats Poor water clarity, light penetration which may impact epiphytes on mangrove trunks, branches and leaves Loss of water column productivity/epiphytic algal productivity</td>
<td>Loss of flood-driven sedimentation processes within the estuary; loss of deposition environment. Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on the substrate, the large-form flora, and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of primary production; disruption of nutrient cycling (Duke et al., 1998).</td>
<td>Erosion of coastal landforms; loss of mangrove habitat (Duke et al., 1998).</td>
<td>Synthesis of literature.</td>
</tr>
</tbody>
</table>
4.1.2 SEAGRASS

**Asset description**

Seagrasses are marine angiosperms (flowering plants) that provide valuable habitat and food resources for dugongs, sea turtles and other marine animals, and support local and regional biodiversity. In addition to being eaten directly by dugongs (e.g. Roelofs et al., 2005) and turtles, seagrasses provide a substrate for epiphytic algae, which provides a basal resource for marine food webs (Moriarty et al., 1990). Detrital seagrass biomass is an important food source for the Northern Prawn Fishery; juvenile prawn production in the north-east Gulf seems to be mostly based on seagrass-derived organic matter (Loneragan et al., 1997b).

Seagrasses provide marine animals with shelter from currents and predation, and stabilise bed sediments. They are important in providing nursery habitat for juvenile fish and prawns, including commercially important species (Coles et al., 2004; Roelofs et al., 2005).

Seagrass distribution is limited by light (seagrasses only occur where sufficient light reaches the bottom of the water column, down to around 60 m depth in clear waters) and suitable substrate. Seagrasses in relatively deep water are especially vulnerable to changes in water quality that affect light penetration or epiphyte growth. Some seagrass species are entirely subtidal, while others thrive in intertidal areas and depend on tidal exposure. Some species are intolerant of exposure to freshwater influences, while others thrive in brackish water and rely upon freshwater input to marine environments.

In addition to roles in providing habitat and food, seagrasses are important as stores of ‘blue carbon’ (carbon stored in above and especially below-ground marine biomass). A substantial fraction of the carbon stored within seagrass biomass is often beneath the surface, in root and rhizome biomass. Seagrasses can also reduce water velocity near the bottom, and hence influence coastal morphology by increasing sediment accretion or reducing erosion.

**Distribution in Assessment study areas**

Table 4-2 describes the current status of seagrasses in the Assessment study areas.

**Table 4-2 Current status of seagrasses in northern Australia**

<table>
<thead>
<tr>
<th>RIVER</th>
<th>MARINE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fitzroy</td>
<td>Seagrasses in the marine area influenced by the Fitzroy River are illustrated in Figure 4-5. At least 14 species of seagrasses have been observed on the northwest Australian coast (Walker and Prince, 1987), including in the marine area (including King Sound) influenced by the Fitzroy River.</td>
</tr>
<tr>
<td>Adelaide</td>
<td>A survey conducted in 2004 found more than 2000 ha of seagrass beds in van Diemen’s Gulf (Roelofs et al., 2005). Seagrasses may be present in clear waters and island fringes further offshore but have not been comprehensively mapped.</td>
</tr>
<tr>
<td>Finniss</td>
<td></td>
</tr>
<tr>
<td>Mary</td>
<td></td>
</tr>
<tr>
<td>Wildman</td>
<td></td>
</tr>
</tbody>
</table>
Approximately 15 species of seagrasses from three families (Cymodoceae, Hydrocharitaceae and Zosertaceae) are present in the Gulf of Carpentaria and immediate surrounds (Coles et al., 2004). The coastal area of the Gulf of Carpentaria is dominated by vast, highly productive seagrass beds (Burford et al., 2009), amounting to 900 km² when surveyed during the 1980s (Coles et al., 2004), however seagrasses in deeper waters (below 15 to 20 m) have not been properly surveyed, despite anecdotal evidence for the presence of seagrasses in clearer deep water of the Gulf (Coles et al., 2004). Seagrasses in surveyed study areas mapped in Figure 4-8. Coles et al. (2004) present a more thorough description of seagrass distribution and speciation in the Gulf of Carpentaria, and the associated environmental values.

Figure 4-5, Figure 4-6, Figure 4-7 and Figure 4-8 illustrate the distribution of surveyed seagrasses in the study areas. Surveyed areas are primarily nearshore, but more extensive seagrass beds may exist further offshore. The datasets used for this mapping are:

- Atlas of Living Australia
- National Intertidal/Subtidal Benthic habitat classification (NISB)
- Seagrass meadows of Arnhem Land and Gulf of Carpentaria by Department of Environment and Natural Resources.
Figure 4-5 Distribution of surveyed seagrasses in the Fitzroy catchment and marine study area
Figure 4-6 Distribution of surveyed seagrasses in the Darwin catchments and marine study area
Figure 4-7 Zoomed-in view of surveyed seagrass distribution near the mouths of the Darwin catchments showing greater detail of speciation
Conceptual model

Figure 4-9 shows the conceptual model for seagrasses, with the accompanying narrative given in Table 4-3.
Figure 4-9 Conceptual model illustrating the relationship between threats, drivers, effects and outcomes for seagrasses in northern Australia
### Conceptual model narrative

#### Table 4-3 Knowledge underpinning the conceptual model for seagrasses in northern Australia

<table>
<thead>
<tr>
<th>1. HABITAT</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deep, clear water</td>
<td>Climate change</td>
<td>Increased marine water depth</td>
<td>Deeper areas of current benthic habitat for corals, seagrasses and some macroalgae receive too little light due to increased water depth. May be ameliorated by migration to newly inundated areas in the case of seagrasses and macroalgae. Corals likely to be replaced with algae or seagrasses as coral reef migration is too slow.</td>
<td>Possible loss of sea grass beds, depending on details of local morphometry.</td>
<td>Relationships between water depth and seagrass coverage are very well established in the literature. Short and Neckles (1999) and Duarte (2002) review impacts of climate change, including sea-level rise, on seagrasses. Saunders et al. (2013) implemented models to predict the impacts of sea-level rise on seagrasses in Moreton Bay (south-east Queensland), and show that two factors (benthic irradiance, which is closely correlated with depth, and wave height) are sufficient to map seagrass extent with 83% accuracy. Similarly, Beaman et al. (2016) demonstrated that water depth is a key determinant of benthic habitat in the Coral Sea. Similar work has demonstrated the relationship in marine habitats around the world.</td>
</tr>
<tr>
<td>Active coastal areas</td>
<td>Increased severity of tropical storms</td>
<td>Uprooting of seagrasses. Increased turbidity of marine and estuarine waters due to resuspension.</td>
<td>Loss of seagrass habitat.</td>
<td>Recent global climate change modelling predicts a reduction in the frequency of tropical cyclones (Tory et al., 2013) and observational evidence supports this (Haig et al., 2014), however some modelling and data analysis suggests that storms may become more severe (Lin and Emanuel, 2016; Wasko and Sharma, 2015). Severe storm events have often been observed to result in loss of seagrass beds (Carlson et al., 2010; Cote-Laurin et al., 2017; Preen et al., 1995), including in tropical northern Australia (Rasheed et al., 2014), though they may recover if conditions are otherwise suitable and the frequency of disturbance is not too high (Rasheed et al., 2014).</td>
<td></td>
</tr>
<tr>
<td>All</td>
<td>Altered hydrology</td>
<td>Potential effects similar to effects of direct alteration of hydrology by humans (e.g. river regulation).</td>
<td>Synthesis of the literature.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>All</td>
<td>Increased temperature</td>
<td>Increased metabolic rates</td>
<td>Likely increased gross primary production in</td>
<td>Effects of temperature on seagrass metabolism (photosynthesis and respiration) are well established (e.g. Lee et al., 2007; Masini et al., 1995).</td>
<td></td>
</tr>
<tr>
<td>1. HABITAT</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
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</tr>
<tr>
<td>Coastal habitats</td>
<td></td>
<td></td>
<td>Death of seagrasses during heatwaves.</td>
<td>Loss of seagrass beds, especially in shallow intertidal areas.</td>
<td>Recent marine heatwaves have been associated with a major seagrass loss event in north-western Australia (Thomson et al., 2015)</td>
</tr>
<tr>
<td>Coastal habitats, especially shallow intertidal zones</td>
<td></td>
<td></td>
<td>Change in dominant species (some species thrive at higher temperatures, others cannot survive with elevated temperatures).</td>
<td>Changes in seagrass community composition.</td>
<td>Differences among seagrass species in thermal tolerance have been documented repeatedly, both experimentally (Campbell et al., 2006; Evans et al., 1986) and in effects on seagrass distribution (Mcmilian, 1984).</td>
</tr>
<tr>
<td>Seagrass beds, especially near-shore</td>
<td>Land clearing</td>
<td>Increased erosion</td>
<td>Increases in suspended sediment loads.</td>
<td>Increased turbidity reduces the depth at which seagrasses are able to grow, resulting in the permanent loss of deeper seagrass beds.</td>
<td>Land use intensification in northern Australia has been associated with dramatic increases in erosion from landscapes and sediment delivery to marine waters (Bartley et al., 2014). Irrigated croplands produce higher sediment loads (per unit area) than grazing lands, which in turn produce higher sediment loads than uncleared land (Neil et al., 2002). River sediment loads have been shown to reduce photic depths (i.e. depth of light penetration) in the Great Barrier Reef many months after flood flows (Fabricius et al., 2016; Fabricius et al., 2014) and similar results can be expected in other regions of northern Australia. Seagrass distribution is clearly limited by photic depth (e.g. Brodie et al., 2015). Seagrasses of species common in the study areas have also been experimentally shown to be susceptible to both shading and smothering effects due to suspended sediments and sediment deposition (Benham et al., 2016).</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Increase in nutrient loads associated with sediments</td>
<td>Increased phytoplankton and epiphyte production, leading to reduced light and loss of seagrasses, particularly in deeper waters.</td>
<td>Eutrophication (i.e. increasing nutrient loads) is strongly associated with declines in seagrass growth, biomass and extent around the world. Ralph et al. (2007). Review some of this literature and describe the relevant mechanisms, which include shading by phytoplankton, shading by epiphytes, sediment anoxia and in extreme cases, nitrogen toxicity.</td>
</tr>
<tr>
<td>1. HABITAT</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
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<tr>
<td></td>
<td>Farming practices</td>
<td>Increased nutrient loads associated with use of fertilisers and production of animal manure.</td>
<td>Reduced nutrient limitation of seagrasses.</td>
<td>Increased biomass and coverage of seagrasses in low-nutrient marine waters (e.g. the Kimberley coast and offshore from the Mitchell River). Change in species composition to favour more nutrient-tolerant families such as Zostera.</td>
<td>Relationships between land use, farming practices and nutrient loads are well established, including in Great Barrier Reef catchments of the wet tropics (Brodie and Mitchell, 2005; Mitchell et al., 2009). Although increased nutrient loads are much more commonly associated with declining seagrass coverage and biomass and this is a more likely outcome (Ralph et al., 2007), increases are possible and have been observed in areas where marine nitrogen and phosphorus concentrations are very low and seagrass growth is nutrient limited. In parts of the Great Barrier Reef, for example, seagrass coverage and biomass increased with land use intensification (Udy et al., 1999).</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Increased phytoplankton production leading to increased light attenuation.</td>
<td>Shading leading to reduced biomass and eventual loss of seagrasses and seagrass</td>
<td>Seagrass decline and changing seagrass community structures (Cardoso et al., 2004) in response to increased nutrient loads to marine waters have been repeatedly established around the world. A large...</td>
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<td>1. HABITAT</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
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<td></td>
<td>Increased epiphyte production on seagrass leaf surfaces, leading to direct shading of seagrasses.</td>
<td>Habitat, especially in deeper waters.</td>
<td>body of evidence is reviewed by Burkholder et al. (2007) and (in more summary form) by Ralph et al. (2007).</td>
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<tr>
<td>Herbicides and pesticides reaching marine waters.</td>
<td>Seagrass toxicity.</td>
<td>Loss of seagrass biomass and coverage.</td>
<td>A wide range of herbicides and pesticides commonly used in northern Australia have been found to be toxic to seagrasses (e.g. Flores et al., 2013; Negri et al., 2015; Wilkinson et al., 2015). These herbicides are frequently detected in waters of the Great Barrier Reef (e.g. Flores et al., 2013; King et al., 2013b; Negri et al., 2015; Waterhouse et al., 2012). There is already some limited use of herbicides in the study catchments, particularly for weed control in the Darwin catchments (Finlayson et al., 1999) but there has been little study of herbicides in marine waters associated with these regions. Likely herbicide usage patterns depend on crop choice (Waterhouse et al., 2012), and delivery to marine waters depends on usage, management practices, soil types and location.</td>
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<td>Groundwater extraction or river regulation</td>
<td>Reduced baseflow</td>
<td>Reduced extent of dry-season freshwater incursion to marine waters.</td>
<td>Small increase in suitability of seagrass habitat near mouths of rivers.</td>
<td>Seagrasses are generally intolerant of freshwater for more than brief periods of exposure (though degree of tolerance varies by species) (Adams and Bate, 1994; Collier et al., 2014), and can be harmed by direct exposure to flood plumes (Collier et al., 2014). The area affected by base flows, however, is likely to be small.</td>
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<tr>
<td>Dry-season dam releases for non-consumptive purposes</td>
<td>Increased baseflow</td>
<td>Increased extent of dry-season freshwater incursion to marine waters.</td>
<td>Small reduction in suitability of seagrass habitat near mouths of rivers.</td>
<td>See above.</td>
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<tr>
<td>Large dams</td>
<td>Reduced wet-season flow; reduced flood plume size</td>
<td>Reduced extent of short-term freshwater incursion to marine waters. Reduced extent of flood plume shading and reduced area of sediment deposition.</td>
<td>Small increase in seagrass habitat towards mouths of rivers.</td>
<td>Evidence for negative effects of seagrass exposure to freshwater is described above. Seagrass beds, however, can and do survive in areas exposed to flood plumes annually (Devlin and Schaffelke, 2009), so this effect is likely to be small if present. Evidence for negative effects of sediment shading and smothering is also discussed above.</td>
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4.1.3  SALT FLATS

Asset description

Salt flats are shallow coastal basins that are beyond daily tidal reach and can extend up to 40 km inland. They are low-gradient, featureless and covered in a thick, salt crust for the majority of the year. Because of their high salinity, salt flats are mostly devoid of vegetation, except for vertically accreting algal mats and salt-tolerant grasses. The habitat of salt flats discussed within this Assessment are those areas defined as the coastal supra-tidal salt flats of Australia’s wet–dry tropics in the Fitzroy, Darwin and Mitchell catchments. In the literature they are also referred to as mud flats (and more specifically as saline supra-tidal mudflats). The location of salt flats within the estuarine concept model is shown in Figure 4-10.

The salt flats of tropical northern Australia are extensive and still remain in a mostly pristine state. In the Gulf of Carpentaria, these flats cover several thousand square kilometres (Bayliss et al., 2014). In some areas, salt flats are mined for salt, such as in the Fitzroy estuary in Queensland. Intertidal salt flats or mudbanks are wetted daily during every tide cycle and have a very different composition and flow regime. They are found adjacent to coastal mangrove forests and intertidal mudbanks, which are not discussed here.

![Figure 4-10 Location of salt flats within the estuarine concept model](image)

The salt flats and wetlands of Australia’s wet–dry tropics are strongly affected by the seasons. While river flow may be non-existent during the dry season, flows are often high in the 3 to 4-month-long wet season (Finlayson and Spiers, 1999). Salt flats are characterised as a harsh environment for the majority of the year, with their meagre, salt-tolerant vegetation and senescent algal crust offering little opportunity to form a diverse biota salt flat community. However, annual wet-season floods cause river banks to overflow and inundate the salt flats, or significant local rainfall may flood the salt flats for long periods (Bayliss et al., 2014). When inundated, the ecosystem services provided by salt flats change dramatically. Wetted algal crust, which covers such a large aerial extent, begins to photosynthesise and produce carbon within the system. Primary production can be...
consumed in-situ or exported to estuaries and creeks that channel the salt flats. The biogeochemistry that leads to algal growth is not well known, but nutrient production from inundated salt flats into neighbouring areas has been demonstrated (Ridd et al., 1988). The large area of salt flats can contribute 16% of the system’s primary production in years of high flood flow. Crustaceans and fish colonise the newly available habitat and a diverse aquatic community may exist on the inundated salt flats for 1 to 3 months in years of high flows. When inundated, the salt flat community provides a key food source for fishery species, such as barramundi, and other fish species. These salt flat communities also support these species’ expansive use of flooded areas as nursery and adult habitats in years of high flow. For example, Metapenaeid prawns were found on salt flats in high numbers after flooding (Kenyon et al., 2012). In years of flooding, the known benefits of access to supralittoral, riparian and palustrine habitat for fishery species cannot be capitalised upon without abundant prey in these habitats, and the habitats that are transited to access them.

At a local scale, salt flats provide habitat for algae and for shorebirds. Metapenaeid shrimps were abundant in salt flats after inundation (Bayliss et al., 2014). Bayliss et al. (2014) provided a risk assessment of salt flats, which gave a moderate risk level due to localised impact.

Salt flats are affected by changes in the flow regime, such as those caused by water resource development, climate change and surface/groundwater extraction. These events may alter the timing, duration and size of flooding events. Salt flats contribute significantly to primary production in coastal areas (Burford et al., 2016). Consequently, a change in the amount and length of freshwater flows in the wet season will affect nutrient release, and further on, primary production in coastal areas (Burford et al., 2016; Burford et al., 2012). Inundation length was particularly found to promote algal production (Burford et al., 2016). While any level of inundation is sufficient for primary production to commence (since benthic algae grow directly on the soil surface), the duration of inundation is critical. Seven to ten days after algae become productive, food is available for higher trophic levels (Burford et al., 2010). In some instances of excessive inundation where the water is sufficiently deep and turbid, inundation-regime ecological functions may be disrupted, since light cannot penetrate to the soil surface.

At present, only minor pond-based aquaculture exists in the three Assessment study areas. Potential marine, pond-based operations would be sited near salt flats within 100 to 200 m of the coastline and, on average, occupy an area of 100 to 300 ha. Generally, the water required by such an operation is pumped from an estuary or creek, and after use is returned near to the source with little change in water quality. Hence, the current impact of aquaculture on salt flats appears to be negligible. The potential for future aquaculture facilities to cause a change in the flow regime that could affect the function of salt flats is likely to be low. This is due to the size of the operation, the water being sourced and discharged from a tidal water source downstream from salt flats, and the ponds being located above maximum possible flood extent.
Distribution in Assessment study areas

For the Fitzroy and Darwin catchments using the National Intertidal/Subtidal Benthic habitat classification scheme (NISB) dataset and for the Mitchell catchment, a combination of both the NISB and the Queensland Regional Ecosystem Mapping and Vegetation Communities database datasets provides the maximum possible extent for consideration. The resulting areas contain salt flats as well as salt marshes. Hence, salt flats are a sub-set of these areas. Burford et al. (2012) determined the salt flats of the Norman River, Queensland basin to be 356 km². Based on the same proportion, the salt flats of the Mitchell River could cover an area of approximately 290 km². For the Mitchell catchment, the saltmarsh features of the NISB dataset (which includes saltmarshes and salt flats) amount to 414 m²; hence, the estimate of 290 km² for the salt flats appears to be realistic. The topology of the Darwin catchments and especially the Fitzroy catchment are not similar, and therefore an estimate was not undertaken for these.

Figure 4-11, Figure 4-12 and Figure 4-13 show the distribution of salt flats in the Assessment catchments and marine study areas.

Figure 4-11 Distribution of salt flats in the Fitzroy catchment and marine study area
Figure 4-12 Distribution of salt flats in the Darwin catchments and marine study area
Figure 4-13 Distribution of salt flats in the Mitchell catchment and marine study area
QREM = Queensland Regional Ecosystem Mapping; NISB = National Intertidal/Subtidal Benthic (NISB) habitat classification scheme.

Conceptual model

Figure 4-14 shows the conceptual model for salt flats, with the accompanying narrative given in Table 4-4.
Figure 4-14 Conceptual model showing the relationship between threats, drivers, effects and outcomes for salt flats biodiversity and ecosystem function in northern Australia.
### Conceptual model narrative

**Table 4-4 Knowledge underpinning the conceptual model for salt flats in northern Australia**

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<td>River regulation</td>
<td>Changes to timing/magnitude of high flows</td>
<td>Continued flooding increases the net flow of water from the salt flats to estuarine areas and to marine. Soil-bound salts are transported to lower-estuarine areas reducing salinity in the salt flat.</td>
<td>Causes a change in availability of prawns in the trophic food chain in near-shore and deeper areas. Change in availability of nutrients in estuarine and near-shore areas.</td>
<td>Magnitude and duration of flooding, specifically reduction in wet-season flows or flow regulation affects the productivity of the system (Burford et al., 2010)</td>
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<td>Climate change/change in flow (wetter scenario)</td>
<td>Increased flooding duration during wet season</td>
<td>Wet-season flows with duration of at least several days.</td>
<td>Initially, increased nutrient and carbon export. Salinity decreases in salt flats due to flushing effect from flood waters.</td>
<td>Banana prawns and juvenile prawns which were abundant in salt flats pre-flood, do not tolerate low salinity and loss of food (meiofauna) and emigrate in large numbers from the estuary. Increase in juvenile prawns in near-shore areas and reduced mortality. Subsequent prawn migration to deeper waters.</td>
<td>Banana prawns move out of salt flats and into deeper marine areas as salinity in salt flats drop due to flooding (Kenyon et al., 2012). Low salinity and estuarine inundation during flood events stimulate juvenile prawns to emigrate from the estuary. A drop in salinity to zero for periods greater than a few days resulted in significant loss of meiofaunal abundance from the intertidal mudflats. Meiofauna are a key food source for the prawns and they did not recover until higher salinity was re-established (Burford et al., 2010).</td>
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<tr>
<td>Climate change/water development change in flow (drier scenario)</td>
<td>Reduction in size of flooded area</td>
<td>In wet season, flooding events cause inundation of salt flats which transport nutrients, carbon and juvenile prawns to estuarine and near-shore areas</td>
<td>Reduction in nutrient and carbon export.</td>
<td>Reduced estuarine and near-shore coastal productivity</td>
<td>Since the coastal zone is likely to be an area of high productivity, reduction in flow will reduce nutrient loads and hence coastal productivity. Primary production rates depended on areal extent of inundation rather than duration of inundation if it was longer than minimum period for PP to occur. Salt flats are, therefore, an important source of carbon and nutrients for the estuary and near-shore areas. Flow resulted in increased nutrient loads (but not higher nutrient concentrations) originating from the catchment and salt flats. This is</td>
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<tr>
<td>Fewer inundation events/shorter duration</td>
<td>In wet season, flooding events cause inundation of salt flats which transport nutrients and juvenile prawns to estuarine and near-shore areas</td>
<td>Reduction in algal growth, and nutrient and carbon release events</td>
<td></td>
<td></td>
<td>Transported to the coastal zone and is likely to play a critical role in promoting productivity. Wet-season flow is critical to causing floodplain and salt flat inundation, resulting in quantified increased nutrient and carbon loads to the coastal zone (Burford et al., 2010). Wet-season freshwater flows result in PP decrease (Burford et al., 2012). Rainfall events of a duration shorter than 7-10 days do not result in sufficient nutrient release to support local prawn and fish populations (Burford et al., 2010).</td>
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</table>
4.1.4 CORAL

Asset description

Corals are an obligate cnidarian with an algal symbiont. Many coral species can build a significant exoskeleton that forms major coastal reef structures or dominates the sediments and substrates in coastal habitats (Rosen, 1982). The reef-building corals create structured habitats for algae, seagrasses, crustaceans and fish. This enhances the productivity of the littoral zone by orders of magnitude greater than that likely for sympatric bare muds, sands or rocky substrates. The algal symbiont zooxanthellae provide the products of photosynthesis to the benefit of the host corals (Noonan et al., 2013).

Corals flourish in an oligotrophic environment, and on a worldwide scale over the last 100 years have been impacted by deteriorating water quality due to catchment erosion and runoff (De'ath et al., 2012; Hoegh-Guldberg et al., 2007). Corals and coral reef ecosystems are in major decline due to overfishing, reef (habitat) destruction, eutrophication, sedimentation (local and regional scale), and ocean warming resulting in bleaching and decalcification due to the increased acidification of oceans (global scale) (Dove et al., 2013; Hughes et al., 2017a; Pollock et al., 2014). There is evidence that heating in the Gulf of Carpentaria was also an important source of heat contributing to the Great Barrier Reef mass bleaching event in 2016 (Wolanski et al., 2017). Corals are good indicator species of water quality and change in hydrodynamics, thus providing indication of change in coastal habitats due to changes in catchment-derived flow.

Corals reef ecosystems are usually associated with continental shelf or terrestrial landforms. Fringing reefs exist in close proximity to landforms and ‘fringe’ coastlines, and are usually less extensive than barrier reefs. Worldwide, fringing reefs largely are in decline or extinct due to anthropogenic-instigated, terrigenous-derived impacts (Fabricius, 2005). Barrier reefs or lagoon reefs are supported by shallow continental shelf waters and often are hundreds or thousands of kilometres in linear extent. They are often 5 to 200 km from land and the bathymetry surrounding these reefs may be 50 to 200 m deep (De'ath et al., 2012; Gischler et al., 2000).

Pinnacle reefs (atolls) may be found remote from continents as a geologic limestone cap on a terrigenous pinnacle or formation that cause the original habitat for the settlement of coral larvae to be within the photic zone (i.e. ~50 m depth). Over geologic time, the limestone cap may grow to be thicker than 50 m due to geologic subsidence or sea-level rise (Rosen, 1982). Likewise, remote limestone islands are formed from raised reefs that become terrestrial islands with a fringe of coral. Pinnacle reef ecosystems may be in the form of multiple lagoons extending for thousands of square kilometres of ocean. Often, abyssal depths surround these oceanic reef ecosystems (Gardner, 1970; Rosen, 1982).

The most elaborate and extensive coral ecosystems are found on remote barrier or pinnacle geologic formations that provide tropical, oligotrophic waters. Corals thrive in clear, oceanic waters with low nutrient and sediment loads. Corals derive nutrients and organic compounds from their symbiont zooxanthellae; they are not dependent on background
nutrient levels, nor are they enhanced by elevated nutrient supply. Thus, the obligate condition for background nutrient level as an environmental source has a narrow range of tolerance that represents oligotrophic conditions. Eutrophic water conditions usually affect corals negatively and promote the growth of competitors, such as foliose algal communities or predators such as crown-of-thorns starfish, whose larvae benefit from abundant phytoplankton stimulated by nutrient ‘fertilisation’ (Brodie et al., 2012; Fabricius, 2005).

**Distribution in Assessment study areas**

Figure 4-15 and Figure 4-16 show the distribution of coral in two of the Assessment catchment and marine study areas.

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**Figure 4-15** Distribution of corals (*Anthozoa*) in the Fitzroy catchment marine study area
Figure 4-16 Distribution of corals (Anthozoa) in the Darwin catchments and marine study area based on records

Habitat use

Coral growth in shallow littoral and sub-littoral coastal and oceanic habitats in tropical waters to a depth of about 70 m in latitudes from about 30 °N to 30 °S (http://coralreef.noaa.gov/education/coralfacts.html; accessed 28 June 2017). They require a mean water temperature in the range 23 to 29 °C and optimal salinities are high (32 to 42 ppt) http://oceanservice.noaa.gov/facts/coralwaters.html; accessed 28 June 2017). Corals require a stable substrate on which to settle and grow, such as a substrate supplied by continental shelf or volcanic pinnacle formations that reach shallow ocean depths (<100 m). Subsequent geologic subsidence may render an atoll as a coral cap on top of the volcanic formation. Corals thrive in clear, oligotrophic waters. They derive nutrients and organic compounds from their symbiont zooxanthellae, thus reducing their reliance on external nutrient supply. As noted above, sediment and nutrient loads actually cause coral decline (Brodie et al., 2012; Fabricius et al., 2016; Pollock et al., 2014).
Coral growth is prolific in the northern section of King Sound, the receiving waters of the Fitzroy River (Gilmour et al., 2017). Corals grow as fringing reefs on the islands in the region. They inhabit Darwin Harbour and the southern coast of van Diemen Gulf, receiving waters for the Darwin rivers (ALA 2017; Cardno, 2013).

In the Darwin catchments few coral surveys have been undertaken. From 2012 to 2014, corals were surveyed in Darwin Harbour and adjacent control sites as part of the Ichthys Gas Field Development Project pipeline and onshore processing plant baseline and dredging-impact studies. Transects were established at Channel Island and East Arm (upper Darwin Harbour) and at Weed Reef (two sites, mid-harbour) and Mandorah and Charles Point (outer Darwin Harbour). Hard corals cover ranged from 13 to 23% at most sites, apart from less than 2% at one inner harbour site. Corals of the families Dendrophylliidae, Faviidae, Pectiniidae and Poritidae form encrusting, foliose, submassive and massive coral formations on a consolidated base. From 25 to 60% bare substrate of sand or silt covered the substrate between the corals. During the impact studies, coral bleaching caused 6 to 7% mortality of coral at both the impact and control sites, and had a much greater impact than dredging and marine works in the vicinity of the processing facility or pipeline.

In the Darwin rivers, corals do not form substantive reef structures as found in tropical seas and oceans. However, they are extensive and common in the littoral zone as continuous patchy growth forms on a hard, terrigenous substrate. Corals are not common in the eastern estuaries, which are dominated by mangrove forest/mud substrates and turbid waters. They are common on Coburg Peninsula bounding van Diemen Gulf, the receiving waters of the Darwin rivers. Likewise, rivers to the west of Darwin Harbour have a sandy substrate without inshore coral habitat.

Waters adjacent to the Mitchell River are poorly surveyed for corals. If evident, sparse coral colonisation of these waters is expected.

**Anthropogenic impacts on corals**

Currently, anthropogenic impacts within the catchments of all Assessment study areas are relatively low. Consequently, catchment load impacts on estuarine and near-shore ecosystems and communities are low. However, irrigated agriculture may change this situation. Case studies from elsewhere in Australia provide illustrations of possible impacts from modified flows and loads from catchments and rivers.

**Case study: land use change in eastern Queensland and the effects on the Great Barrier Reef**

The deleterious impact of anthropogenic, land-based and coastal activity on coral reef ecosystems is evident worldwide. One of the best documented cases is the Great Barrier Reef, where land use change in east Queensland coast river catchments has increased sediment loads in river discharge, with flow-on effects for reef health in adjacent coastal waters. Of the 2000 km north–south extent of the reef:

- the northern region (11.9 to 15.4 °S) lies adjacent to relatively natural lands and is relatively pristine and little altered by human activity
• the central region (15.4 to 20.0 °S) is situated adjacent to intense agriculture and grazing, and increasing coastal development

• the southern region (20.0 to 23.9 °S) is adjacent to significant land use change to support agricultural, infrastructure and urban production.

The reefs of the central and southern regions suffered a 50% loss in coral cover in the 27 years to 2012, which has been attributed to the combined effects of agricultural development, crown-of-thorns starfish and storm damage (De'ath et al., 2012). In 2016 and 2017, mass coral bleaching events due to climate-change-related, marine heat waves bleached more than 50% of the corals of the Great Barrier Reef, leading to the death of 29% of remaining reefs in 2016 and further damage in 2017 (Great Barrier Reef Marine Park Authority, 2017; Hughes et al., 2017a). The capacity for corals to recover from such bleaching events is believed to be reduced by other stressors.

In the northern region adjacent to Cape York, land is mainly used for rangeland grazing or is in natural condition, and supports better water quality of exported flows with much less impact on near-shore reefs. In the central and southern regions, river discharge carries 5 to 9 times the pre-1800s nutrient and sediment loads, which increases coral mortality by reducing water quality and stimulating impacts from other sources, such as crown-of-thorns starfish predation and competition from macroalgae (De'ath et al., 2012). Great Barrier Reef-wide, there is an inshore-to-offshore improvement in reef health that corresponds with better offshore water quality. The ability of reefs to re-establish after regional impacts is also affected by a reduced capacity for coral calcification and growth due to thermal stress and ocean acidification (De'ath et al., 2012; Dove et al., 2013), which are worldwide phenomena.

Both nutrient and sediment loads have affected the central and southern Great Barrier Reef, while the northern reef has remained relatively pristine. This outcome has significant implications for what might occur within the coastal communities of rivers in the Assessment study areas. Agricultural development has the scope to increase both sediment transfer and nutrient loading for catchment flows. Modification of flows, including floodflows, have less impact on coral than changes in loads that flows transport seaward (see Brodie and Mitchell, 2005; Pollock et al., 2014).

Climate change threats to corals

Over the last two decades, corals worldwide have been subject to global climate stress and atmospheric warming impacts. These fall into two major categories: warming hydrosphere (and atmosphere), and increasing acidification of the marine hydrosphere. Warming coastal and oceanic waters cause thermal stress to corals (Heron et al., 2016; Hughes et al., 2017b). Temperatures above 30 °C typically cause coral bleaching, though corals living in the Kimberley are adapted to water temperatures that exceed 30 °C for months at a time (Schoepf et al., 2015). However, even these thermally resilient corals are subject to bleaching if temperatures increase further (Schoepf et al., 2015).
During bleaching, the algal symbiont is expelled from the coral tissue with subsequent loss of vitality by the coral. The corals can recover if the heat event is shorter than about 10 days, but extended heat above 32 °C causes death and eventual decay of the coral skeleton (Ainsworth et al., 2016).

A second aspect of increasing carbon dioxide levels in the atmosphere is that the ocean has become more acidic and continues to increase in acidity. Since the industrial revolution, the ocean has absorbed approximately 30% of all anthropogenic CO₂ released into the atmosphere. Acidity changes the ability of organisms with carbonate body parts, skeletons or shells to metabolise carbon from marine water (Dove et al., 2013). On the Great Barrier Reef, coral calcification rates have decreased over the last 25 years (De'ath et al., 2013). Global impacts due to ocean warming and ocean acidification will continue and steadily increase for the foreseeable future; they may have greater impacts on coastal corals than nutrient or sediment loads delivered on modified flows. Throughout the latitudinal spread of corals, the heat tolerances of corals are being exceeded. The oceans are warming and will continue to warm (Barnett et al., 2001).

**Conceptual model**

Figure 4-17 shows the conceptual model for corals, with the accompanying narrative given in Table 4-5.

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**Figure 4-17** Conceptual model showing the relationship between threats, drivers, effects and outcomes for coral in northern Australia
### Conceptual model narrative

#### Table 4-5 Knowledge underpinning the conceptual model for coral in northern Australia

<table>
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<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
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<tr>
<td>Climate change</td>
<td>Thermal stress</td>
<td>Coral bleaching</td>
<td>Death of corals</td>
<td>Loss of coral cover</td>
<td>The relationship between heat stress and coral bleaching is very well established (e.g. Brown, 1997; Eakin et al., 2009; Fitt et al., 2001).</td>
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<td></td>
<td></td>
<td>(expulsion of symbiotic algae)</td>
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<td>Loss of biodiversity</td>
<td>Mass global bleaching events in 2016, including on the Great Barrier Reef, are described by Hughes et al. (2017a). Effects of 2015-16 and 2016-17 mass bleaching events on the Great Barrier Reef are described by the Great Barrier Reef Marine Park Authority (Great Barrier Reef Marine Park Authority, 2017): bleaching of more than 50% of the Great Barrier Reef resulted in an estimated loss of 29% of shallow-water coral Reef-wide, with the worst effects in the northern wet tropics.</td>
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<td>Wolanski et al. (2017) describe heating of the Gulf of Carpentaria during 2016 and the role this played in heating of the Great Barrier Reef.</td>
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<td>Ocean acidification</td>
<td>Reduced calcification</td>
<td>Reduced capacity for coral to grow</td>
<td>Loss of coral cover</td>
<td>Reduced resilience to other stressors</td>
<td>Effects of ocean acidification on coral calcification has been demonstrated theoretically, experimentally, and in the field (Albright et al., 2016).</td>
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<td>Chan and Connolly (2013) present a meta-analysis of evidence for inhibition of coral calcification by ocean acidification, showing a consistent effect of varying magnitude, on average producing an expected 15 to 20% decline in coral calcification by 2100 under business-as-usual conditions.</td>
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<td>De’ath et al. (2013) show that coral calcification rates have in fact decreased in northern Australian waters in recent decades.</td>
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<td>Mongin et al. (2016) modelled ocean acidification in the Great Barrier Reef, showing that the magnitude of acidification may be higher than previous estimates have suggested due to local intensification around coral reefs.</td>
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<td>Increased storm intensity</td>
<td>High shear stress</td>
<td>Physical damage to corals</td>
<td>Loss of coral cover</td>
<td>Recent global climate change modelling predicts a reduction in the frequency of tropical cyclones (Tory et al., 2013) and observational evidence supports this (Haig et al., 2014), however some modelling and data analysis suggests that storms may become more severe (Lin and Emanuel, 2016; Wasko and Sharma, 2015).</td>
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<td>Damage to corals from intense storms has been frequently reported in the literature (Cheal et al., 2002; Dollar and Tribble, 1993; Emslie et al., 2008; Harmelin-Vivien, 1994; Kjerfve et al., 1986; Osborne et al., 2011; Rogers et al., 1983; Scoffin, 1993).</td>
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<td>1. HABITAT AND THREATS</td>
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<td>Recovery by recolonisation of damaged areas will occur if conditions are favourable, but other stressors such as ocean acidification can inhibit recolonisation and calcification (Madin et al., 2012). De’ath et al. (2012) estimated that storm damage due to tropical cyclones accounted for a loss of 24% of the total coral cover of the Great Barrier Reef in the 27 years to 2012. Storm damage to coral reefs in the focus areas for NAWRA has not been well studied.</td>
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<tr>
<td>Land use intensification</td>
<td>Increased erosion</td>
<td>Increased sediment loads and associated nutrient loads</td>
<td>Increased turbidity leading to reduced coral growth rates</td>
<td>Loss of coral cover</td>
<td>Land use intensification in northern Australia has been associated with dramatic increases in erosion from landscapes and sediment delivery to marine waters (Bartley et al., 2014). Irrigated croplands produce higher sediment loads (per unit area) than grazing lands, which in turn produce higher sediment loads than uncleared land (Neil et al., 2002). River sediment loads have been shown to reduce photic depths (i.e. depth of light penetration) in the Great Barrier Reef many months after flood flows (Fabricius et al., 2016; Fabricius et al., 2014) and similar results can be expected in other regions of northern Australia. Although some corals can obtain energy through heterotrophic as well as autotrophic activity and can thus make use of particulate organic material from river plumes (Anthony and Fabricius, 2000), coral species richness is lower in turbid areas (Fabricius and De’ath, 2001) and disease risk is higher (Pollock et al., 2014). Overall, reduced photic depth due to terrestrial influences has been associated with reduced calcification, coral tissue thickness, photosynthetic activity and adult colony survival (Fabricius, 2005). Near-shore corals can also be impacted by sedimentation (sediments settling on coral surfaces). A review by Fabricius (2005) shows that sedimentation has been associated with reduced colony size, reduced coral cover, and reduced coral biodiversity, among other negative effects. Fabricius (2005) also reviewed effects of terrestrial runoff on corals more generally.</td>
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<tr>
<td>Use of fertilisers</td>
<td>Increased nutrient loads and resulting increase in marine phytoplankton growth</td>
<td>Increased turbidity leading to reduced coral growth rates</td>
<td>Loss of coral cover</td>
<td>Relationships between land use, farming practices and nutrient loads are well established, including in Great Barrier Reef catchments of the wet tropics (Brodie and Mitchell, 2005; Mitchell et al., 2009). Although photosynthesis by coral symbionts increases with increasing dissolved nutrient concentrations, this is countered by reduced photic depth (Fabricius, 2005). Dissolved nutrients delivered with river plumes are taken up rapidly by phytoplankton, increasing turbidity and decreasing light penetration to coral reefs (Furnas et al., 2005; Mallela et al., 2013).</td>
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<tr>
<td>Use of herbicides and pesticides</td>
<td>Dispersion of herbicides to marine environments and Reduced growth and increased mortality of corals</td>
<td>Loss of coral cover</td>
<td>A wide range of herbicides and pesticides commonly used in agriculture in northern Australia. These herbicides are frequently detected in waters of the Great Barrier Reef (e.g. Flores et al., 2013; King et al., 2013b; Lewis et al., 2009; Negri et al., 2015; Shaw et al., 2010; Smith et al., 2012; Waterhouse et al., 2012). Several of these herbicides have been</td>
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<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>subsequent exposure of corals</td>
<td>Reduced resilience to thermal stress and other stressors</td>
<td>shown to be toxic to corals (Ings et al., 2009; Jones, 2005; Jones et al., 2003; Negri et al., 2005; Shaw et al., 2009). There is already some limited use of herbicides in the study catchments, particularly for weed control in the Darwin catchments (Finlayson et al., 1999) but there has been little study of herbicides in marine waters associated with these regions. Likely herbicide usage patterns depend on crop choice (Waterhouse et al., 2012), and delivery to marine waters depends on usage, management practices, soil types and location.</td>
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<tr>
<td>Water extraction and regulation</td>
<td>Reduced end-of-system flood flows</td>
<td>Reduced exposure of near-shore corals to freshwater and sediments</td>
<td>Small increase or decrease in coral extent near-shore, due to reduced heterotrophic food, sedimentation and reduced exposure to freshwater</td>
<td>Evidence for effects of sediment and nutrient loads on corals is discussed above. These issues are particularly acute in near-shore corals exposed to flood plumes. Corals also bleach if subjected to low salinities from flood plumes (Berkelmans et al., 2012; Kerswell and Jones, 2003).</td>
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<tr>
<td>Reduced flows</td>
<td>Reduced sediment and nutrient loads</td>
<td>Reduced turbidity</td>
<td>Small increase or reduction in coral growth near-shore due to higher light near-shore but reduced terrestrial subsidy to heterotrophic subsidy</td>
<td>Evidence for effects of sediment and nutrient loads on corals is discussed above.</td>
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</table>
4.2 Category: Species of significance

4.2.1 WHITE BANANA PRAWN

Asset description

The white banana prawn (*Fenneropenaeus merguiensis*) is a short-lived, fast-growing crustacean species that is an important major commercial fishery resource across tropical Australia. They are distributed from the Red Sea across the Indo–Pacific to south-east Asia, Australia and western Oceania (Dall et al., 1990). In Australia, they are found from Shark Bay, WA to Moreton Bay, Queensland, and exist at the southern extremity of their global range (Dall et al., 1990). White banana prawns complete their life cycle within a year and so can be wild harvested annually. Their stock is tied to key environmental drivers, particularly annual flood flow (Staples and Vance, 1986; Vance et al., 1985); rather than annual abundance being driven by a stock-recruitment relationship. Hence, the well-managed harvest of the previous year’s stock is not a major determinant of likely catch. Each year’s catch of white banana prawns is highly stochastic, dependent on temporal cycles of monsoonal rainfall and river flows. White banana prawns are well studied in Australia and elsewhere (Loneragan et al., 2005; Vance, 2003; Vance et al., 1998), though they have demonstrated regional specificity in their behavioural responses to environmental cues (Robins et al., 2005), potentially limiting the transferability of prior research to the target rivers within the Assessment study areas.

In addition to forming a major constituent of a high-value fishery, white banana prawns are an important ecological species and a key component of marine and estuarine food webs. They provide a significant food source for a myriad of commercially and recreationally valuable fish species in the coastal ecosystem, including high-level predators such as barramundi (*Lates calcarifer*), the Talang queenfish (*Scomberoides commersonnianus*) and king threadfin (*Polydactylus macrochir*) (Salini et al., 1990). In a tropical estuary near Townsville, sub-adult barramundi were found to be the most important predators of juvenile white banana prawns, which constituted 22% of their diets by volume (Robertson, 1988). In the Norman River estuary, around 70% of the most abundant predator species ate juvenile penaeids (Salini et al., 1998).

The annual variability in white banana prawn catch means they may contribute to a high proportion of the annual prawn harvest across tropical Australia, with up to 7000 to 8000 tonnes taken. This falls to approximately 2000 tonnes available in years of poor rainfall and low flood flows (Barwick, 2013).

Rationale for key species selection

A significant body of research has investigated the life history, growth, behaviour and habitat use of the white banana prawn across multiple life stages to help inform the management of the Northern Prawn Fishery (Crocos and Kerr, 1983; Haywood and Staples, 1993; Rothlisberg et al., 1985; Staples, 1980a; 1980b; Staples and Heales, 1991; Staples and
Vance, 1985; 1986; Vance et al., 1996; Vance et al., 1990; 1992; Vance et al., 1985; Wang and Haywood, 1999). The Northern Prawn Fishery represents a very well-managed fishery of high economic value. Within this fishery, white banana prawns make up the majority of catch in the Assessment study areas and have hence been the subject of a large body of literature.

A primary finding within this research is that there is a strong mechanistic relationship between white banana prawn catch, catchment rainfall and resultant flow. This has been evident since the 1970s, when strong offshore prawn emigration was evident during runoff events. Specifically, Vance et al. (1985) found a positive relationship with rainfall and subsequent adult recruitment in the Gulf of Carpentaria, while rainfall accounted for 70% of the observed variation in monthly emigration of juveniles in the Norman River (Staples and Vance, 1986). Larger flow events are believed to increase prawn catch through greater juvenile emigration from estuaries to offshore habitats, where growth is enhanced and mortality is lower for the sub-adult and adult phases (Robins et al., 2005). Importantly, recent studies suggest that nutrients exported during flood flows support enhanced growth and survival and enhance food availability through primary and secondary production in near-shore habitats (Burford et al., 2010). Assessing the potential impact of altered natural hydrological regimes through water resource development is a critical issue, especially for white banana prawns, whose life cycles are intrinsically linked to natural flow regimes.

**Distribution in Assessment study areas**

White banana prawns are present in all of the Assessment catchment areas (Figure 4-18 and Figure 4-19; There are no records for the Fitzroy catchment). They are the only prawn species caught in the Mitchell catchment, and are the primary prawn species caught (>80% of all catch) across the Darwin rivers catchment Northern Prawn Fishery regions, which include Fog Bay, Melville and Port Essington (Barwick, 2013).

Yearly catches of white banana prawns within Assessment study areas are highly variable. For example, in the Mitchell catchment between 1995 and 2012, catches have ranged from around 71 t in 2006, to 601 t in 2002 (Barwick, 2013). Within the Darwin catchments, the Northern Prawn Fishery fishing region of Fog Bay has seen catches vary almost four-fold, from 123 t in 2005 to 448 t in 1995 (Barwick, 2013).
Figure 4-18 Observations of white banana prawn (*Fenneropenaeus merguiensis*) in the Darwin catchments and marine study area
Observations based on commercial datasets are also not shown.
Figure 4-19 Observations of white banana prawn (*Fenneropenaeus merguiensis*) in the Mitchell catchment and marine study area
Observations based on commercial datasets are also not shown.

**Habitat use**

The white banana prawn is a short-lived species with a larval life-history strategy heavily reliant on rainfall and subsequent flows (Dall et al., 1990). Spawning patterns indicate that primary spawning events occur offshore in marine zones from August through to December between depths of around 15 to 20 m, and are associated with increasing water temperatures (Crocos and Kerr, 1983). Spawned eggs hatch as larvae and are advected inshore by prevailing currents and winds, where they settle as benthic post-larvae within estuarine mangrove habitats up to 85 km upstream from the river mouth (Rothlisberg et al., 1985; Staples, 1980b). The majority of movement into estuarine zones occurs on large incoming flood tides in the weeks preceding seasonal rainfall (Staples and Vance, 1985). Post-larval prawns spend several months in the estuary between October to February, where they feed and shelter from predators. The primary habitats used by juveniles during this period include low-flow velocity and soft, fine sediments associated with mangrove
forests. The highest densities of post-larval and juvenile prawns caught are within the upper reaches of small creeks within the estuary (Vance et al., 1998). Juveniles will often move into larger creeks and the main river channel as they grow (Burford et al., 2010; Kenyon et al., 2004; Vance et al., 1998). The brackish conditions of tropical estuarine habitats are ideal for prawn growth, as temperatures of around 28 °C and salinities of 25 ppt have shown to be optimal (Staples and Heales, 1991). Juvenile growth has been demonstrated to be impeded at temperatures below 20 °C, and remains relatively rapid between 25 and 35 °C. Similarly, extreme salinities (5 and 55 ppt) impeded growth, while ranges between 20 and 45 ppt were more optimal (Staples and Heales, 1991). In fact, in drier conditions and seasons with little rainfall, hypersaline conditions can inhibit juvenile growth until rain returns habitats to brackish conditions.

Following an estuarine growth phase, both juveniles and sub-adults migrate back to marine waters in response to seasonal flows, which reduce saline levels and induce a physiological response whereby the prawns are unable to osmoregulate in freshwater, moving instead to more saline conditions (Dall, 1981). In addition to this emigration response to flows, larger flow events can reduce estuarine habitat suitability via a reduction in primary production and food availability, potentially providing an extra stimuli for prawns to move offshore into habitats with greater resources (Burford et al., 2010). When in marine waters, prawns have been associated with sediments high in mud content (Somers, 1987). Behaviourally, white banana prawns are quite unique among penaeids, as they form dense aggregations. The aggregations can become so large that they stir up sediments, forming blooms that can be spotted from the air (Lucas et al., 1979). Hence, throughout the fishing season, spotter planes are often used to identify schools, creating a highly targeted fishery.

**Fishery**

The Northern Prawn Fishery is one of Australia’s largest and most profitable fisheries. It targets five commercial species of prawns, including the white banana (*F. merguiensis*), red-legged banana (*F. indicus*), brown tiger (*Penaeus esculentus*), grooved tiger (*P. semisulcatus*), blue endeavour (*Metapenaeus endeavouri*) and red endeavour (*M. ensis*) (AFMA, 2015). Extending from Cape Londonderry in WA to Cape York in Queensland (Figure 4-19), the gross value production of the Northern Prawn Fishery averages around $100 million per year. However, the fishery is prone to fluctuations in catch and subsequent value due to the short life cycle and high sensitivity of prawns to environmental conditions, particularly seasonal rainfall and subsequent flow throughout northern Australia. Using the knowledge that seasonal catches are strongly influenced by rainfall and flow events, the fishery usually operates from April to June (Barwick, 2013). Fishing ceases when the fishery hits a catch rate trigger (Zhou et al., 2015), allowing the fishery to remain highly controlled and sustainable.

The Northern Prawn Fishery can be broken into two major fisheries that differ both spatially and temporally. One fishery solely targets white banana prawns, and another multi-species fishery targets tiger, endeavour, and red-legged banana prawns. The most significant contributors to the fishery are the white banana and tiger prawns. In years of high catch,
white banana prawns are the largest contributors to the fishery’s catch in terms of both biomass and economic value, despite being less valuable than tiger prawns per unit (Bath and Green, 2016).

**Conceptual model**

Figure 4-20 shows the conceptual model for white banana prawns, with the accompanying narrative given in Table 4-6.

The conceptual model developed for white banana prawns focuses on water extractions and regulation, along with climate change as the primary threats that are likely to impact the ecological and economic success of the Northern Prawn Fishery.
### Conceptual model narrative

**Table 4-6 Knowledge underpinning the conceptual model of white banana prawn in northern Australia**

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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<tr>
<td>Estuarine habitats</td>
<td>Altered wet-season flood magnitude</td>
<td>Seasonal, high-magnitude flow events are critical in supporting the white banana prawn life cycle as they act as the primary cue for juveniles to move out of the estuary and into offshore zones.</td>
<td>Water extraction and regulation has the potential to reduce/dampen the magnitude of wet seasonal flood events and is likely to reduce strength of emigration cue for juveniles to access offshore habitats. A drying climate is also likely to have similar effects to water extraction, reducing the magnitude of wet-season flood events, however, a wetter climate could lead to increased magnitude flows (though with potentially altered timing as discussed below).</td>
<td>Recruitment of juveniles and sub-adults prawns will likely decline due to a reduction in numbers of prawns exiting estuary. Decreased total banana prawn catch is probable. Furthermore, banana prawns exhibit high mortality rates in the estuary (Wang and Haywood, 1999) compared to offshore (Zhou et al., 2015). Increased magnitude of flow events may increase juvenile recruitment and emigration.</td>
<td>Strong body of evidence to support the mechanistic relationship between wet-season flows and juvenile prawn migration offshore (Vance et al., 1998). Vance et al. (1985) found a positive relationship with rainfall and subsequent adult recruitment in the Gulf of Carpentaria. Staples and Vance (1986) found that rainfall accounted for 70% of the observed variation in monthly emigration of juveniles in the Norman River.</td>
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<tr>
<td>Water extraction and river regulation</td>
<td>Life-history of prawns adapted to respond to flow variation and physiologically linked to salinity which is affected by flows.</td>
<td>Altered flow variability</td>
<td>Water extraction and regulation is likely to reduce flow variability. However, this may actually have benefits as optimal flows that coincide with key life stages and transitions of prawns could be delivered. The provision of flows that correspond with key life-history transitions may occur and provide a means to stabilise and provide predictable catches, given they are of sufficient magnitude and duration to flood salt flats and stimulate primary production. Reduced estuarine salinity may lead to potential declines in juvenile growth and abundance in estuary. Subsequent reductions in numbers moving offshore to</td>
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<tr>
<td>Climate change</td>
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<th>1. HABITAT AND THREATS</th>
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<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
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<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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<tr>
<td>Regulation of flows may also mean that dry-season flows are much greater compared to usual dry season, reducing estuarine salinity. Climate change may lead to increased variability in inter-annual flood events.</td>
<td>fishery leading to decreased catches and lower future spawning stock. Increased inter-annual variability may lead to highly variable recruitment of juveniles into offshore spawning population. Leads to more variable fishery catches, having socio-economic implications.</td>
<td>Rather than a single flow event to trigger offshore movement, flood pulses throughout wet season are required as they are highly linked with offshore movement. Different size classes and abundance of prawns can be moved offshore depending on the timing of the pulse (Staples and Vance, 1987). Increased inter-annual variability likely under projected climate models (Giorgi et al., 2004).</td>
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<tr>
<td>Altered timing of seasonal flows</td>
<td>The timing of seasonal flows are critical as juvenile prawn movement is so dependent on it. Nutrients derived from both terrestrial sources and salt flats thought to be highly important for offshore productivity.</td>
<td>Increased variability of weather events under climate change may mean that wet-season rain timing shifts and become early and potentially more intense. Early seasonal flows may result in a loss of synchrony between adequate post-larval development and environmental cues to migrate. In regulated rivers, dams may capture the majority of first-season flows, which will then provide little hydrological benefit to the estuary and prawns.</td>
<td>Juvenile prawns may be inadequately developed when they migrate offshore, reducing survival and subsequent recruitment. There may be a loss of synchrony between nutrients transported to near-shore, and the migration cue delivered by wet-season rainfall and floods.</td>
<td>Likely to see changes in the timing, frequency, magnitude, and extent of extreme climate events such as floods and droughts (Cahill et al., 2013). Seasonal shifts towards early stream flows have been confirmed across western North America (Stewart et al., 2005). While long-term trends in northern Australia have shown increased magnitude and duration of seasonality, the expected seasonal rainfall has increased in uncertainty (Feng et al., 2013). Banana prawns are on the southern extremity of their range and the ‘evolved-life-history-seasonality’ best fits prawns in their northern range; e.g. major spawning event in autumn is lost in lower southern hemisphere and populations (and subsequent catches) are primarily reliant on spring spawning event (Rothlisberg et al., 1985). So timing is critical, the species is already in the ‘edge’ of its life-cycle tolerances and further disynchrony may have severe negative population consequences.</td>
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<td>Altered dry-season flows</td>
<td>Lack of significant dry-season flows play important role supporting post-larval and juvenile prawns in the estuary, providing increase in dry-season flows through releases of water to serve irrigators downstream (as the unused portion then continues downstream)</td>
<td>With both reduced and increased dry-season flows, reduced estuarine recruitment is likely during dry season and preceding seasonal floods, leading to lower recruitment</td>
<td>In the Ord River, water resource development led to an increase in freshwater flows entering the river downstream from irrigated farmland during the dry season (increases of up to 439% - Pusey et al. (2011)), subsequently reducing salinity, primary production and</td>
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<td>1. HABITAT AND THREATS</td>
<td>2. DRIVER/STRESSOR</td>
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<tr>
<td><strong>Mangrove and saltpan/floodplain</strong></td>
<td>Change to wet-season flood magnitude</td>
<td>Saltpan inundation associated with magnitude of seasonal floods.</td>
<td>Optimal growing conditions and abundance food sources. Maintenance of baseflow supports the estuarine ecotone optimal for growth and survival.</td>
<td>Would be a big issue for banana prawns as it may be detrimental to major food sources of juveniles, along with favourable growing conditions. A reduction in dry-season flow expected to have less impact, however, hypersaline conditions also can physiologically inhibit prawn growth.</td>
<td>Decreased dry-season salinity caused by river regulation or climate induced shifts in seasonal rainfall will reduce prawn food sources and compromise growth. Too low salinity may lead to reduced juvenile prawn abundance in estuaries that have the potential to migrate offshore during wet-season flows, i.e. reduced recruitment. Too high salinity may lead to reduced growth rates or high mortality, also reducing the abundance of estuarine juveniles available to emigrate. Altered prawn distribution in estuaries may also have negative consequences on growth, predator avoidance and migration.</td>
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<td><strong>Water extraction and</strong></td>
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Changes in salinity

Both spatially and temporally, salinity is a vital component that influences life-history transitions, growth, and prey abundance of prawns.

Changes in salinity can have cascading effects on prawns as brackish conditions are optimal. Both hypo and hypersaline conditions can affect prawn growth, the abundance of food sources, as well alter important migratory cues. The extent of habitable/optimal saline conditions may shift throughout the estuary, potentially reducing available habitat.

Decreased dry-season salinity caused by river regulation or climate induced shifts in seasonal rainfall will reduce prawn food sources and compromise growth. Too low salinity may lead to reduced juvenile prawn abundance in estuaries that have the potential to migrate offshore during wet-season flows, i.e. reduced recruitment. Too high salinity may lead to reduced growth rates or high mortality, also reducing the abundance of estuarine juveniles available to emigrate. Altered prawn distribution in estuaries may also have negative consequences on growth, predator avoidance and migration.

Experimental studies have demonstrated that salinities of ~25 are optimal for juvenile white banana prawn growth (Staples and Heales, 1991). Extreme salinities (5 and 55) impeded growth while ranges between 20 and 45 were more optimal (Staples and Heales, 1991). Large peaks of emigration out of estuaries are associated with salinity decreases caused by wet-season rainfall (Vance et al., 1998). Prawn osmoregulation has been demonstrated to become inhibited at salinities below 7ppt, while levels of around 3ppt were shown to be the lethal limit (Dall, 1981).

Annual flooding of salt pans expands habitat available to white banana prawn. Desiccated algal crust on salt pans invigorates and begins to photosynthesize. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain. Extent of flooding more important than duration (Burford et al., 2016).
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<tr>
<td>river regulation</td>
<td>Climate change</td>
<td>extensive saltpan habitats. The seasonal pulse of key nutrient sources is dependent on flow magnitude.</td>
<td>estuarine production and its contribution to flood plume/near-shore productivity.</td>
<td>It is likely that the freshwater overbank flows are too fresh for banana prawns, thus they are not inhabiting the saltpans; the export of the nutrients to the flood plume zone is the main factor.</td>
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<td>Altered salinity</td>
<td>Salinity influences saltpan plant community structure and nutrient output to estuary and near-shore.</td>
<td>Climate change (drying) may increase temperature and evaporation rates on saltpans, leading to increases in salinity. Conversely, a wetter climate may reduce saltpan salinity.</td>
<td>Climate change may therefore influence the productivity of saltpan habitats and alter the amount of nutrient they provide to prawns.</td>
<td>Climate change (drying) may increase temperature and evaporation rates on saltpans, leading to increases in salinity. Conversely, a wetter climate may reduce saltpan salinity (Hughes, 2004). Increased frequency of saltpan inundation may stimulate microphytobenthos more often; potentially increasing productivity.</td>
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<td>Sea-level rise associated with climate change</td>
<td>Mangroves are a key habitat component required in the white banana prawn life cycle. Saline intrusion into wetland habitats.</td>
<td>Mangrove sediment accumulation cannot match rates of sea-level rise. Increased extent of shallow littoral habitat available to banana prawns.</td>
<td>A loss of mangrove habitats would be highly detrimental for post-larval and juvenile prawn survival as mangroves provide a habitat to shelter from predators, along with habitat for prawn food sources. Mangrove forest trap sediments and create the mud bank habitats which are used by post-larval and juvenile prawns. Possible increase in juvenile banana prawn population due to increased extent of habitat. Subsequent emigration increases the near-shore and offshore population of banana prawns.</td>
<td>Mangroves are well known to be nursery areas for juvenile prawns (Vance et al., 1990). Climate change is highly likely to negatively affect mangrove habitats through predicted sea-level rise and the increased occurrence of extreme climate events that may physically damage mangrove forests (Gilman et al., 2008). The interchange of expanded littoral saline habitats verses the response of key species that provide ecosystem services for banana prawns is difficult to predict.</td>
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<td>Low flow with water extraction</td>
<td>Mangroves provide key habitat for post-larval and juvenile prawns.</td>
<td>Increased water extraction will increase the relative effect of SLR on mangrove habitats.</td>
<td>Increased mangrove stress and vulnerability to both acute and long-term changes in climate conditions.</td>
<td>Anthropogenic responses to climate change including increasing water extraction for agricultural or human consumption may lead to relative sea-level rise exceeding the adaptive capacity of mangroves (Gilman et al., 2008).</td>
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<td>Near-shore habitat</td>
<td>Reduced wet-season flood magnitude</td>
<td>Nutrient transfer to near-shore and offshore marine zones during large floods support abundant populations of recently emigrated sub-adult prawns.</td>
<td>Reduced carbon input from terrestrial sources. Believed that nutrient inputs from terrestrial zones provide huge amount towards near-shore productivity and prawn growth following emigration from estuary.</td>
<td>Less terrestrial and salt flat inundation and subsequent nutrient input into near-shore zone reduces food sources for recently emigrated sub-adult prawns.</td>
<td>Flood inundation of salt flats provides important source of carbon and nitrogen for near-shore zones (Burford et al., 2016; Burford et al., 2012; Burford et al., 2011). In white banana prawns, fecundity found to be related to carapace length as prawns between 28 – 45mm had between 100 000 and 450 000 ripe ova respectively (Crocos and Kerr, 1983).</td>
</tr>
<tr>
<td>Water extraction and river regulation</td>
<td>Drier climate change</td>
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<tr>
<td>All habitats</td>
<td>Altered hydrological regime and increasing variability</td>
<td>Altered biotic interactions. For example, shifts in predation, competitive interactions.</td>
<td>Little known about how/if biotic interactions (e.g. competition and predation) will change under altered hydrological regimes or under climate change.</td>
<td>Potential for increased pressure from predation if flow alteration alters life histories of larger predators such that flow synchrony favours growth and survival of predators including barramundi. New predators may become evident if feeding pressures shift. Competition for feeding/habitat may reduce prawn growth and survival.</td>
<td>Proposed that biotic interactions may significantly affect fit and predictive ability of bioclimatic models (Araújo and Luoto, 2007). Freshwater input and subsequent turbidity associated with high flows in the Norman River determined types go predators present and changed their prey preferences (Salini et al., 1998).</td>
</tr>
<tr>
<td>Water extraction and river regulation</td>
<td>Climate change (variability and intensity of events)</td>
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</table>
4.2.2 MULLET

Asset description

Mullet (a guild from the genera *Liza*, *Mugil* and *Valamugil*) are catadromous fish (or marine migrants (Morgan et al., 2014) that use ‘catchment to coast’ (freshwater, estuarine and marine) habitats as part of their life history (Marin et al., 2003; Whitfield et al., 2012). They are distributed in tropical and temperate coastal waters worldwide. About 20 species of tropical mullet occur in waters from Townsville on the east coast to Broome in WA (Blaber et al., 2010). Diamond scale mullet (*Liza vaigiensis*), largescale mullet (*L. macrolepis*), greenback mullet (*L. subviridis*), sea mullet (*Mugil cephalus*), roundhead mullet (*Valamugil cunnesius*), bluespot mullet (*V. seheli*) and bluetail mullet (*V. buchanani*) are common species in the Australian tropics. Their distribution is not limited to Australia; they range across the Indo-Pacific (Whitfield et al., 2012).

Mullet are a short-lived, fast-growing productive fish species that are important as both a commercial and Indigenous fish resource. They are an abundant component of the fish community, being both forager and prey in the coastal ecosystem. Mullet are detritivores and are an ecological link species, transferring energy from low to high trophic levels in the estuarine fish community (Górski et al., 2015). Their position as detritivores in the food chain, fast growth rates and high fecundity makes them a species group with high harvest potential.

Distribution in Assessment study areas

Figure 4-21, Figure 4-22 and Figure 4-23 show the distribution of mullet in the Assessment catchments and marine study areas.
Figure 4-21 Distribution of mullet (fish family Mugilidae) in the Fitzroy catchment and marine study area
Figure 4-22 Distribution of mullet (fish family Mugilidae) in the Darwin catchments and marine study area
Physiology

Studies extending back to the 1950s have examined the biology of mullet in Australia. Many of these studies undertaken are from the subtropical and temperate coasts where significant seasonal trends in temperature, rainfall and winds are likely to initiate migration and movement. In the tropics (the Assessment study area), the response of mullet to environmental triggers is less well known, but their marine migrant life history remains dominant. (Grant and Spain, 1975b) and (Grant and Spain, 1975c) studied the biology of three species of tropical mullet in the Townsville (Queensland) region: diamond scale, sea and bluespot mullet.

Diamond scale mullet attained maturity at about 300 mm fork length (FL) (~ 3 years) and grew to about 500 mm FL (Grant and Spain, 1975a). They suggest growth is greatest over the monsoon season (October to March) and males and females grow at the same rates. Maturation and spawning occurs during the monsoon as well, with spawning occurring in February and March. Spawning occurs near-shore in deeper waters (not in estuaries) as eggs
and early larvae require marine-salinity waters. Individual females contained 800,000 to 1,200,000 eggs (Grant and Spain, 1975a).

Sea mullet spawn from May to August, more or less the opposite to diamond scale mullet (Grant and Spain, 1975a; 1975b). Sea mullet range from tropical to temperate waters (Górski et al., 2015) and their spawning strategy likely reflects successful spawning over a range of latitudes. They attain maturity at about 300 mm FL (~2.5 years) and grow to about 450 mm FL and 500 mm FL (males and females; ~7 years) (Górski et al., 2015). They suggest growth is greatest over the monsoon season (October to March). Spawning likely occurs in lower estuaries, as eggs and early larvae require saline waters. Individual females contained 1,500,000 to 4,700,000 eggs and therefore exhibit high fecundity (Grant and Spain, 1975b).

Bluespot mullet attained maturity at about 300 mm FL (~2.5 years) and grew to about 550 mm FL (Grant and Spain, 1975c), but larger individuals have been recorded (627 mm FL, 3.8 kg). They mature and spawn from October to December. Their suggested growth is greatest over the monsoon season (October to March) and males and females grow at the same rates. Spawning occurs in lower estuaries, as eggs and early larvae require saline waters. Individual females contained 275,000 to 1,080,000 eggs (Grant and Spain, 1975c).

More recent studies on the growth (Grant and Spain, 1975c; Kendall et al., 2009) and reproduction (Kendall and Gray, 2008) of co-generic species from south-eastern Australia provide detail on the reproductive biology of the mullet guild. Flat-tail mullet may grow to a maximum length of 45 cm TL, but more commonly are less than 350 mm TL (Carpenter, 1999),(Kendall et al., 2009) In south-eastern Australia, this species is slow-growing, reaching 50% of its maximum length by about age 5 to 8 years, and lives to about 17 years of age (Kendall et al., 2009). Histological analysis of gonads revealed that flat-tail mullet mature early in life, with 50% of females mature by around age 2 to 3 (207 mm TL) (Kendall and Gray, 2008). They spawn in the lower estuary and near-shore waters and spawning season varies geographically. In Lake Macquarie, NSW, the spawning period extends from March to October, whereas it extends from January to April at a more southern location. Juvenile and sub-adult, flat-tailed mullet use brackish and freshwater habitats during their first 1 to 2 years.

As a measure of abundance, the fishery catch of mullet has been correlated to regional river flow or rainfall, winter and annual flow (Loneragan and Bunn, 1999), rainfall and Southern Oscillation Index (SOI) (Meynecke et al., 2006), and total and minimum flow and salinity (Gillson et al., 2009). In contrast, a later study did not detect a relationship between rainfall (proxy for flow) and catch for mullet in south-eastern Queensland (Meynecke and Lee, 2011). This suggested that spawning migrations during winter are more likely to increase the abundance (and hence catch) of mullet in coastal waters.

For sub-adult and adult mullet, seasonal rainfall and flow likely influence their downstream movement, either through responses to changes in salinity or being carried by high flows (see Cardona, 2000) (Gillson et al., 2009). A riverine–estuarine to marine–open-sea spawning migration is a consistent feature of the life history of mullet (Robins et al., 2005). For example, on Australia’s east coast, 3-year lagged, high winter flows are negatively
correlated with the catch of mullet, suggesting that the flows flush newly spawned larvae from the estuaries. In contrast, high summer flows are positive for mullet abundance in estuaries and near-shore, suggesting adult mullet emigrate downstream from riverine habitats to the coast cued by seasonal floods (Gillon et al., 2009). In addition, the local extent of key habitats (wetlands) has a positive relationship to catches of mullet (with wetlands defined as mangrove communities, saltmarsh and intertidal mud or sand flat). Wetland perimeter-to-area ratio and wetland number of patches were 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).

As stated above, few studies of tropical mullet in Australia have been undertaken. The review by (Whitfield et al., 2012) and the high-latitude habitat use study by (Górski et al., 2015) illustrate the plasticity in behaviour by mullet over their range of distribution. Behavioural plasticity suggests that the movement and habitat selection of tropical mullet populations, such as within the Assessment study areas, may differ from elsewhere. Mullet are found in the riverine reaches of rivers in all three study areas, 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). In northern Australia, tropical coastal and riverine ecosystems are episodic systems, dependent on the annual wet season (Jackson et al., 2008). Reduced access to riverine, estuarine and floodplain habitats would affect the abundance of tropical mullet, as it disrupts their life history by reducing access to key habitats. Not only do flood flows facilitate access to floodplain and wetland habitats, they stimulate productivity pulses in floodplain and supra-littoral habitats, including activating dormant primary producers on saltpans (Burford et al., 2016).

Fishery

Mullet can be a reasonably important by-product species to inshore commercial fisheries in the Gulf of Carpentaria. Although catch data from Queensland fisheries is not available by fishery, the majority of the commercial catch comes from the commercial N3 gill-net fishery (N3 is the Gulf of Carpentaria inshore gill-net fishery), which is often used as an important bait species in the mud crab and recreational fisheries. Only a very small proportion of the commercial catch is separated into species. Diamond scale mullet is often separated, but the majority of the catch is recorded as ‘mullet unspecified’, which limits the ability to assess long-term catch trends by species. The annual commercial catch of mullet has varied considerably, ranging from just 170 kg in 2011 to 8 t in 2003. In the past five years the mean annual catch was 1.4 t.

In the recreational sector, mullet (unspecified) were the fourth-most important species caught (by number) in the 2010 Queensland statewide recreational fishing survey. An estimated 32,443 fish were caught, all coming from estuarine areas. Assuming a weight of 200 g, the total catch of mullet in 2010 was approximately 6.5 t, with around 35% of fish estimated as being released. Mullet are an important bait species (live and dead) in the recreational fishery. Mullet catch is not recorded by charter fishers.
The Indigenous fishery also regards mullet as an important food species. The 2000/01 National Research Institute of Fisheries Science recorded mullet (unspecified) as being the fourth-most important species caught (by number) in Queensland. An estimated 68,573 fish were caught, constituting 9.5% of the total Indigenous fish catch. Assuming a weight of 200 g, the total catch of mullet by the Indigenous sector in 2001 was approximately 13.7 t. Catch data from the 1970s suggest that diamond scale mullet comprised 50 to 100% of all mullet catch in the Northern Territory (Grant and Spain, 1975a).

**Habitat use**

Mullet species occur in coastal, estuarine and fresh waters worldwide (Blaber et al., 1992; Blaber et al., 2010; Blaber et al., 1995; Marin et al., 2003; Rahman et al., 2015). At least seven species of mullet are abundant enough to be exploited in tropical Australia. They are detritivores and feed on organic detritus as well as benthic microalgae, foraminiferans, filamentous algae, meiofauna and small invertebrates, which they filter from extraneous matter using their mouthparts and gills (Górski et al., 2015; Soyinka, 2008; Whitfield et al., 2012). The mullet guild exhibit an ontogenetic shift in diet, from planktivorous larvae to juveniles that feed on benthic invertebrates and prey in the surface water column, to detritivore adults (Whitfield et al., 2012). 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, 1975a; Whitfield et al., 2012). They reach approximately 35 cm TL in 4 years, at which stage they have left nursery habitats for lower estuaries and 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, 1975b; Halliday and Robins, 2005; Kailola et al., 1993). The larvae enter the estuaries, and small and large juveniles reside in estuarine habitats and move upstream to freshwater reaches of the rivers (Górski et al., 2015). In locations where the coastal geography and climate limit the physical development and extent of estuaries and access to them, mullet can undertake their life cycle exclusively in coastal inshore marine waters (Whitfield et al., 2012). Commonly, adult mullet are commonly found in both estuarine and freshwater habitats, although otolith microchemistry shows that some individuals occupy wholly marine habitats, despite available access to nearby estuaries (Górski et al., 2015).

Freshwater and brackish water are the preferred habitat for mullet in their juvenile and early adult phase (ontogenetic upstream movement). Experimental studies suggest that juvenile mullet require access to fresh or brackish waters for optimal growth (Cardona, 2000; Whitfield et al., 2012). If these habitats cannot be accessed, their growth and survival is affected. Disrupted connectivity between estuarine and riverine habitats by built barriers may limit their use of freshwater habitats (Grant and Spain, 1975a; Robins and Ye, 2007; Stuart and Mallen-Cooper, 1999). Fishways have been installed in river barriers to enable the upstream migration of mullet and other species, which has been interrupted by barrier construction (Stuart and Mallen-Cooper, 1999). Juvenile fish may suffer high mortality in the
vicinity of fishways, because they accumulate downstream in an attempt to migrate. During periods of low baseflow, fishways may not operate effectively. In addition, poorly designed barriers may create hypersaline conditions downstream in the estuary, resulting in physiological stress to migrating fish.

The growth of juvenile sea mullet showed greater length and weight records for fish grown in low-salinity environments (<5 ppt) than at higher salinities (>15 ppt) (Cardona, 2000). High metabolism at the higher salinities showed that the fish consumed more energy to attain the growth they achieved at higher salinities. In addition, habitat-preference studies found that small mullets (<200 mm FL) use both fresh and minimally saline (oligohaline) waters (<5 ppt), similar to mid-sized individuals (200 to 300 mm FL). Large individuals (>300 mm FL) preferred brackish (polyhaline) and marine waters (seasonally dependent) and avoided freshwater if possible (Cardona, 2000).

Conceptual model

Figure 4-24 shows the conceptual model for mullet, with the accompanying narrative given in Table 4-7.

Figure 4-24 Conceptual model showing the relationship between threats, drivers, effects and outcomes for mullet in northern Australia
## Conceptual model narrative

### Table 4-7 Knowledge underpinning the conceptual model for tropical mullet in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2. DRIVER /STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall.</td>
<td>Maintenance of baseflow. Timing and amplitude of monsoonal rainfall and flooding important in maintaining larval access to estuaries and juvenile habitats (estuarine ecotone).</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus, the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal salinity (~5-10) for juvenile and sub-adult mullet is lost, thus critical habitat is lost. (Cardona, 2000); (Whitfield et al., 2012). Weak/ out-of-sync monsoon low rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. Baseflow occurs out of sync with the spawning season of mullet. The horizontal extent of brackish waters decreases or is lost to the estuary. High or artificially elevated baseflow may impact the ability of larvae to access estuaries in the early stages due to low salinity (Gillson et al., 2009).</td>
<td>The reduction of brackish water habitat reduces the ability of mullet population to forage and grow in the estuary. Mullet populations decline. Strong flows during spawning and larval immigration may impede the movement of larvae from early marine habitats to the estuary.</td>
<td>Preference for and higher growth rates in brackish waters by juvenile and sub-adult mullet (Cardona, 2000; Whitfield et al., 2012). Negative correlation between mullet fishery catch (CPUE) and high winter flows along the NSW coast (Gillson et al., 2009).</td>
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<tr>
<td>Reduction in flood flow inundation/ flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows. Timing of monsoonal rainfall and flooding important in maintaining juvenile habitats (estuarine ecotone)</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the persistence of the ecotone during ‘autumn’ and early winter when the spawning of mullet occurs. Optimal habitat (salinity ~5 to 20) for juvenile and sub-adult mullet is lost. Possible reduction in emigration cue for 2+ mullet that have been resident in estuaries for 2 to 3 years as older mullet prefer</td>
<td>Reduction in brackish water habitat limits the ability of mullet to forage and grow in the estuary (Whitfield et al., 2012). Mullet populations decline. Reduction in emigration cue for adult mullet that</td>
<td>Preference for and higher growth rates in brackish waters by juvenile and sub-adult mullet (Cardona, 2000; Whitfield et al., 2012). Positive correlation between mullet fishery catch (CPUE) and high annual flows along the NSW coast and regions of the Queensland coast (Gillson et al., 2009; Loneragan and Bunn, 1999).</td>
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<td>1 HABITAT AND THREATS</td>
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<td>Marine to polyhaline (Traill and Brook, 2011). However, the spawning migrations of mullet from rivers to the sea confound the stimulus for downstream migration (Robins et al., 2005).</td>
<td>Reduction in flood flow volume</td>
<td>Maintenance of volume and duration of peak flows.</td>
<td>Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the floodplume.</td>
<td>have been resident in estuaries for ~3 years.</td>
<td>Reduction in flood flow volume and duration reduces nutrient export (Burford et al., 2012; Burford et al., 2011).</td>
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<tr>
<td>Inter-annual connectivity of floodplain habitats to riverine and estuarine habitats.</td>
<td>Change in inter-annual sequencing</td>
<td></td>
<td>Juvenile mullet that access floodplain habitats or longstream habitats benefit from access to extensive and productive habitat. But in subsequent years, they must return to the estuary and marine habitats as adults and to spawn.</td>
<td></td>
<td>Catadromous life history (Whitfield et al., 2012).</td>
</tr>
<tr>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion reduces the extent of presence of a brackish ecotone which is optimal habitat for juvenile mullet.</td>
<td>Saltwater intrusion</td>
<td></td>
<td>Reduction in mullet population due to decline in habitat extent and productivity at key times in the annual climate cycle.</td>
<td></td>
<td>Preference for and higher growth rates in brackish waters by juvenile and sub-adult mullet (Cardona, 2000; Whitfield et al., 2012).</td>
</tr>
<tr>
<td>Annual flooding of salt flats and wetlands mark an inundation expansion of habitat available to mullet. Desiccated algal crust on salt pans invigorates and begins to photosynthesise. Carbon and nitrogen production occur with input to the estuarine ecosystem</td>
<td>Estuarine/saltwater wetland and floodplain</td>
<td>Change in inundation frequency, depth and duration Change in land use and management</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation salt pan and wetland adjacent to tropical estuaries reduces key habitat that becomes available to estuarine fauna during the monsoon season. Reduction in habitat available to juvenile and sub-adult mullet during monsoon season large flood flows.</td>
<td></td>
<td>Reduction in nutrient production from flooded salt pans during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients exported from tropical estuaries with subsequent reduced coastal productivity for species with ontogenetic habitat shifts; i.e. species that emigrate from</td>
</tr>
<tr>
<td><strong>1. HABITAT AND THREATS</strong></td>
<td><strong>2. DRIVER / STRESSOR</strong></td>
<td><strong>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</strong></td>
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<tr>
<td>Longstream habitat connectivity</td>
<td>Loss of connectivity between spawning and juvenile habitats (marine, estuarine and riverine reaches of rivers) due to the placement of barriers in the upper estuary or lower river reach; or to either low or nil flow causing the fluvial separation of waterholes above the estuary.</td>
<td>Juvenile mullet migrate upstream from estuaries to freshwater reaches of rivers where they reside for 1 to 2 years as growth and survival is optimal in brackish and freshwater habitats (Whitfield et al., 2012).</td>
<td>Physical barriers block or impede upstream migration of juvenile mullet to key habitats (Rolls et al., 2014). Fishways assist migration over barriers with varying levels of success (Stuart and Mallen-Cooper, 1999). The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal salinity (~5 to 10) for juvenile and sub-adult mullet is lost, thus critical habitat is lost (Cardona, 2000; Whitfield et al., 2012), Reduced access to critical habitat for juvenile and sub-adult mullet. Reduced habitat extent. Slow growth of mullet in marine-salinity estuarine waters (Cardona, 2000).</td>
<td>Reduced access to critical habitat for juvenile and sub-adult mullet. Reduced habitat extent. Slow growth of mullet in marine-salinity estuarine waters (Cardona, 2000).</td>
<td>The real-time impact of barrier construction in the lower reaches of rivers has rarely been documented. Many barriers are evident, but they were placed in the 1800s or early 1900s and no impact assessment was undertaken. Anecdotal accounts support the contention that mullet populations decline drastically when longstream migration barriers are placed. Reduction in the abundance of the mullet population in the Brisbane River, Queensland is an example. Accounts from the 1920's comment on an annual migration of mullet from the sea to the upper reaches of the river that was visually impressive and exploited by local residents - “dad would row out at night with a lantern on the front of the boat and in minutes scoop up a couple of hundred” (Gregory, 1996). The construction of a higher Mt Crosby Weir in 1926 created a barrier to upstream migration and weir construction coincided with a rapid decline in fish abundance due to their inability to migrate. Importantly, public pressure at the time saw the government attempt to physically transport mullet in</td>
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<td>1 HABITAT AND THREATS</td>
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<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
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<tr>
<td>Riverine floodplain wetlands</td>
<td>Reduced access to waterhole/wetland habitats on floodplains.</td>
<td>Juvenile habitat during flood events and monsoon season.</td>
<td>Reduction in access to habitat and seasonal resources required to maintain robust population.</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
<td>Synthesis of literature.</td>
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<td>Reduced access due to less frequent and lower flood events</td>
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<td>Invasion species (e.g. fish fauna)</td>
<td>Competition for resources and degradation of habitat.</td>
<td>Invasive species such as Gambusia compete for the same resources (Darwin and Fitzroy Rivers)</td>
<td>Interspecific competition. Possible predation by exotic species.</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
<td>Synthesis of literature. Gulf of Carpentaria inshore gill-net fishery catch statistics.</td>
</tr>
<tr>
<td>Conflicting land use</td>
<td></td>
<td>Habitat (water quality) degradation by invasive animal species such as pig and buffalo.</td>
<td>Loss of habitat due to degraded water quality due to impacts on waterholes and lagoons from pigs, for example.</td>
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<tr>
<td>Fishing harvest</td>
<td></td>
<td>Modification/draining of wetlands for human use e.g. agriculture.</td>
<td>Loss of reproductive potential and genetic diversity.</td>
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<tr>
<td>Water column habitats all waters</td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation.</td>
<td>Loss of water clarity, light penetration. Loss of water column productivity/ algal productivity.</td>
<td>Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of planktonic and benthic food items for mullet.</td>
<td>Synthesis of literature.</td>
</tr>
<tr>
<td>Sediment loads</td>
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</table>

- tanks upstream to compensate (Gregory, 1996). These events show that a major mullet population decline occurred; coincident with an impediment to migration. Clearly, scooping up hundreds of mullet from a dinghy in the Brisbane River no longer occurs in 2016.
### 4.2.3  MUD CRAB

**Asset description**

The mud crab, *Scylla serrata*, is a short-lived, fast-growing crustacean species that is important as a commercial, recreational and Indigenous fish resource. They are large-clawed, large-bodied decapods (>200 mm carapace width) that are targeted throughout their range as a commercial catch (40,000 t worldwide in 2012) (Alberts-Hubatsch et al., 2016). They are distributed across the Indo–Pacific, from southern Africa to south-east Asia (including southern Japan) and Australia to Samoa in the Pacific Ocean. Of the two species of mud crab found in tropical Australia, *S. serrata* and *S. olivacea* (Alberts-Hubatsch et al., 2016), the former species is more abundant, and is the dominant commercial species in Australia (Alberts-Hubatsch et al., 2016) and the focus of this review. Mud crabs are well studied, though their habits and habitat in specific locations as remote as the Assessment’s target rivers limit knowledge of their use of these rivers.

Mud crabs are an important ecological species. They are a component of the crustacean community, being both predator and prey in the coastal ecosystem. Mud crabs dig burrows in which they rest during the day, reworking mud substrates within mangrove forests and mudbanks. As small juveniles, mud crabs are detritivores; as large juveniles and adults, they are benthic predators feeding on crustaceans, molluscs and fish. Estimates suggest that the mud crab community consumes 650 kg biomass/ha/year in the mangrove forest and 2100 kg biomass/ha/year in mangrove fringe habitat (Alberts-Hubatsch et al., 2016). They likely play a significant trophic role in mangrove ecosystems. Mud crabs are a prized eating species that command a high market price and are fished in remote coastal locations in tropical Australia. The economics of their harvest allow them to be air freighted live out of remote Australia to markets in southern population centres.

The mud crab’s large size and large claws have made it a prized food item throughout the tropics for over 2000 years (Beech and Hogarth, 2002; Beech 2002; Hogarth and Beech, 2001). They are farmed in aquaculture situations, with their 2012 production being 175,000 t (Alberts-Hubatsch et al., 2016). Mud crabs have been studied since the 1940s. They were initially considered one species, but this was revised in 1998 to four species. Mud crabs support major inshore fisheries across tropical and subtropical Australia and typify a life-history strategy that would be significantly affected by interruptions to the natural flows of the dry and wet rivers of Cape York, the Top End and the Kimberley.

**Distribution in Assessment study areas**

Figure 4-25 Distribution of mud crabs in the Fitzroy catchment and marine study area. Figure 4-25 and Figure 4-26 show the distribution of mud crabs in two of the Assessment catchments and marine study areas. There are no records of mud crabs in the Mitchell catchment.
Figure 4-25 Distribution of mud crabs in the Fitzroy catchment and marine study area
Habitat use

Over 40 years of scientific research has explored the behaviour, habitats, feeding and ontogenetic movement of each stage of their life history (Gopurenko and Hughes, 2002; Heasman et al., 1985; Hill, 1980; 1994; Hyland et al., 1984; Moser et al., 2003). Crabs across large areas of the tropics are genetically mixed stocks, but significant barriers to hydrological connectivity (e.g. Cape York and Torres Strait) also create distinct populations (Gopurenko and Hughes, 2002).

Mud crabs have a larval life-history strategy: the adults mate in the estuary and the females migrate to spawn in the offshore zone (September to November; larvae require marine salinity) (Hill, 1975; 1994; Welch et al., 2014). Their eggs hatch to larvae that transform to megalopae, which are advected inshore where they settle as benthic juveniles in estuarine mangrove and mudflat habitats (Alberts-Hubatsch et al., 2016; Meynecke et al., 2010). The larval form facilitates not only ontogenetic migration to their inshore habitats, but long-distance dispersal and genetic mixing (Gopurenko and Hughes, 2002; Gopurenko et al., 2003). Mud crabs remain in the estuary for several years as sub-adults and adults, before
the females emigrate to spawn (Hill, 1994). They are fished all year round. For some river systems, the annual wet season and subsequent runoff is a significant determinant of their recruitment strength and total catch (possibly lagged by one to two years) in the estuary and near-shore (Meynecke et al., 2010).

Mud crab megalopae, juveniles and adults use different estuarine and coastal micro habitats. However, mud crabs do not demonstrate ontogenetic emigration from their juvenile habitat. Both juvenile and adult habitats are associated with well-established mangrove forests (Alberts-Hubatsch et al., 2014; Meynecke et al., 2010). Megalopae settle to become benthic juveniles in muddy estuarine habitats, predominantly within the intertidal mangrove fringe, and those that settle in adjacent muddy banks move to the forest shortly after (Alberts-Hubatsch et al., 2014). The adult habitats are spatially differentiated sub-tidal zones of the estuary or the close-in coastal zone (Hill et al., 1982; Hyland et al., 1984). Juvenile crabs (especially 20 to 80 mm carapace width) are resident in the intertidal zone where they select for structured habitat (Webley et al., 2009); whereas sub-adult (100 to 150 mm carapace width) and adult crabs (>150 mm carapace length) move into the intertidal zone on the flood tide, but retreat to the subtidal zone as the tide recedes. Some adult crabs remain in the intertidal zone, where they construct burrows as shelter from heat and to prevent desiccation. Adult males remain in the littoral zone, while ovigerous females emigrate offshore to spawn.

As a proxy for abundance, the relationship between mud crab commercial catch and environmental cues is not as clear as for some fishery species. Positive correlations between flow and crab catch have been identified in tropical, central and south-east Queensland (Loneragan and Bunn, 1999; Meynecke and Lee, 2011; Robins et al., 2005), though flow seasonality and two-year lag effects suggest that relationship is not clear cut. In southern Australia (NSW), temperature explained 30 to 50% of the seasonal and monthly mud crab catches. However, in northern Australia there is a correlation (30 to 40%) between the SOI and subsequent fishery catch (e.g. in the Flinders River) (Meynecke et al., 2010). A high positive SOI is associated with higher rainfall and temperatures in coastal Queensland, so the correlation between crab catch and environmental flow (runoff lagged by 6 months or 1 year) from south-east Gulf of Carpentaria rivers is supported via the SOI (Meynecke et al., 2010). In the Gulf of Carpentaria, large floods such as the 1-in-50-year flood of 2009 can reduce the coastal catch of mud crabs. The inshore zone becomes a freshwater habitat and crabs migrate elsewhere, suggested to be due to their intolerance of very low salinities, i.e. freshwater or near fresh. This observation is supported by the findings of (Ruscoe et al., 2004), where the survival of juvenile crabs in fresh water was nil, while survival in salinities of 5 ppt was high. In fact, survival between the salinities of 5 to 40 ppt that were tested was high and not significantly different.

Specific relationships between flow and catch have been identified for rivers within the Assessment study areas. The annual mud crab catch from the Mitchell River (catch per unit effort, CPUE) was significantly correlated with rainfall lagged by two years from the year of harvest (Meynecke et al., 2010). Similarly, the annual mud crab catch from the Adelaide River (CPUE) was significantly correlated with rainfall for the year of harvest, and for up to
one and two years after (Meynecke et al., 2010). Annual catch was also correlated with the level of the SOI.

**Physiology**

Mud crabs spawn offshore during September to November as their larvae require marine salinities (25 to 30 ppt) and warm waters (26 to 30 °C) for optimal growth (Welch et al., 2014), (Alberts-Hubatsch et al., 2016). Megalopae are tolerant of 15 to 45 ppt salinity, facilitating their occupation of diverse inshore habitats where physical parameters are more 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 and a salinity of 10 to 20 ppt (growth) and 10 to 30 ppt (survival) (Meynecke et al., 2010; Ruscoe et al., 2004). For short periods, they can tolerate cool conditions (<20 °C), but require temperatures greater than 20 °C to grow and function (~25 to 30 °C is optimal). Juvenile mud crabs that are resident in estuaries can tolerate a broader salinity range (5 to 45 ppt); they benefit from 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 crabs may be inhibited until first rains and low flows reduce the salinity in the tropical estuaries to brackish levels. Adult mud crabs are a euryhaline phase of the life cycle, capable of living in freshwater-flooded to hypersaline waters (<5 to 45 ppt) (Alberts-Hubatsch et al., 2016).

**Fishery**

Mud crabs have been fished commercially in the Gulf of Carpentaria since the 1970s, in the Northern Territory since the 1980s and in the Kimberley region since 2003 (though West Australian catch data exist from 1977) (Meynecke et al., 2010). *S. serrata* is the target species of the Queensland pot fishery, but is also an important by-product species in the N3 inshore gill-net fishery. *S. serrata* is the dominant mud crab species in the Northern Territory fishery, though approximately 1% of crabs caught in the Darwin catchments and to the west likely are *S. olivacea*. In northern West Australia, the highest catches of mud crabs are taken in King Sound, with Admiralty and Cambridge Gulfs other locations of good catches. *S. olivacea* comprises a significant portion of the West Australian mud crab catch and perhaps is the dominant crab species in King Sound, the receiving waters of the Fitzroy River (Meynecke et al., 2010).

Mud crab is a highly important species in the Gulf of Carpentaria and Northern Territory for commercial, recreational and Indigenous fisheries (Table 4-8). The total commercial catch of mud crabs in the Queensland region of the Gulf of Carpentaria has steadily increased from 23 t in 1993 to 199 t in 2012 (Griffiths et al., 2014). Catches have remained reasonably stable over the years from 2008 to 2013 averaging 183 t/year.

The Northern Territory commercial fishery has existed since the 1980s, but expanded geographically in the 1990s due to increased road access to remote coastal locations.
Territory-wide, mud crab catch increased from approximately 200 t in the early 1990s to approximately 1000 t from 1998 to 2001, then declined to approximately 400 t in 2008.

In WA, the commercial mud crab fishery is at an exploratory phase and the catch is small (~6 t in 2008) (Table 4-8). Commercial fishing began in 2003 (early annual catches ~1 t) and continues today. Effort increased ten-fold over the 2000s. The remote location of mud crab habitats in northern WA relative to southern markets limits the economic viability of the fishery. The majority of commercial fishing in WA occurs in King Sound and the estuary of the Fitzroy River.

Mud crabs are important to recreational fishers. The Queensland 2010 statewide survey estimated 13,000 mud crabs were caught by recreational fishers, equating to around 13 t of mud crabs (assuming a mean weight of 1 kg). Approximately 50% of this catch was released (QFish online database). In the Gulf of Carpentaria, the charter catch averaged only 682 kg over the past 5 years. The 2000–2001 National Recreational and Indigenous Fishing Survey (Henry and Lyle, 2003) estimated 12,874 mud crabs were caught by Indigenous fishers, equating to 12.8 t: a similar catch to the non-Indigenous take. However, these estimates are for all of Queensland.

<table>
<thead>
<tr>
<th>2008 DATA</th>
<th>QUEENSLAND</th>
<th>NORTHERN TERRITORY</th>
<th>NEW SOUTH WALES</th>
<th>WESTERN AUSTRALIA</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recreational bag limit</td>
<td>10</td>
<td>10</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Commercial licences</td>
<td>431</td>
<td>49</td>
<td>217</td>
<td>6</td>
</tr>
<tr>
<td>Commercial catch (t)</td>
<td>1025</td>
<td>412</td>
<td>107</td>
<td>6.4</td>
</tr>
<tr>
<td>Commercial effort (d)</td>
<td>38000</td>
<td>11122</td>
<td>15000</td>
<td>323</td>
</tr>
<tr>
<td>Commercial CPUE (kgd⁻¹)</td>
<td>27</td>
<td>37</td>
<td>7</td>
<td>20</td>
</tr>
<tr>
<td>Recreational catch (t)</td>
<td>600-800</td>
<td>~135</td>
<td>30-60</td>
<td>21 est</td>
</tr>
<tr>
<td>Management measures</td>
<td>Log books, annual fishery independent survey</td>
<td>Demographic data collected from commercial samples (monthly)</td>
<td>Port based monitoring of commercial catch</td>
<td>Logbooks, no regular monitoring</td>
</tr>
</tbody>
</table>

Source: (Meynecke et al., 2010)

**Conceptual model**

Figure 4-27 shows the conceptual model for mud crabs, with the accompanying narrative given in Table 4-9.
Figure 4-27 Conceptual model showing the relationship between threats, drivers, effects and outcomes for mud crabs in northern Australia.
### Conceptual model narrative

**Table 4-9 Knowledge underpinning the conceptual model for mud crabs in northern Australia**

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/ regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow. Timing and amplitude of monsoonal rainfall and flooding important in maintaining a brackish estuarine ecotone as juvenile habitat. Possible immigration cues for larval/megalopae access to estuaries.</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus, the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal salinity (~10 to 20) for juvenile mud crabs is lost, thus critical habitat is lost (Alberts-Hubatsch et al., 2016). Weak/out-of-sync monsoon low rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. Baseflow occurs out of sync with the spawning season of mud crabs. The horizontal extent of brackish waters decreases or is lost to the estuary. High or artificially elevated baseflow may impact the ability of larvae to access estuaries during their early stages due to unseasonal very low salinity during the spawning season (Alberts-Hubatsch et al., 2016; Ruscoe et al., 2004).</td>
<td>The reduction of brackish water habitat reduces the ability of mud crab population to forage and grow in the estuary. Mud crab populations decline. Strong flows during spawning and megalopae immigration may impede the movement of larvae from early marine habitats to the estuary; both due to freshwater extent and physical displacement by outflows.</td>
<td>Preference for and higher growth rates in brackish waters by juvenile mud crabs (Ruscoe et al., 2004). Positive correlation between mud crab fishery catch and summer flows in the Logan River, Queensland (Loneragan and Bunn, 1999).</td>
</tr>
<tr>
<td>Reduction in flood flow inundation/ flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows. Timing of monsoonal rainfall and flooding important in maintaining estuarine ecotone as juvenile habitat.</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the persistence of the ecotone during ‘autumn’ and early winter when juvenile crabs are growing. Optimal habitat (salinity ~10 to 20) for juvenile mud crabs is lost.</td>
<td>Reduction in brackish water habitat limits the ability of mud crabs to forage and grow in the estuary (Alberts-Hubatsch et al., 2016; Meynecke et al., 2010; Ruscoe et al., 2004). Crab populations</td>
<td></td>
<td>Preference for and higher growth rates in brackish waters by juvenile mud crab (Alberts-Hubatsch et al., 2016; Ruscoe et al., 2004). Positive correlation between mud crab fishery catch (CPUE) and high</td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/ STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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</tr>
<tr>
<td>Loss of habitat extent Habitat degradation/ homogenisation</td>
<td>Saltwater intrusion</td>
<td>Elevation in salinity regime in extensive coastal wetland habitats in low-gradient geo-scapes in tropical coastal ecosystems across northern Australia. Spatial expansion of saline habitat.</td>
<td>Possible reduction in cue for adult crabs to move to the lower estuary and become available to fishers.</td>
<td>Annual flows in tropical rivers of the Northern Territory and Queensland; and southern regions of the Queensland coast (Loneragan and Bunn, 1999; Meynecke and Lee, 2011; Meynecke et al., 2010).</td>
<td></td>
</tr>
<tr>
<td>Estuarine/ saltpan wetland and floodplain</td>
<td></td>
<td>Elevation in salinity regime in extensive coastal wetland habitats in low-gradient geo-scapes in tropical coastal ecosystems across northern Australia. Spatial expansion of saline habitat.</td>
<td>Reduced monsoonal flows into coastal freshwater wetlands resulting in increased saltwater intrusion. Storm intensity increase and sea-level rise as a result of climate change causes saltwater intrusion to freshwater wetlands. Saltwater intrusion increases the extent or presence of saline influence in supralittoral habitats which may increase the spatial extent of optimal habitat for juvenile and adult mud crabs.</td>
<td>Increase in mud crab population due to expansion in brackish water habitat extent and productivity at key times in the annual climate cycle. Increased extent of brackish water habitats where flora, gastropods, annelids, crustaceans and fish are prey species consumed by juvenile and sub-adult mud crabs.</td>
<td>Higher growth rates in brackish water habitats by juvenile mud crabs (Alberts-Hubatsch et al., 2016; Ruscoe et al., 2004; Welch et al., 2014). The productivity of coastal juvenile habitats is likely to be a major driver of mud crab abundance. The diverse range of possible drivers in coastal habitats requires more scientific investigation (Alberts-Hubatsch et al., 2016; Welch et al., 2014).</td>
</tr>
<tr>
<td>Change in inundation frequency, depth and duration</td>
<td></td>
<td>Annual flooding of salt flats mark an inundation expansion of habitat available to mud crabs. Desiccated algal crust on saltpans invigorates and begins to photosynthesise. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016).</td>
<td>Disruption to the frequency and depth (extent) of large areas of very low elevation and slope saltpan and wetland adjacent to tropical estuaries due to reduced flood flow reduces key habitat extent that becomes available to estuarine fauna during the monsoon season.</td>
<td>Reduction in population size of juvenile and sub-adult mud crabs and changes in distribution/habitat use. Crowding of adult mud crabs in the lower estuary.</td>
<td>Loss of habitat extent during moderate flood flows as overbank inundation does not occur. Reduction in nutrients exported from tropical estuaries with subsequent reduced coastal productivity for species with ontogenetic habitat shifts; i.e. species that emigrate from estuarine nursery habitats to near-shore habitats (Burford et al., 2016b; Burford et al., 2012).</td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/ STRESSOR</td>
<td>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
<td>4. EFFECT</td>
<td>5. OUTCOME</td>
<td>6. EVIDENCE AND PATHWAYS</td>
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<tr>
<td>Conflicting land use/ modification of land surface</td>
<td>Competition for coastal resources and degradation of habitat</td>
<td>Modification/infill/drainage of mangrove forests for human use e.g. coastal infrastructure.</td>
<td>Loss of habitat due to infill; modification of tidal regimes; degraded water quality due to impacts.</td>
<td>Reduction in population size or changes in distribution/habitat extent.</td>
<td>Impact on optimal conditions for crab growth and survival (Ruscoe et al., 2004).</td>
</tr>
<tr>
<td>Longstream habitat connectivity Construction of barriers</td>
<td>Truncation of estuary due to the construction of a barrier Loss of juvenile and adult habitats (estuarine reaches of rivers) due to the placement of barriers in the upper estuary Damming of low flows at low river levels causing the loss of brackish ecotone in the estuary</td>
<td>Juvenile mud crab use the full extent of the estuary; habitat extent is lost due to truncation of the estuary. Freshwater water flow to the estuary is lost and the estuary may become hypersaline. High salinity is not optimal for growth of juvenile mud (Ruscoe et al., 2004).</td>
<td>Physical barriers block the estuarine inflow of freshwater. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal salinity (~10-20) for juvenile mud crab is lost, thus critical habitat is lost (Alberts-Hubatsch et al., 2016; Ruscoe et al., 2004).</td>
<td>Reduced access to critical habitat for juvenile and adult mud crab. Reduced habitat extent. Slow growth of mud crabs in marine-salinity estuarine waters (Ruscoe et al., 2004).</td>
<td></td>
</tr>
<tr>
<td>Fishing harvest</td>
<td>Commercial fishing occurs in Gulf of Carpentaria and Darwin rivers</td>
<td>Licensed harvest. Interspecific competition. Possible predation by exotic species. Loss of habitat due to degraded water quality due to impacts on waterholes and lagoons from pigs, for example.</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
<td>Harvest of ~200 tonnes of mud crabs in Queensland water and ~400 tonnes in Northern Territory waters.</td>
<td></td>
</tr>
<tr>
<td>Water column habitats all waters Sediment loads</td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation Maintenance of depositional environment in the lower estuary that sustains mangrove habitats</td>
<td>Loss of water clarity, light penetration. Loss of water column productivity/ algal productivity. Loss of epibenthic productivity. Stable mangrove habitats support stable mud crab populations.</td>
<td>Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of planktonic and benthic food chain that sustains mud crabs.</td>
<td>Loss of water column and epibenthic photosynthesis destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
</tr>
</tbody>
</table>
4.2.4   LONGBUMS

Asset description

The longbum and the lesser longbum (mudwhelks, shellfish) (*Telescopium telescopium* and *Terebralia palustris*) are large, thick-shelled, gastropod molluscs (100 to 130 mm and ~120 mm, respectively; Potamididae) that inhabit mangrove forests and tidal mudflats throughout the Indo-west Pacific. In Australia, *T. telescopium* is distributed from Onslow in WA to Bowen in Queensland (Willan, 2013) and they continue to be abundant in coastal mangrove habitats. The lesser longbum, *T. palustris*, is distributed from Shark Bay to Townsville (Willan, 2013). Longbums have a larval life-history strategy, but little is known of their settlement and juvenile habitats, growth, movement, longevity or reproductive cues. Longbums are an important ecological species: they are detritivores that cycle detritus and leaves from primary producers (mangroves) and make nutrients available to other species in the food chain. They dominate the epibenthic biomass in some mangrove forests and play a key role in the coastal ecosystem.

Longbums typify an estuarine and mangrove-dependent mollusc that could be significantly affected by interruptions to natural river flows maintaining seasonal nutrient and sediment delivery, including flood flow flushing, to coastal habitats in tropical northern Australia.

Distribution in Assessment study areas

Figure 4-28 and Figure 4-29 show the distribution of longbums in two of the Assessment catchments and marine study areas. No records exist for the Mitchell River estuary; however, they do for other locations on western Cape York.
Figure 4-28 Distribution of mud whelk or ‘longbums’ (*Telescopium telescopium*) in the Fitzroy catchment and marine study area
Habitat use

Longbums occur in tropical coastal and estuarine mangrove forest and mudflat habitats, including creek banks and runoff gullies (Buckworth, 1995; Willan, 2013). They are amphibious and live in the intertidal zone, where they are active when exposed at low tide. They gain food, shading (temperature regulation), protection from predators and habitat substrate from the mangrove forests (Willan, 2013). In contrast, in some locations in South-East Asia and the Middle East, they survive in non-forested intertidal substrates (Willan, 2013). Longbums are most abundant in the mid-intertidal, associated with Ceriops forests (also mid-intertidal) in Darwin Harbour (Willan, 2013). In addition, the lesser longbum is found on the upper intertidal margin, as well as the mid-intertidal. Longbums have been observed clustering the shade of the mangrove canopy when resting during the heat of the day (Buckworth, 1995; Lasiak and Dye, 1986). In South-East Asia, they bury themselves as the incoming tide sweeps over them and re-emerge as the tide ebbs.
Longbums feed on detritus and algae found on the intertidal substrates. *T. telescopium* imbibes both particulate mud and the organic detritus that they target (including bacteria and contaminants) (French, 2013; Zaman and Jahan, 2015). *T. palustris* undergoes an ontogenetic shift in diet; as adults, the teeth in their radula change shape to allow them to rasp mangrove leaves. As detritivores, longbums imbibe many contaminants and chemicals associated with urban settlement adjacent to coasts, though to date the contaminants have not affected their health (French, 2013). Some studies show that they feed when exposed at low tide, during cool times within the 24 hour period, and during the day in winter and at night in summer (Zaman and Jahan, 2015). Other studies show that they became active and foraged during tidal inundation and moved up to 4 to 5 m/day. Mudwhelks that lived in the upper tide zone were inactive for long periods (days) between spring tide inundation, while those in the mid-tide zone were active when inundated by both spring and neap tides (Lasiak and Dye, 1986). They sheltered under the mangrove canopy when not inundated. Due to their impenetrable shell, large specimens have few predators except for mud crabs (Buckworth, 1995). Harvest of longbums may have local impacts on population dynamics in localities near coastal communities, and particularly in the Darwin urban footprint. Habitat loss due to urban or industrial development is evident in Darwin Harbour. However, the current footprint of habitat loss is small relative to the total extent of their remote habitats. Longbums can be affected by the loss of mangrove habitats through water and heat-stress dieback caused by reduced watering due to lower flood flow levels.

**Physiology**

The sexes are separate. Adults spawn eggs in the shallow subtidal waters, in gelatinous strings that hatch into free-swimming larvae that feed on phytoplankton (Buckworth, 1995; Willan, 2013). Little to no research has been conducted on their longevity, maturity, recruitment from the planktonic phase, growth rate, optimal environment or population resilience (Buckworth, 1995; Willan, 2013). Haque and Choudhury (2015) suggest that the snails can grow up to 80 to 100 mm/year. Due to the remote coastlines over much of tropical Australia and the low exploitation rate of longbums, it is highly probable that their current reproduction and population dynamics are not reduced by low adult stocks or low food abundance.

Seasonal studies of mudwhelks have shown that they survive in air temperatures of 24 to 34 °C and water temperatures of 14 to 29 °C. Salinities ranged from 11 to 23 ppt (Haque and Choudhury, 2015). Mudwhelks have been shown to spawn during summer in habitats with temperatures of 29 to 32 °C and salinities of 32 to 36 ppt. The egg mass has up to 50,000 eggs and larvae were observed hatching 96 hours after the eggs were spawned (Ramamoorthi and Natarajan, 1973).

Longbums accumulate chemical contaminants in their flesh, though with no detectable adverse effects. However, altered flows that reduce the flushing of point-source effluent by reduced flows might affect longbums, as the level of chemical contaminants increase in the habitats close to point-source effluent (French, 2013). In addition, longbums may be
affected by the transport of chemicals from irrigated lands associated with water resource development and anthropogenic additions to loads transported by floodwaters.

**Harvest**

Longbums are an abundant and easily accessible food source (Buckworth, 1995). Over historical time, species of *Telescopium* and *Terebralia* have been eaten by coastal peoples throughout their range (Beech and Kallweit, 2001; Bourke, 2015; Feulner, 2002). Throughout the Northern Territory, longbums and smaller mudwhelks (other *Terebralia* sp.) have been a small but consistent portion (<5%) of the shellfish diet of coastal Indigenous people for at least 2000 years (Bourke, 2015; Buckworth, 1995).

In the Darwin catchments, they continue to be consumed by resident and visiting Indigenous people, who collect them from the mangrove forests within Darwin Harbour and in smaller communities scattered around the mostly remote Northern Territory coastline (Bourke, 2015; Buckworth, 1995; Willan, 2013). However, the degree of exploitation is unknown.

The non-Indigenous population does not exploit longbums, apart from using them at low levels for bait. Little is known of the exploitation of longbums in the estuaries of the Fitzroy and Mitchell Rivers. However, the town of Derby is situated on King Sound and a significant percentage of Indigenous peoples make up the town’s population, and are possible consumers of longbums. The town of Kowanyama, which is in the lower reach of the Mitchell River adjacent to the estuary, has a mostly Indigenous population who are also likely to consume longbums.

The longbum *T. telescopium* is the only species consumed in quantity in the Northern Territory, as the taste of *T. palustris* is less preferred.

**Conceptual model**

Figure 4-30 shows the conceptual model for longbums, with the accompanying narrative given in Conceptual model narrative Table 4-10.
Figure 4-30 Conceptual model showing the relationship between threats, drivers, effects and outcomes for longbums (mudwhelks) in northern Australia.
Conceptual model narrative

Table 4-10 Knowledge underpinning the conceptual model for longbum mudwhelks in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/ STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/ regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow. Timing and amplitude of monsoonal rainfall and flooding important in maintaining euryhaline salinity regime in the mangrove community (estuarine ecotone).</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus, the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. The estuarine ecotone available to longbums is lost. Weak/ out-of-sync monsoon early rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. Baseflow occurs out of sync with the spawning season of longbums (spring/summer; pre-wet and wet). The horizontal extent of brackish waters decreases or is lost to the estuary. High or artificially elevated baseflow during the spawning season may impact the ability of larvae to survive in freshwater or oligohaline waters. Freshwater would cause the loss of microphytobenthos in estuarine sediments, the food of detritivores.</td>
<td>The reduction of brackish water habitat reduces the ability of longbums population to forage and grow in the estuary. Longbum populations decline. Strong flows during spawning and larval hatching may wash larvae from the estuary or cause freshwater conditions not favourable to larval survival. Strong flows creating freshwater conditions would cause the loss of meiofauna, detritus and detritus cycling in the upper substrates; reducing the food availability for longbums.</td>
<td>Loss of microphytobenthos and meiofauna in estuarine sediments during period of freshwater influx. (Burford et al., 2012; Duggan et al., 2014).</td>
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<tr>
<td>Reduction in flood flow inundation/ flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows. Timing of monsoonal rainfall and flooding important in maintaining mangrove habitats and a turbid estuarine ecotone.</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the persistence of the ecotone during the early-to-mid dry season. Optimal habitat (brackish turbid, non-hypersaline waters) for juvenile and adult longbums is lost. Reduction in depositional environment for the maintenance and expansion of mangrove habitats; key habitats for longbums in northern Australia (Asbridge et al., 2016).</td>
<td>Reduction in brackish turbid water habitat limits the ability of longbums to forage and grow in the estuary. Reduction in estuarine turbidity may cause predation on juvenile longbums to High abundance of longbums in the upper-to-mid intertidal zone in mangrove communities in tropical estuaries. Expansion of mangrove community extent both upstream within an estuary and seaward as the deposition delta expands during years of</td>
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<td><strong>1 HABITAT AND THREATS</strong></td>
<td><strong>2 DRIVER/ STRESSOR</strong></td>
<td><strong>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</strong></td>
<td><strong>4. EFFECT</strong></td>
<td><strong>5. OUTCOME</strong></td>
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<td>Reduction in flood flow volume</td>
<td>Maintenance of volume of water leaving the estuary to the plume zone; and duration of peak flows.</td>
<td>Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the floodplume. Reduction in flood flow volume may reduce flushing of estuaries, leading to contaminant build up in sediments (French, 2013). Very high flows impact the viability of benthic phytoplankton and meiofaunal communities in estuarine substrates due to freshwater conditions (Burford et al., 2012; Duggan et al., 2014).</td>
<td>Limitation of primary production in the floodplume and near-shore waters due to nutrient limitation. Export of terrestrial-sourced nutrients through the estuary. Contamination of sediment on which detritivores feed. Possible enhancement of detritus food resources for longbums if very large floods are reduced.</td>
<td>high rainfall and riverflows (Asbridge et al., 2016). Longbums forage and shelter among mangrove trees in the intertidal zone in northern Queensland and the Darwin catchments (Buckworth, 1995; Lasiak and Dye, 1986).</td>
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<td></td>
<td>Change in inter-annual sequencing</td>
<td>Inter-annual enhancement and flushing of estuarine habitats during high flood flows.</td>
<td>Longbums inhabit brackish and marine-salinity littoral habitats year-round benefit from seasonal-productive pulses and flushing habitat. Loss of annual sediment deposition with mangrove habitats that maintain their integrity.</td>
<td>Loss of seasonal productivity stimuli and mangrove habitat maintenance; productive habitats in which longbums thrive.</td>
<td>Maintenance of mangrove community extent supported by annual deposition of estuarine sediments (Asbridge et al., 2016).</td>
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<td>1 HABITAT AND THREATS</td>
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<td>Saltwater intrusion</td>
<td>Expansion of saline habitats in the upper littoral and supra-littoral zone on the outer margins of estuaries.</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion may flood and expand upper littoral habitats for longbum colonisation. In contrast, saline intrusion in the main estuary reduces the extent or presence of a turbid, brackish ecotone which may support longbum populations.</td>
<td>Contrasting effects of expanded upper tidal habitats on the outer margins of estuaries, offset by a reduction in the extent of optimal salinity in estuarine waters. The micro-habitat use of longbums is poorly understood, consequently the resultant impact requires investigation.</td>
<td>Synthesis of literature</td>
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<tr>
<td>Estuarine/saltpan wetland and floodplain</td>
<td>Longbums do not use salt flat habitats, however salt flats contribute up to 13% additional primary production to estuaries during inundation.</td>
<td>Annual flooding of salt flats and mark an inundation stimulus of productivity from these habitats. Desiccated algal crusts on saltpans invigorates and begins to photosynthesise. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016).</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation saltpan adjacent to tropical estuaries reduces key primary productivity that becomes available to estuarine fauna during the monsoon season.</td>
<td>Reduction in nutrients transferred within and exported from tropical estuaries with subsequent reduced coastal productivity for resident species (Burford et al., 2016; Burford et al., 2012).</td>
<td>Reduction in nutrient production from flooded saltpans during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients transferred within tropical estuaries with subsequent reduced coastal productivity and flow-on impacts for intertidal estuarine species (Burford et al., 2012).</td>
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<tr>
<td>Change in inundation frequency, depth and duration Change in land use and management</td>
<td>Truncation of an estuary due to the placement of barriers in a reach of the estuary. Either low or nil baseflow causing the fluvial separation of riverine reaches and the estuary.</td>
<td>Loss of aerial extent of estuary due to truncation. Growth and survival likely is optimal in brackish to non-hypersaline habitats supported by baseflow entering the estuary year-round or seasonally.</td>
<td>Physical barriers truncate the estuary and remove a considerable extent of the available estuarine habitat for longbums. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of turbid and brackish waters decreases or is lost to the estuary. Optimal high turbidity and moderate salinity for longbums is lost, thus critical habitat is lost.</td>
<td>Reduced habitat extent for juvenile and adult longbums. Reduction in population size of longbums.</td>
<td>The optimal salinity for the growth and survival of juvenile and adult longbums has not been explored under laboratory experimental conditions. Field-based observations of their estuarine distribution suggest longbums are found in the mid-to-upper tide zone in brackish to marine...</td>
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<td><strong>Water column habitats all waters</strong></td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation</td>
<td>Loss of water clarity, light penetration. Loss of water column productivity/ algal productivity. Loss of epibenthic productivity.</td>
<td>Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising estuarine food webs, detritus cycling in particular (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of planktonic and benthic food-web within the estuary. Reduction in microphytobenthos and detritus accumulation in estuaries.</td>
<td>Synthesis of literature</td>
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</table>
4.2.5 CROCODILES

Asset description

Two species of crocodile occur in Australia: the saltwater (estuarine) crocodile (*Crocodylus porosus*) and freshwater crocodile (*C. johnstoni*). This review focuses on the saltwater crocodile, given their use of estuarine and coastal waters.

Saltwater crocodiles are found throughout coastal regions of southern India and Sri Lanka, south-east Asia and as far south as northern Australia (Johnson, 1973; Webb et al., 2010). Saltwater crocodiles are a large, long-lived, late-maturing reptile that are an apex predator in tropical Australia (life span can be up to 70 years) (Webb et al., 1984). Males' mean length is approximately 5 m and can reach 6 to 7 m. Weighing 1000 kg, males are sexually mature at length of 3.4 m or 16 years of age. Female mean length is less than males, at about 3 m; they are sexually mature at length 2.3 m or 12 years of age (Webb and Manolis, 1998). Saltwater crocodiles have the capacity to modify the faunal composition of coastal habitats and, within their pristine habitats, are a robust competitor with humans.

The saltwater crocodile populations have recovered strongly across northern Australia in the 40 years since the species was protected (Fukuda et al., 2011). The saltwater crocodile population of the Northern Territory has achieved full recovery from uncontrolled hunting in 1945 to 1971 (Fukuda et al., 2011). In contrast, populations in Queensland and WA have shown limited growth, with low to moderate increases over time (CALM, 2003; Read et al., 2004). The species is listed as vulnerable in Queensland (EPA, 2007), although regional populations across northern Queensland have shown slow recovery relative to the Northern Territory following protection some 40 years ago from unregulated hunting (Fukuda et al., 2011; Fukuda et al., 2008; Read et al., 2004). The Australian population of saltwater crocodiles is estimated to be greater than 75,000 (Isberg et al., 2005).

Distribution in Assessment study areas

Figure 4-31 and Figure 4-32 show the distribution of saltwater crocodiles in two of the Assessment catchments. There were no records for the Fitzroy catchment.
Figure 4-31 Observation of saltwater crocodiles (*Crocodylus porosus*) in the Darwin catchments
An analysis of saltwater crocodile abundance surveyed in river catchments across northern Australia, together with possible environmental drivers, indicated that temperature, extent of favourable freshwater vegetation and rainfall seasonality (as a proxy for flooding and ecosystem productivity) were important determinants of distribution (Fukuda et al., 2008). Favourable vegetated habitats in wetlands, swamps and floodplains showed a particularly strong association with crocodile density. The extent of favourable habitat for saltwater crocodiles in the eastern Gulf of Carpentaria is higher north of the Embley River, where extensive freshwater wetlands with low risk of seasonal flooding are found (Magnusson et al., 1980). In the Northern Territory, both the extent of favourable habitat and the abundance or biomass of saltwater crocodiles (sightings per kg per km surveyed 150 km above the river mouth) are high in rivers such as the Finnis, Adelaide, Mary and Wildman rivers (5.32, 3.41, 5.99 and 7.06 sightings per km, respectively). In Queensland, crocodile density is lower, although they are still present (0.51 for the Mitchell River) (EPA, 2007; Fukuda et al., 2008; Magnusson et al., 1980). Eight habitat regions have been defined for
saltwater crocodiles in Queensland, with the Flinders and Gilbert rivers falling in the Southern Gulf Plains and Northern Gulf Plains, respectively (EPA, 2007). The only significant concentrations of saltwater crocodiles in the Southern Gulf Plains are found in the middle reaches of the Norman River (Fukuda et al., 2008). Surveys (1994 to 2000) of the eight crocodile regions around the northern Queensland coast found that less than 5% of hatchling crocodiles and less than 14% of older crocodiles were observed in the Southern and Northern Gulf plain regions collectively (Fukuda et al., 2008). In WA, it is considered that the total population of saltwater crocodiles is now only of the order of a few thousand non-hatchlings. Nesting habitat is limited, with very few nests located during the surveys conducted to date (CALM, 2003). The population size of saltwater crocodiles in the Fitzroy River is low compared with elsewhere across its range, due to lack of suitable nesting sites near the river mouth (Pusey and Kath, 2015; Semeniuk et al., 2011).

Reproduction commences during the wet season (EPA, 2007; Webb et al., 1977; Webb et al., 1983). Courtship and mating are probably triggered by rainfall and the relatively cooler temperatures associated with the end of the dry season (Isberg et al., 2005; Webb et al., 1977). Breeding and recruitment occur principally in rivers with significant freshwater input or freshwater wetlands (Fukuda and Cuff, 2013; Webb et al., 1977). Eggs (~50) are laid in nests composed of mounds of vegetation above the flood level, either adjacent to rivers or in freshwater wetlands (Thorbjarnarson, 1996; Webb et al., 1983). There are likely to be multiple factors influencing the placing of nests, including the availability of deep freshwater as refuges for the females. Suitable nest sites are more available in freshwater habitats than saline habitats, and freshwater habitats also support suitable ground vegetation for constructing nests (EPA, 2007; Fukuda and Cuff, 2013; Magnusson, 1980; Webb et al., 1977). Nests are rarely positioned 100 m or more from permanent deep water (Webb et al., 1983). The presence of fresh water appears to be important for saltwater crocodile reproduction. However, as nesting occurs during the wet season, egg mortality due to flooding can be high (Magnusson, 1982; Webb et al., 1977; Webb et al., 1983). Females may watch the eggs during the incubation period (70 to 90 days) and have been observed to help hatchlings reach water (Magnusson, 1979; Webb et al., 1977).

Saltwater crocodiles feed on animals along the water’s edge, birds from the water surface, fish and crustaceans (Johnson, 1973; Webb and Messel, 1978). Growth rates of juvenile crocodiles are fastest during the wet season, when prey is abundant and flooded habitats are extensive (Magnusson and Taylor, 1981; Webb and Messel, 1978). Thus, the abundance of prey is crucial to crocodile population stability. Many prey species, such as fish, birds and crustaceans, are cued by the monsoonal stimuli characteristics of the wet–dry tropics. Key examples of these relationships and how they are affected by flood flows are described elsewhere in this report.

**Commercial use**

The wild population of crocodiles is not harvested as such, but crocodile eggs are harvested under licence to support the farmed harvest of crocodiles. Under licensed harvest, a proportion of the eggs are harvested from the wild-laid stock, and then hatched and grown
under farm conditions. Farmed crocodiles are harvested at a size about 1.5 m and both the skin and the meat are used. In the Northern Territory, crocodile farming turned over $25 million from 2014 to 2015. Crocodile farming has expanded year-on-year from 2002 to 2011 (CFANT, 2014).

Conceptual model

Figure 4-33 shows the conceptual model for crocodiles, with the accompanying narrative given in Conceptual model narrative

Table 4-11.

Figure 4-33 Conceptual model showing the relationship between threats, drivers, effects and outcomes for crocodiles in northern Australia
### Conceptual model narrative

#### Table 4-11 Knowledge underpinning the conceptual model for saltwater crocodiles in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/ STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Timing and amplitude of monsoonal rainfall and post-wet-season flows is important in maintaining saltwater crocodiles access between estuarine feeding habitats and nesting/early juvenile habitats</td>
<td>Adult saltwater crocodiles may experience reduced access between feeding habitats in estuaries and nesting habitats in wetlands (particularly for male crocodiles during the breeding season). Reduced habitat extent for optimal estuarine condition. Early juvenile habitats may lose connectivity to estuaries allowing ontogenetic habitat shifts. Estuarine ecotone supports juvenile crocodile prey species such as fish and crustaceans.</td>
<td>Slow juvenile crocodile growth rate due to low prey abundance. And less nests constructed and less embryos killed. Reduced nesting success for crocodiles during breeding season (Isberg et al., 2005)</td>
<td>Hatchlings emerging early would be able to take advantage of the higher growth rates possible during the wet season and hence outgrow their predators (Magnusson and Taylor, 1981). Saltwater crocodiles nesting coincided with the annual wet season. Flooding killed all embryos in 24 of 30 nests with eggs under study (Magnusson, 1982; Webb et al., 1977). Low water levels between August and October, combined with high temperature conditions, are generally associated with a reduced nesting effort. In contrast, high water levels and cool conditions result in maximal nesting activity (Isberg et al., 2005).</td>
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<td>Reduction in flood flow inundation/ flooding regime</td>
<td>Maintenance of timing, amplitude, volume and duration of flood peaks</td>
<td>Timing of monsoonal rainfall and flooding important in maintaining estuarine ecotone</td>
<td>Early wet-season flows may be captured by water extraction, diversion or impoundment. Seasonal floods inundate salt flats that contribute to the primary productivity of the estuarine system. Low to medium floods likely have a positive effect on juvenile crocodile abundance; high floods may reduce juvenile reptile abundance. Poor floods may limit crocodile’s access to inundated wetland habitats suitable for nesting and where prey species are abundant.</td>
<td>Early wet-season floods stimulate crocodile breeding cycle. Moderate floods enhance estuarine production and monsoon habitats, contributing to crocodile abundance. In addition, moderate floods enhance the abundance of crocodile prey and thus crocodile growth and survival. The crocodilian population increases in years following moderate floods. In contrast, high floods may negatively impact estuarine</td>
<td>Higher than mean rainfall (flooding or higher water level) at the end of dry season and coupled with low temperature will trigger mating and courtship (Isberg et al., 2005). Growth rates of juvenile crocodiles are fastest during the wet season when prey is abundant and flooded habitats are extensive (Magnusson and Taylor, 1981; Webb and Messel, 1978).</td>
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<td><strong>1 HABITAT AND THREATS</strong></td>
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<td>Reduction in flood flow volume</td>
<td>Maintenance of volume and duration of peak flows</td>
<td>Estuarine habitats cannot sustain optimal environment conditions for saltwater crocodile’s physical condition for mating and egg-laying opportunities.</td>
<td>A short flood season would negatively impact the crocodiles breeding cycle, therefore reducing reproductive success with flow-on effects on the crocodile population. Both egg incubation and juvenile survival would be impeded; as well as a reduction in prey abundance in estuarine and adjacent wetland habitats.</td>
<td>Higher than mean rainfall (flooding or higher water level) at the end of dry season and coupled with low temperature will trigger mating and courtship (Isberg et al., 2005). A truncated flood profile would reduce the seasonal monsoon-induced ecotone with estuaries.</td>
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<td>Change in inter-annual sequencing</td>
<td>Inter-annual connectivity of floodplain habitats to riverine and estuarine habitats</td>
<td>Juvenile crocodiles benefit from access to floodplain and supra-littoral habitats in moderate flood years. Several high flood years in succession allow access to floodplain habitat, maintain life during dry seasons (billabong permanence); re-connectivity and emigration from those habitats to-sub-adult estuarine habitat in subsequent year or years.</td>
<td>Regular moderate floods enhance the annual recruitment success of young-of-the-year crocodiles, increasing the population and the extent of habitat available to the population. Regular floods increase the extent and productivity of estuarine habitats for both crocodilians and their prey; enhancing crocodile growth rate and survival. Extended series of dry years and low flows ensure non-optimal breeding environment over sequential years; thus</td>
<td>Breeding and recruitment take place principally in rivers with significant freshwater input or in freshwater swamps. As saltwater crocodiles grow they encounter larger territorial animals, and many sub-adult crocodiles appear to be excluded from the breeding areas and are forced to occupy marginal habitats such as higher salinity rivers. Mortality among these intermediate-sized crocodiles also appears to be very high (Ross, 1998).</td>
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<td>Estuarine saltpan/wetland and riverine wetland and floodplain habitats</td>
<td>Saltwater intrusion</td>
<td>Sea-level rise or erosion of coastal habitat-barriers may facilitate the intrusion of saline water into estuarine supra-littoral habitats</td>
<td>The salinity of waters peripheral to the estuary may not be suitable for juvenile crocodiles.</td>
<td>The presence of freshwater appears to be important for saltwater crocodile reproduction (Fukuda and Cuff, 2013; Webb et al., 1977).</td>
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<td>Change in inundation frequency, depth and duration</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Timing and amplitude of monsoonal rainfall and post-wet-season flows is important in maintaining wetland integrity and connectivity between freshwater and saltwater habitats</td>
<td>Adult saltwater crocodiles may experience reduced access between feeding habitats in estuaries and nesting habitats in wetlands (particularly for male crocodiles during the breeding season). Reduced habitat extent for optimal wetland condition. Early juvenile habitats may lose connectivity to estuaries allowing ontogenetic habitat shifts. Freshwater wetlands support juvenile crocodile and their prey species such as fish and crustaceans.</td>
<td>Reduced breeding success for crocodiles during breeding season. Flooding is the major cause of mortality. Nests beside rivers are more likely to be flooded than nests in swamps. Low water levels between August and October, combined with high temperature conditions, are generally associated with a reduced nesting effort. In contrast, high water levels and cool conditions result in maximal nesting activity (Isberg et al., 2005). Nests are rarely sited 100m or more from permanent deep water (Webb et al., 1983).</td>
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<td>Change in land use and management</td>
<td>Reduction in flood flow inundation/flooding regime</td>
<td>Floodflows ensure the maintenance of freshwater supply to supra-littoral wetland habitats during and after the wet-season annual flooding of salt flats and wetlands mark an inundation expansion of habitat available to crocodiles, Early wet-season flows may be captured by water extraction, diversion or impoundment. Seasonal floods inundate freshwater wetlands that are suitable as nesting habitat, increasing its extent. High floods may inundate and drown incubating nests. Low to medium floods likely have a positive effect on crocodile egg survival and juvenile reptile growth. Early wet-season floods stimulate crocodile breeding cycle. Moderate floods enhance egg production and survival. As well, they enhance the abundance of crocodile prey and thus crocodile growth and survival. The crocodylian population increases in years following moderate floods. Higher than mean rainfall (flooding or higher water level) at the end of dry season and coupled with low temperature will trigger mating and courtship (Isberg et al., 2005). Nests are rarely sited 100m or more from permanent deep water (Webb et al., 1983). Higher annual egg mortality is caused by rainfall (flooding) washing away nest and exposing eggs (Magnusson, 1982; Webb et al., 1977). Growth rates of juvenile crocodiles are fastest during the wet season when prey is abundant.</td>
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<td>especially for nesting habitat</td>
<td>abundance; high floods may cause high egg mortality and reduce juvenile reptile abundance. Poor floods may limit crocodile’s access to inundated wetland habitats suitable for nesting and where prey species of juvenile reptiles are more abundant.</td>
<td>In contrast, high floods negatively impact the nesting cycle and cause mortality of eggs. The year’s juvenile recruitment is lost. Very high floods with extensive fast-flowing water may limit the habitat of aquatic and marine species and reduce the growth rate of current inhabitants. Flooding is the major cause of mortality. Nests beside rivers are more likely to be flooded or washed away than nests in swamps.</td>
<td>and flooded habitats are extensive (Magnusson and Taylor, 1981; Webb and Messel, 1978). During poor flow years, dry/hot conditions for egg incubation cause mortality prior to hatching. Dry wetlands are poor habitat for early juveniles.</td>
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<td>Reduction in flood flow volume</td>
<td>The maintenance of facultative wetland flood levels fails. Wetland habitats cannot sustain optimal environment conditions for saltwater crocodile’s mating, egg-laying and incubation opportunities</td>
<td>A short flood season would negatively impact the crocodiles breeding cycle, therefore reducing reproductive success with flow-on effects on the crocodile population. Both egg and juvenile survival would be impeded; as well as a reduction in prey abundance in wetland habitats.</td>
<td>A truncated flood profile would reduce the availability and suitability of nesting habitat and the incubation of eggs.</td>
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<td>Juvenile crocodiles benefit from access to floodplain and supra-littoral habitats in moderate flood years. Several high flood years in succession allow access to floodplain habitat, maintain life during dry seasons (billabong permanence); re-connectivity and emigration from those habitats to sub-adult estuarine</td>
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<td>Saltwater intrusion</td>
<td>Sea-level rise or erosion of coastal habitat barriers may facilitate the intrusion of saline water into wetland habitats</td>
<td>Saltwater intrusion into freshwater wetlands may reduce the extent of optimal habitats for crocodile nesting and incubation. Loss of freshwater wetland habitat for crocodile prey species</td>
<td>Habitat in subsequent year or years.</td>
<td>Prey; enhancing crocodile growth rate and survival. Extended series of dry years and low flows ensure non-optimal breeding environment over sequential years; thus reducing recruitment to the crocodile population.</td>
<td>The presence of freshwater appears to be important for saltwater crocodile reproduction (Fukuda and Cuff, 2013; Webb et al., 1977).</td>
</tr>
<tr>
<td>Longstream habitat connectivity</td>
<td>Loss of connectivity between breeding and juvenile habitats (marine, estuarine and riverine reaches of rivers) due to the placement of barriers in the upper estuary or lower river reach; or to either low or nil flow causing the fluvial separation of waterholes above the estuary</td>
<td>Loss of connectivity between breeding and juvenile habitats may reduce the extent of optimal habitats for crocodile nesting and recruitment.</td>
<td>Breeding and recruitment take place principally in rivers with significant freshwater input or in freshwater swamps. As saltwater crocodiles grow they encounter larger territorial animals, and many sub-adult crocodiles appear to be excluded from the breeding areas and are forced to occupy marginal habitats such as higher salinity rivers. Mortality among these intermediate-sized crocodiles also appears to be very high (Ross, 1998).</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Loss of migration pathways between estuarine and riverine habitats</td>
<td>Physical barriers block or impede upstream migration of juveniles and subsequent emigration as adults downstream to the estuary. Barriers in the lower river reaches may impede movement to key freshwater nesting habitats. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary.</td>
<td>Reduced access to critical habitat for juvenile and sub-adult crocodiles. Reduced access to optimal crocodile nesting habitats. Increased egg survival rate Reduced longstream habitat extent. Reduced optimal brackish water habitat for crocodilian prey species; thus reducing growth and survival of juvenile crocodiles.</td>
<td></td>
<td></td>
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</tbody>
</table>

Breeding and recruitment take place principally in rivers with significant freshwater input or in freshwater swamps. As saltwater crocodiles grow they encounter larger territorial animals, and many sub-adult crocodiles appear to be excluded from the breeding areas and are forced to occupy marginal habitats such as higher salinity rivers. Mortality among these intermediate-sized crocodiles also appears to be very high (Ross, 1998).
<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/ STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riverine floodplain wetlands</td>
<td>Reduced access to waterhole/wetland habitats on floodplains</td>
<td>Juvenile habitat during flood events and monsoon season</td>
<td>Reduction in access to habitat and seasonal resources required to maintain robust population.</td>
<td>Reduction in population size or changes in distribution/habitat use.</td>
<td>Currently, GOC and NT coasts remain relatively pristine. In Northern Territory, feral animals such as the Buffalo destroy wetland habitat and reduce vegetation (Grigg and Kirshner, 2015; Webb et al., 1984; Webb et al., 1987) (<a href="http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1774">http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1774</a>)</td>
</tr>
<tr>
<td>Riverine floodplain wetlands</td>
<td>Anthropogenic modification of habitat</td>
<td>Habitat degradation/ homogenisation</td>
<td>Tropical habitats of crocodilian populations are largely pristine; however, water resource development and economic development in coastal landforms may change the ecosystem services provided by tropical, coastal habitats. Loss of habitat due to degraded water quality due to impacts on wetland and reducing vegetation from Buffalo.</td>
<td>Sedimentation impacts can be positive and negative impacts on crocodile populations. Significant wetland sedimentation would reduce productivity of the habitats of the prey species of juvenile crocodiles. Estuarine deposition may enhance the extent of crocodile habitat. Water pollution may directly or indirectly affect the species.</td>
<td>Currently, there is no evidence that such pollution is having any effects on contemporary crocodile populations (Fukuda et al., 2008). However, the tropical habitats of crocodilian populations are largely pristine; water resource development and economic development in coastal landforms may change the nature of tropical, coastal ecosystems.</td>
</tr>
<tr>
<td>Water column habitats all waters</td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation</td>
<td>Loss of water clarity, light penetration in estuaries and wetlands</td>
<td>Loss of food-chain integrity and prey abundance within estuaries and wetlands. High-flow sediment deposition may destroy wetland habitats. The productivity of wetlands and floodplains may also be reduced by pollution (Cosser, 1997) and changes in hydrographical regime in catchments (Kingsford, 2000). Mangrove habitats expand in the estuarine delta.</td>
<td>Sedimentation impacts can be positive and negative impacts on crocodile populations. Significant wetland sedimentation would reduce productivity of the habitats of the prey species of juvenile crocodiles. Estuarine deposition may enhance the extent of crocodile habitat. Water pollution may directly or indirectly affect the species.</td>
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</tr>
<tr>
<td>Reduced access due to less frequent and lower flood events</td>
<td>Conflicting land use</td>
<td>Habitat degradation</td>
<td>Loss of habitat extent; loss of natural condition of wetland habitats.</td>
<td>Loss of habitat extent due to transformation of coastal habitat from natural to developed landforms. Interactions between crocodiles and humans may increase and cause the removal of crocodiles from perimeter habitats. Feral animal negative impact on wetland habitat and vegetation.</td>
<td>Currently, GOC and NT coasts remain relatively pristine. In Northern Territory, feral animals such as the Buffalo destroy wetland habitat and reduce vegetation (Grigg and Kirshner, 2015; Webb et al., 1984; Webb et al., 1987) (<a href="http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1774">http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1774</a>)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Competition for resources and degradation of habitat</td>
<td>Tropical habitats of crocodilian populations are largely pristine; however, water resource development and economic development in coastal landforms may change the ecosystem services provided by tropical, coastal habitats. Loss of habitat due to degraded water quality due to impacts on wetland and reducing vegetation from Buffalo.</td>
<td>Sedimentation impacts can be positive and negative impacts on crocodile populations. Significant wetland sedimentation would reduce productivity of the habitats of the prey species of juvenile crocodiles. Estuarine deposition may enhance the extent of crocodile habitat. Water pollution may directly or indirectly affect the species.</td>
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</table>
4.2.6 SNUBFIN DOLPHIN

Asset description

The Australian snubfin dolphin (*Orcaella heinsohni*) is a coastal mammal that inhabits tropical estuarine and close-coastal habitats from Exmouth Gulf in WA to Brisbane in Queensland (Parra et al., 2006; Palmer, 2011; Palmer et al., 2014; Parra, 2006).

The snubfin dolphin is a distinct species, co-generic with the south-east Asian Irrawaddy dolphin (*O. brevirostris*) (Beasley et al., 2005; Palmer et al., 2011). Until the early 2000s, all Asian and Australian snubfin dolphin were considered to be the Irrawaddy dolphin. Beasley et al. (2005) published a morphological and genetic study demonstrating that the Australian snubfin dolphin was a distinct species (*O. heinsohni*). Further genetic studies confirmed the two distinct species (Palmer et al., 2011) and since the late 2000s published papers refer to the Australian snubfin dolphin. A similar species differentiation occurred for the Australian humpback dolphin (Frere et al., 2011).

The rivers of tropical Australia are relatively pristine, especially those of the Gulf of Carpentaria, Arnhem Land, and the Kimberley region. On a worldwide scale, the rivers in these regions are the some of the last undisturbed refugia for large-bodied, coastal mammals, such as dolphins (Halpern et al., 2008; Palmer et al., 2014). They are abundant in estuaries and inshore coastal waters of Cape York (especially Princess Charlotte Bay), the Top End (including rivers and estuaries within the Darwin catchment study area), and the Kimberley (including King Sound) (Brooks and Pollock, 2013; Brooks and Pollock, 2015a; Brooks and Pollock, 2015b; Brown et al., 2016; Palmer et al., 2011; Parra et al., 2002). The remoteness of these regions leads to low reporting of the fine-scale distribution of the species. Few records of snubfin dolphin are present in the Gulf of Carpentaria, and four only in the eastern Gulf (Karumba, and the estuaries of the Gilbert, Mitchell and Wenlock rivers (ALA, 2017). The observations in the Gilbert and Mitchell rivers were made by the author and colleagues during biological survey work in the Mitchell, Gilbert and Flinders rivers (Gulf of Carpentaria) in November 2016. Snubfin dolphin have been recorded in the receiving waters of the Fitzroy River (Brown et al., 2016) and inhabit the river’s estuary in the vicinity of Derby and Doctors Creek (~17.28 °S, 123.58 °E).

In Australia, dolphins are poorly studied across all of the tropics. There is a dearth of studies of dolphin populations in the Northern Territory and northern WA, and few studies of coastal dolphins – snubfin dolphin and the Australian humpback dolphin (*Sousa sahulensis*) – anywhere in Australia (Palmer et al., 2014). A recent study of three dolphin species in five embayments in the Kimberley region of WA has provided key information on snubfin dolphin distribution and habitat use in Australia’s north-west (Brown et al., 2016). Palmer et al. (2014) suggest that the lack of data on these less common species (both geographically and the few single-location studies only) – and the lack of data specific to the remote tropics for (otherwise) well-studied dolphin species – limits the ability to assess and manage their community wellbeing.

Studies to date show that snubfin dolphin use near-shore habitats, such as embayments and estuaries, which are protected from strong oceanic and weather influences, as critical habitat. They are also dependent on pristine tropical rivers with abundant prey. Modification of river flows, fluvial loads and sedimentation regimes will affect estuarine systems and inshore waters, which are the prime habitats of snubfin dolphin and their prey.
Snubfin dolphins are highly social animals that associate in small groups and pairs. They are likely to have small ranges and emigration from specific locations is not common. Their diet of small fish species and cephalopods are critical components of their niche in the tropical inshore habitats. As a consequence, the pristine condition of receiving waters and inshore waters are critical to the survival of snubfin dolphin. They are a top predator in estuarine ecosystems, and adults are only likely to be hunted by sharks and crocodiles.

**Distribution in Assessment study areas**

Figure 4-34, Figure 4-35 and Figure 4-36 show the recorded sightings of the snubfin dolphin in the Assessment catchments and marine study areas.
Figure 4-35 Distribution of snubfin dolphin (*Orcaella heinsohni*) in the Darwin catchments and marine study area
**Habitat use**

Snubfin dolphin occur in coastal and estuarine waters up to 50 km upstream (Allen et al., 2012; Palmer, 2011; Parra et al., 2002). In the estuaries, they are sympatric with the Australian humpback dolphin, though in the Northern Territory rivers, humpback dolphin were observed only 30 km upstream. Within rivers in Kakadu National Park, adult snubfin dolphin were observed with new-borns and juveniles (Palmer, 2011), suggesting that these Arnhemland rivers may be critical parenting and nursery habitat for the species. Their significant presence in estuaries and upper estuaries is a defining feature of snubfin dolphin and represents a distribution further upstream than usual for other coastal dolphin species. Elsewhere in the Northern Territory and in the Kimberley region of WA snubfin dolphin were observed in:

- large, multi-fluvial estuaries (upper Cambridge Gulf)
- near-enclosed marine embayment habitats (Port Essington, Coburg Peninsula; King Sound, WA)
Embellishments are critical habitat for snubfin dolphins. In Cygnet and Roebuck Bays, Dampier Peninsula, they were common and their groups included calves and juvenile dolphins (Brown et al., 2016). Snubfin were less common in inshore coastal habitats with marine orientation (Allen et al., 2012; Brown et al., 2016). Few dolphin groups were observed in the estuarine habitat in Cambridge Gulf, though much less sample effort was deployed there. In the Northern Territory studies (Darwin environs and Port Essington), no mention is made of juvenile snubfin when describing dolphin distribution. It may be the case that snubfin dolphin calve in the mid-to-upper estuary of large rivers or sheltered bays in the study area (see Palmer, 2011) and move to marine habitats as large juveniles and adults. To date, no micro-habitat studies have been undertaken of snubfin dolphin to determine their micro-habitat use or behaviour during the reproductive phase of their life cycle. However, reports of their distribution suggest that close-inshore areas within embayments and estuaries are key habitats for juvenile snubfin dolphins and their mothers.

In Princess Charlotte Bay and in Cleveland Bay adjacent to Magnetic Island (Queensland), snubfin dolphin occupy similar inshore habitats as in the Northern Territory (Parra et al., 2006; Parra, 2006). As documented for the Northern Territory, they share the waters with the Australian humpback dolphin. In Cleveland Bay, snubfin dolphin preferred shallow waters 1 to 2 m deep and were often found over seagrass habitats, while humpback dolphin preferred waters 2 to 5 m deep and hunted in channel habitats (Parra, 2006). Both species were found in the vicinity of river mouths. In Princess Charlotte Bay, each species showed similar micro-habitat selection as they did in Cleveland Bay. Snubfin dolphins were found in shallower water (mean ~8 m) than humpback dolphins (mean ~12 m). The majority of sightings of both species occurred within 5 km of land and snubfin dolphins generally were found closer to river mouths than were humpback dolphins (Parra et al., 2006).

Snubfin dolphin are abundant in embayments within the King Sound complex in WA, which are the receiving waters of the Fitzroy River. A population of about 50 individuals resides in Cygnet Bay, north-west King Sound (east Dampier Peninsula). A population resides in Cone Bay, north-east King Sound and further study is required to determine its size. No published information is available on the use of the Fitzroy River estuary by snubfin dolphin. In King Sound, snubfin dolphin co-exist with Australian humpback and bottlenose dolphin (Tursiops aduncus). Elsewhere on the Dampier Peninsula, populations of snubfin dolphin are found in embayments close by (~200 km), but connectivity between each local group seems to be negligible (Brown et al., 2016; Brown et al., 2014).

Dolphin surveys have not been undertaken systematically throughout all of the estuaries and nearshore areas of the Darwin rivers. However, the INPEX Ichthys Gas Field Development project has supported studies of a range of biota in Darwin Harbour as part of monitoring during construction of the Ichthys liquefied natural gas facility at East Arm in the harbour (Brooks and Pollock, 2013; 2015a; Brooks and Pollock, 2015b). Dolphin abundance and movement within and between Bynoe Harbour, Darwin Harbour and Shoal Bay were measured. Australian snubfin, Australian humpback and bottlenose dolphins were studied. Within the three embayments, the population of snubfin
dolphins increased from a maximum of 32 observed per sampling early in the study, to 45 to 70 after three years. The population of humpback dolphin varied around 90 individuals throughout the 3.5 years of the surveys (eight sampling surveys). Between 20 and 30 bottlenose dolphins were recorded at each sampling in the Assessment area over the 3.5 years. There was a trend for a reduced number of humpback dolphins in Darwin Harbour during pile driving operations (Brooks and Pollock, 2015b). The study showed significant emigration of individuals from the study area. It is likely that snubfin dolphins inhabit poorly surveyed estuaries within rivers such as the Wildman, Mary and Adelaide to the west of the Kakadu region. Moreover, it is possible that a restricted range of newborn and juvenile snubfin and humpback dolphin in western Arnhemland estuaries may represent refugia for key life-history stages of these near-threatened species in Australia. Snubfin dolphin have been observed in the Gilbert and Mitchell River estuaries on the west coast of the Cape York Peninsula.

**Population biology and behaviour**

Few studies of the behaviour and feeding or prey items of snubfin dolphins have been undertaken in tropical Australia. Parra and Jedensjo (2009) examined the gut contents of snubfin dolphin on the Queensland coast and found cardinal fish, cuttlefish, squid and the toothpony fish to be the numerically dominant prey. Cardinal and toothpony fish are common in estuaries and nearby bays (~10 to 35 m depth) in the Gulf of Carpentaria, including within mangrove and seagrass habitats in the lower estuary, and seagrass habitats in coastal embayments (Blaber et al., 1989; Blaber et al., 1990; Blaber et al., 1992). They inhabit both brackish and marine waters (Blaber et al., 1989). Other common fish prey species such as grunts (*Pomadasys* sp.) are common in estuaries and exhibit seasonal preference for mid or upper reaches of estuaries (Blaber et al., 1989). Similarly, cuttlefish and squid occur throughout the Gulf of Carpentaria, including the embayments and estuaries (Dunning et al., 1994; Kenyon et al., 2015). Recent field observations show that snubfin feed on bait fish at the confluence of creeks with the main river channel in the Prince Regent estuary in the Kimberley.

Being mammals, juvenile dolphin rely on parental care and parental suckling for approximately 2 years, perhaps longer (Ross, 2006). They reproduce only every 2 to 3 years (Palmer et al., 2011). Consequently, they co-locate in habitats with their mothers. The absence of juvenile dolphin from Northern Territory coastal waters suggests that mothers might limit their range during lactation, remaining close to their offspring in estuaries. Both the Darwin area and Port Essington studies documented emigration from, and immigration to, the study areas by snubfin dolphins. This suggests that despite having stable populations of known individuals, at times dolphins move significant distances. The reproductive behaviour of well-studied dolphin species (e.g. bottlenose dolphin in subtropical and temperate regions) is not relevant to these tropical dolphin species. In Cygnet Bay, King Sound, multiple sightings of individuals suggest high survival (0.95) and residency of snubfin dolphin: a similar trend as in Roebuck Bay, adjacent to Broome. In addition, an individual’s association with other snubfin dolphins is non-random (particularly for males), suggesting ongoing associations between individuals (Brown et al., 2016). Populations of snubfin dolphin are geographically and likely genetically isolated (Cagnazzi, 2011; Cagnazzi et al., 2013): for example, in Keppel Bay on the Capricorn Coast of Queensland. Reinforcing this observation, (Brown et al., 2016) found site fidelity and population isolation over relatively small geographic distances (~200 km) in the Kimberley region, north-west Australia.
**Conservation status**

Australian snubfin dolphin are a protected species and are considered near-threatened (IUCN, ; http://www.iucnredlist.org/details/136315/0; accessed 23 June 2017). They are considered vulnerable in Queensland under the *Nature Conservation Act 1992*. The conservation status, threats and current status of snubfin dolphin in Australian waters has been documented by the Commonwealth’s Department of Environment and Heritage (Ross, 2006). There are no significant records of their use by Indigenous peoples (although dugong are hunted and eaten); however, anecdotal reports of their opportunistic consumption do exist (Ross, 2006). The pristine coastal ecosystem of tropical Australia represents a refuge where snubfin dolphin have remained abundant and continue to exist and associate as they have for centuries. Their stable population structure contrasts with the co-generic Irrawaddy dolphin elsewhere in the Asia–Pacific, which are critically endangered and suffer high mortality. The extent of their habitat is much reduced and continues to be threatened (Baird and Beasley, 2005; Beasley, 2007).

**Conceptual model**

Figure 4-37 shows the conceptual model for snubfin dolphins, with the accompanying narrative given in Table 4-12.

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Figure 4-37 Conceptual model illustrating the relationship between threats, drivers, effects and outcomes for snubfin dolphin in northern Australia
### Conceptual model narrative

#### Table 4-12 Knowledge underpinning the conceptual model for snubfin dolphin in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/ STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
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</tr>
</thead>
<tbody>
<tr>
<td>Estuarine habitats including habitats of snubfin dolphin prey (i.e. mangrove, seagrass and mudbank habitats)</td>
<td>Extraction/ regulation and Climate change</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow Timing and amplitude of monsoonal rainfall and flooding important in maintaining habitat for prey (prey abundance) for lactating mothers and micro-habitat for nursing juvenile dolphin in estuaries (estuarine ecotone)</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus, the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal salinity for dolphin prey and prey habitat may be lost. Estuaries with a brackish ecotone may be critical habitat for juvenile snubfin dolphin as they are observed in estuaries and close-inshore habitats (Brown et al., 2016; Palmer, 2011). Weak/ out-of-sync monsoon low rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. The horizontal extent of brackish waters decreases or is lost to the estuary. High or artificially elevated baseflow may cause estuaries to become fresh or near fresh with subsequent loss of habitat for dolphin prey.</td>
<td>The reduction of brackish water habitat reduces the linear (and aerial) extent of habitat for dolphin prey and thus prey abundance for snubfin mothers in the estuary. The health and ability of mothers to lactate may decline, with a subsequent decline in snubfin populations. Elevated freshwater flows due to artificial releases also may deplete prey (fish and cephalopod) habitats by making estuarine waters fresh and thus impede snubfin hunting and foraging in habitats within the estuary. Estuarine ecotones may not be obligate for nursing snubfin per se; but protected waters with low wave and energy disturbance may be critical. Thus, prey distribution that allows snubfin to occupy sheltered waters such as estuaries is critical.</td>
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<tr>
<td>Reduction in flood flow inundation/ flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows Timing of monsoonal rainfall and flooding important in maintaining habitats for snubfin prey (estuarine ecotone)</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the persistence of the ecotone during ‘autumn’ and early winter. Optimal salinity for snubfin prey and prey habitat may be lost. Loss of large flood flows that reduce salinity in estuaries. Natural</td>
<td>Reduction in brackish water habitat for snubfin prey limits the ability of snubfin mothers to forage successfully and lactate in the estuary. Snubfin populations migrate or decline. Reduction in emigration cue for juvenile/adult species that have</td>
<td>Positive correlation between fish and crustacean abundance in coastal waters and high annual flows along the tropical and subtropical coast (Gilson et al., 2009; Loneragan and Bunn, 1999; Robins et al., 2005).</td>
<td></td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
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<tr>
<td>Estuarine/saltpan wetland and floodplain</td>
<td>Reduction in flood flow volume</td>
<td>Maintenance of volume and duration of peak flows</td>
<td>Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the floodplume.</td>
<td>Limitation of primary production in the floodplume and near-shore waters due to nutrient limitation.</td>
<td>(Burford et al., 2012).</td>
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<td></td>
<td>Change in inter-annual sequencing</td>
<td>Inter-annual connectivity of floodplain habitats to riverine and estuarine habitats</td>
<td>Snubfin dolphin prey access floodplain habitats or longitudinal habitats and benefit from access to extensive and productive habitat. In subsequent years, they must return to the estuary and marine habitats as adults and to spawn.</td>
<td>Loss of annual connectivity of floodplain, upstream and estuarine habitats causes mortality of juvenile and sub-adult catadromous fish and other prey species in refuge and diminishing pondage.</td>
<td>Expert opinion.</td>
</tr>
<tr>
<td></td>
<td>Saltwater intrusion</td>
<td>Estuary become saline or subject to a tidal saline wedge due to marine intrusion</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced river flows resulting in increased saltwater intrusion. Saltwater intrusion may reduce habitat for snubfin prey and subsequent prey abundance.</td>
<td>Reduction in brackish water habitat for snubfin prey limits the ability of snubfin mothers to forage successfully and lactate in the estuary. Snubfin populations migrate or decline.</td>
<td>Expert opinion.</td>
</tr>
<tr>
<td>Loss of habitat extent Habitat degradation/ homogenisation</td>
<td>Loss of habitat extent Habitat degradation/ homogenisation</td>
<td>Annual flooding of salt flats and wetlands creates a pulse of nutrients from primary production. Desiccated algal crust on saltpants invigorates and begins to photosynthesise. Nitrogen and phosphate production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016).</td>
<td>Disruption to the frequency and depth (extent) of large areas of very low elevation saltpan and wetland adjacent to tropical estuaries reduces primary production from key habitat during the monsoon season.</td>
<td>Reduction in nutrients exported to tropical estuaries with subsequent reduced coastal productivity for resident and emigrant species (Burford, 2011).</td>
<td>Reduction in nutrients exported from tropical estuaries with likely subsequent reduced coastal productivity for species with estuarine and near-shore habitats (Burford, 2011).</td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
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<tr>
<td><strong>Water column habitats all waters</strong></td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation</td>
<td>Loss of water clarity, light penetration Loss of water column productivity/ algal productivity Loss of epibenthic productivity</td>
<td>Riverine and estuarine production is sustained by microalgae, periphyton, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column.</td>
<td>Loss of planktonic and benthic food species for fish and mollusc population which are preferred prey for snubfin dolphin.</td>
<td>High turbidity causes of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2016; Burford et al., 2012; Faggotter et al., 2013; Jardine et al., 2013).</td>
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</table>
4.2.7 THREADFIN

**Asset description**

King threadfin (*Polydactylus macrochir*, previously *P. sheridani*) are large (>1.5 m), non-diadromous, carnivorous fish (order Perciformes) that are endemic to northern Australia, southern Papua New Guinea and Irian Jaya (Motomura et al., 2000). King threadfin range from the Ashburton River/Exmouth Gulf in WA to the Brisbane River in Queensland. They are long-lived (22 years), fast-growing fish that begin life as males and change to females as they age (protandrous hermaphrodite). King threadfin complete their entire life cycle in estuaries and inshore marine waters, can tolerate a wide range of salinities and do not use freshwater habitats (Moore et al., 2012a; Moore et al., 2012b). They are an important ecological species: a top predator in the fish community, consuming a range of fish and crustacean prey in the coastal ecosystem (Blaber et al., 1995; Salini et al., 1990). Their body form and quality of flesh makes them a prized table fish.

King threadfin is often the second-most important target fish species in commercial, inshore gill-net fisheries (after barramundi), as well as being a target species for recreational and Indigenous fisheries throughout wet–dry tropical Australia (Moore et al., 2011). Young fish inhabit estuaries, particularly the mid to upper estuary, while older fish inhabit estuarine and marine systems. They are a voracious predator capable of modifying the estuarine fish and crustacean community in which they live (Salini et al., 1998; Salini et al., 1990). King threadfin typify an estuarine-dependent fish that could be significantly affected by interruptions to the natural river flows maintaining the turbid, brackish ecotone of the rivers in tropical northern Australia.

King threadfin are a fish of cultural significance for the Indigenous community and in key localities in the vicinity of Indigenous townships in the Northern Territory, they are subject to management plans specifying bag limits (see: https://nt.gov.au/__data/assets/pdf_file/0008/204956/fishing-the-daly-river-guide.pdf (2016, accessed 27th June 2017)).

**Distribution in Assessment study areas**

Figure 4-38 shows the distribution of king threadfin across Australia.
Habitat use

King threadfin occur in tropical and subtropical turbid coastal waters, estuaries and mangrove creeks (Blaber et al., 1995). Adults probably spawn in inshore coastal waters and lower parts of estuaries (Halliday and Robins, 2005; Welch et al., 2014). High salinity (>32%) is important for survival of pelagic eggs, and spawning occurs in marine waters away from the mouths of rivers where outflows may render salinity levels lower (Halliday et al., 2008; Robins and Ye, 2007; Welch et al., 2014). Juveniles move into shallow near-shore waters and estuaries.

On the east coast of Queensland, early juveniles are reported in estuaries between December and May, across a wide range of salinities but not in fresh water (Halliday et al., 2008; Welch et al., 2014). Juvenile king threadfin likely enter estuaries during the wet season when prawns and other prey species are seasonally abundant (Halliday et al., 2008). King threadfin juveniles 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). They are not found in ephemeral freshwater habitats created or connected during the wet season (Robins and Ye, 2007; Russell and Garrett, 1983). However, the extent and patchiness of wetland and salt flat habitats are likely to be important in supporting king threadfin production (Meynecke et al., 2008), perhaps via availability of prey.

In the Fitzroy River annual, spring and summer river flow was positively correlated with the year–class strength of king threadfin (Halliday et al., 2008). Halliday et al. (2008) and Halliday et al. (2012) postulated five reasons for high year–class strength correlating with high flows:
4. Outflows create chemical cues that enhance the immigration of post-larvae and juveniles into an estuary.

5. High flows enhance biological productivity of the estuary, thus enhancing prey abundance and resultant growth rates and survival of king threadfin.

6. High flows increase the extent of low-to-medium-salinity habitat, the estuarine ecotone, with benefits to energy budgets.

7. High flows reduce predation risk in highly turbid waters.

8. High flows stimulate the downstream movement of threadfin to increase their movement and density in areas where gill nets are set by commercial fishers.

An increase in extent of the ecotone may also enhance the abundance of prey species, as many small fish and crustaceans prefer euryhaline waters, rather than either very low or marine salinities. Halliday et al. (2012) also postulated that high flows may negatively affect king threadfin recruitment, because:

- they wash eggs and post-larvae away from the estuary mouth, that is, away from the advective envelope adjacent to the river mouth

- very high flood flows may create a freshwater environment in an estuary, which is a habitat unsuitable for threadfin.

While the year–class strength of king threadfin commercial catches was positively correlated with flood flow and coastal rainfall in central Queensland, the relationship did not hold for other locations. There was no relationship between high flows and catch in the Mitchell River on the west coast of Cape York; fishing effort was the dominant determinant of catch (Halliday et al., 2012). In the same river, the commercial catch of barramundi (controlled for fishing effort) was positively correlated with flow, lagged by two to three years.

In the Flinders River in the south-east Gulf of Carpentaria, river flow (lagged by one year) was significantly correlated with commercial catch of king threadfin (and barramundi, both controlled for fishing effort) (Halliday et al., 2012). The year–class strength of king threadfin was also positively correlated with catchment rainfall in the Flinders River (catch-year rain and wet-season rain). In the Roper River, western Gulf of Carpentaria, no flow data were available, but spring rainfall (lagged by three years) was positively correlated with threadfin and barramundi catch (Halliday et al., 2012). Autumn rain (same year) was negatively correlated with the threadfin catch. In the Daly River (east Joseph Bonaparte Gulf), king threadfin catch was dominated by fishing effort, and rainfall was not correlated with catch (adjusted for effort) or year–class strength of king threadfin (Halliday et al., 2012).

Movement along coastlines may confound potential relationships between river flow and catch (or year–class strength) (Halliday et al., 2012) and overfishing may cause inter-estuary and greater movement, as seems to be the case in the south-east Gulf of Carpentaria. In the subtropical Fitzroy River estuary, the large decadal variability in flow and failure of the wet
season, compared with the more predictable monsoonal rainfall in the tropics, could account for these positive correlations in the subtropical river (Halliday et al., 2012).

The lagged relationship between flow and catch in some rivers shows that early phases in their life-history benefit from high flows. The high productivity and prey abundance in estuarine habitats during low to moderate floods may support juvenile threadfin growth and survival, as suggested by (Halliday et al., 2012). Small fish and crustaceans (especially penaeid prawns), the prime prey of king threadfin (Blaber et al., 1995; Salini et al., 1990), are abundant in tropical estuaries in the pre-wet. However, many prey species respond to significant drops in salinity and emigrate from estuaries to near-shore shallows (Kenyon et al., 2012; Vance et al., 1998), so it is likely king threadfin would move from estuaries as their prey move offshore. To date, exploration of the relationship between flow and subsequent catch of king threadfin shows that the link is not strong. However, researchers have suggested that salinity cues associated with high flood flows may stimulate the movement of king threadfin out of estuaries, thus increasing their vulnerability to the fishery (Halliday and Robins, 2005; Robins and Ye, 2007). In winter, adults can be found in the middle to upper reaches of estuaries, when saline waters intrude up the estuary. However, they leave the estuaries with the onset of wet-season high flows (Halliday et al., 2008).

**Population biology and physiology**

King threadfin has a potential lifespan of at least 22 years, but lifespan varies markedly by geographic location, and individuals in many locations only reach 10 to 12 years of age (Moore et al., 2012b). They are a large fish (up to 1700 mm FL) and commonly reach approximately 1200 mm FL, though size differs between location and spatially separate populations may have a maximum size of less than 1000 mm FL (Moore et al., 2012b). They reach maturity between 2 and 5 years and individuals change sex from male to female between 2 and 10 years old, mostly at 4 to 10 years old (Moore et al., 2012b).

The length of the spawning season and timing of peak spawning varies among study areas. Spawning occurs over October to January (spring and summer) on the east coast of Queensland, between late winter and spring in the Gulf of Carpentaria and from October to March in the Northern Territory (Halliday et al., 2008; Moore et al., 2011).

King threadfin span a huge range of the tropical Australian coastline, and evidence suggests that the population is comprised of multiple stocks (Moore et al., 2012b; Newman et al., 2010). However, spatial separation per se is not always the driver of heterogeneity. On the Queensland subtropical coast, distinct stocks exist in the Fitzroy, Mary and Brisbane rivers, showing approximately 500 km separation; where length, age, growth rate, age at maturity and age at sex change all vary spatially (Moore et al., 2011). In contrast, in the Gulf of Carpentaria at spatial scales of 500 km, there was no evidence of separate stocks from either parasite load quantification (Moore et al., 2012a) or otolith stable isotope ratios and mitochondrial DNA signatures (Horne et al., 2012; Moore and Simpfendorfer, 2014). Thus, king threadfin in south-east Gulf waters form one discrete stock of post-recruit fish with little adult or larval interchange from other populations. Australia-wide, considerable variation in growth rates, age at sex change and maximum age for fish were found across
northern Australia (sampled 2007 to 2010), suggesting that a number of geographically or reproductively distinct groups exist (Moore et al., 2012b; Newman et al., 2010).

Distinct stocks in limited spatial ranges render any population of fish vulnerable to impacts from overfishing, loss of habitat and change to key environmental drivers. It has been suggested that the population of king threadfin in the Gulf of Carpentaria may be heavily overfished, resulting in a large proportion of small, young females and earlier age at sex change in fish. Overfishing may be the cause of a temporal instability in genetic structure compared with other regions, as fish move between estuaries over large spatial scales (Horne et al., 2012; Moore et al., 2012b; Welch et al., 2010). Moreover, change in natural flood flows due to water resource containment or diversion in one catchment can severely affect the distinct population (stock) that is restricted to that downstream estuarine system. Currently, the management of the king threadfin population is undertaken with no consideration of the presence of multiple stocks, and by three state jurisdictions (Newman et al., 2010). This strategy opens the population to localised depletion on small spatial scales – possibly from overfishing, but also from modification at the catchment scale of environmental drivers and ecosystem services that sustain the local stock.

**Fishery**

King threadfin are an important commercial species. They are fished by gill net from Eighty Mile Beach (WA) across northern Australia, to Rockhampton in Queensland, including the Gulf of Carpentaria (Halliday and Robins, 2005; Newman et al., 2010; Robins et al., 2005). Together with the blue threadfin (*Eleutheronema tetradactylum*), their catch ranges from 800 to 1200 tonnes and the stocks span three state jurisdictions, each of which has its own management plan.

King threadfin are a highly important species to commercial, charter and recreational fisheries in the Gulf of Carpentaria. Although catch data from Queensland fisheries is not available by fishery, the majority of the commercial catch comes from the commercial N3 gill-net fishery. The commercial catch of king threadfin in the Gulf of Carpentaria has been stable since 1989 with peaks in 1991 and 2001 of 486 and 473 t, respectively. Catches have been stable at around 300 t since 2003, with only 2013 recording a catch lower than 200 t. The recreational catch in 2010 was estimated to be 7193 fish, equivalent to around 21 t if assuming the mean fish weight was 3 kg. Interestingly, the catch of king threadfin by the charter fishery has been very low in the Gulf of Carpentaria, averaging less than 1 t over the past 5 years.

**Conceptual model**

Figure 4-39 shows the conceptual model for king threadfin, with the accompanying narrative given in Table 4-13.
Figure 4-39 Conceptual model illustrating the relationship between threats, drivers, effects and outcomes for king threadfin in northern Australia
### Conceptual model narrative

#### Table 4-13 Knowledge underpinning the conceptual model for king threadfin in northern Australia

<table>
<thead>
<tr>
<th>1. HABITAT AND THREATS</th>
<th>2. DRIVER/STRESSOR</th>
<th>3. SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow Timing and amplitude of monsoonal rainfall and flooding important in maintaining larval access to estuaries (immigration cue) and juvenile habitats (estuarine ecotone)</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. The estuarine ecotone available to king threadfin is lost, thus critical habitat is lost. (Halliday et al., 2012; Welch et al., 2014).</td>
<td>The reduction of brackish water habitat reduces the ability of king threadfin population to forage and grow in the estuary. King threadfin populations decline. Strong flows during spawning and larval immigration may impede the movement and successful recruitment of larvae from early marine pelagic habitats to the demersal habitats in the estuary.</td>
<td>Exclusive recruitment of early juveniles to estuaries and close-inshore waters. Preference for brackish waters by juvenile king threadfin possibly due to prey abundance supporting survival and growth; and protection from larger fish predators via turbid waters (Halliday et al., 2012; Halliday et al., 2008). Negative correlation between king threadfin fishery catch (CPUE) and high autumn rainfall in the Roper River, western GOC (Halliday et al., 2012). Autumn flows may disrupt the estuarine habitats of juvenile king threadfin or cause adults to leave the estuary if it becomes near-freshwater.</td>
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<tr>
<td>Extraction/regulation and Climate change (drier scenario)</td>
<td>Weak/out-of-sync monsoon early rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. Baseflow occurs out of sync with the spawning season of king threadfin (spring/summer). The horizontal extent of brackish waters decreases or is lost to the estuary.</td>
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<td>High or artificially elevated baseflow during the spawning season may impact the ability of larvae to access estuaries in the early stages due to strong outflows and low salinity (Halliday et al., 2012; Halliday et al., 2008).</td>
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<tr>
<td>Reduction in flood flow inundation/flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows Timing of monsoonal rainfall and flooding important in maintaining juvenile habitats (a turbid estuarine ecotone) Seasonal flood flows cause king threadfin to move downstream within an estuary where they may be available to fishing nets, resulting in higher catches However, they move towards saline waters more suitable for spawning as well</td>
<td>Reduction in flow inundation/flooding regime</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the persistence of the ecotone during the early-to-mid dry season when juvenile king threadfin are abundant in estuaries. Optimal habitat (brackish turbid waters) for juvenile and sub-adult king threadfin is lost. Reduction in emigration cue for 2+ to 3+ king threadfin that have been resident in estuaries for 2-3 years as older threadfin prefer marine waters (Halliday et al., 2012; Halliday et al., 2008).</td>
<td>Reduction in brackish turbid water habitat limits the ability of king threadfin to forage and grow in the estuary (Halliday et al., 2008). Reduction in estuarine turbidity increases the likelihood of king threadfin being consumed by large predators while estuarine residents. King threadfin populations decline. Adult king threadfin that have been resident in estuaries for ~3 years remain distributed throughout the estuary; some in areas unsuitable for spawning success.</td>
<td>High abundance of king threadfin in the mid-to-lower reaches of a tropical estuary during the dry season; retreating to the lower reach in the late dry when mid-to-upper reaches become hypersaline due to nil baseflow and evaporation (Cyrus and Blaber, 1992). High abundance of king threadfin in turbid waters (&gt;10 NTU) (Cyrus and Blaber, 1992; Halliday et al., 2012). Positive correlation between king threadfin fishery catch (CPUE) and high annual flows along the Queensland and Gulf of Carpentaria coasts (Halliday et al., 2012; Halliday et al., 2008).</td>
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<td>Reduction in flood flow volume</td>
<td>Maintenance of volume of volume of water leaving the estuary to the plume zone; and duration of peak flows</td>
<td>Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the floodplume.</td>
<td>Limitation of primary production in the floodplume and near-shore waters due to nutrient limitation. Export of terrestrial-sourced nutrients through the estuary. Reduction in primary production within estuaries during floods; (Burford et al., 2012) perhaps stimulating estuarine to near-shore emigration.</td>
<td>During a 1-in-50 year flood, an estimated 4300 tonne of nitrogen and 800 tonne of phosphorus was exported he Norman River estuary (southeast GOC) and deposited in the near-shore zone (Burford et al., 2012).</td>
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<tr>
<td>Change in inter-annual sequencing</td>
<td>Inter-annual expansion of estuarine habitats during high flood flows</td>
<td>Juvenile king threadfin that access supra-littoral habitats during the wet season benefit from access to extensive and seasonally-productive habitat.</td>
<td>Loss of seasonal expansion of habitats adjacent to the estuary or within the coastal zone causes mortality of juvenile king threadfin seeking a niche in inshore habitats.</td>
<td>Use of brackish, turbid waters in supra-littoral habitats by juvenile and sub-adult king threadfin (Cyrus and Blaber, 1992; Halliday et al., 2012; Halliday et al., 2008; Robins and Ye, 2007).</td>
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<tr>
<td>Saltwater intrusion</td>
<td>Expansion of saline habitats in the upper littoral and supra-littoral zone on the outer margins of estuaries</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion may flood and expand supra-littoral habitats and increase the extent of shallow-tidal habitats available to king threadfin. In contrast, saline intrusion in the main estuary reduces the extent or presence of a turbid, brackish ecotone which is optimal habitat for king threadfin.</td>
<td>Contrasting effects of expanded upper tidal habitats on the outer margins of estuaries, offset by a reduction in the extent of optimal salinity in estuarine waters. The micro-habitat use of juvenile king threadfin is poorly understood, consequently the resultant impact of saltwater intrusion on king threadfin population requires investigation.</td>
<td>Use of brackish and turbid waters in supra-littoral, littoral and sub-littoral habitats by juvenile (brackish waters) and adult king threadfin (Cyrus and Blaber, 1992; Robins and Ye, 2007).</td>
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<tr>
<td>Estuarine/saltpan wetland and floodplain</td>
<td>Loss of habitat extent Habitat degradation/ homogenisation</td>
<td>Annual flooding of salt flats and wetlands mark an inundation expansion of habitat available to king threadfin. Desiccated algal crust on salt pans invigorates and begins to photosynthesise. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016).</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation salt pan and wetland adjacent to tropical estuaries reduces key habitat that becomes available to estuarine fauna during the monsoon season. Reduction in habitat available to juvenile king threadfin during monsoon season large flood flows.</td>
<td>Reduction in population size of juvenile king threadfin and changes in distribution/habitat use. Reduction in nutrients transferred within and exported from tropical estuaries with subsequent reduced coastal productivity for resident and emigrant species (Burford et al., 2016; Burford, 2011). Reduction in nutrient production from flooded salt pans during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients transferred within tropical estuaries with subsequent reduced coastal productivity and flow-on impacts for likely prey species for estuarine and near-shore demersal predatory fish (Burford, 2011).</td>
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<tr>
<td>Change in inundation frequency, depth and duration Change in land use and management</td>
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<td>Longstream habitat connectivity Loss of migration pathways between estuarine and riverine habitats</td>
<td>Loss of continuation and extent of juvenile and adult habitats (marine waters and estuarine reaches of rivers) due to the placement of barriers in a reach of the estuary</td>
<td>Juvenile and adult king threadfin move upstream within estuaries and inhabit lower, mid- and upper reaches of estuaries. Growth and survival likely is optimal in brackish habitats supported by baseflow entering the estuary year-round or seasonally.</td>
<td>Physical barriers truncate the estuary and may remove a considerable extent of the available estuarine habitat for king threadfin. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation.</td>
<td>Reduced habitat extent for juvenile and adult king threadfin. Reduction in population size of king threadfin.</td>
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<td>Reduced habitat extent for juvenile and adult king threadfin. Reduction in population size of king threadfin.</td>
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<td>The optimal salinity for the growth and survival of juvenile and adult king threadfin has not been explored under laboratory experimental conditions. Field-based distribution observations of their estuarine distribution suggest king threadfin prefer brackish waters as juveniles and turbid waters for juveniles and adults.</td>
<td>Use of brackish and turbid waters in supra-littoral, littoral and sub-littoral habitats by juvenile (brackish waters) and adult king threadfin (Cyrus and Blaber, 1992; Robins and Ye, 2007). Reduction in nutrient production from flooded salt pans during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients transferred within tropical estuaries with subsequent reduced coastal productivity and flow-on impacts for likely prey species for estuarine and near-shore demersal predatory fish (Burford, 2011).</td>
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<td>Either low or nil baseflow causing the fluvial separation of riverine reaches and the estuary</td>
<td>The horizontal extent of turbid and brackish waters decreases or is lost to the estuary. Optimal high turbidity and moderate salinity for king threadfin is lost, thus critical habitat is lost.</td>
<td>NA</td>
<td>adults. Their distribution within an estuary varies between the wet season, the early and the late dry seasons as salinity and turbidity regimes vary (Cyrus and Blaber, 1992).</td>
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<tr>
<td>Riverine floodplain wetlands</td>
<td>Reduced access to waterhole/wetland habitats on floodplains</td>
<td>NA</td>
<td>NA</td>
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<td>Reduced access due to less frequent and lower flood events</td>
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<td>Water column habitats all waters</td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation</td>
<td>Loss of water clarity, light penetration Loss of water column productivity/ algal productivity Loss of epibenthic productivity</td>
<td>Riverine and estuarine production is sustained by microalgae, macro algae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of planktonic and benthic food-web within the estuary. Reduction in dependent higher trophic level species that are prey items for king threadfin.</td>
<td>Synthesis of literature.</td>
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</table>
GRUNTER

Asset description

Grunter \((Pomadasys kaakan)\), also known as barred javelin, are medium-sized (~0.7 m), non-diadromous, carnivorous fish (order Perciformes) that are widely distributed throughout the Indo–Pacific from the Red Sea and the east coast of Africa to south-east Asia and northern Australia (Froese and Pauly, 2014; Welch et al., 2014). In Australia, grunter can refer to three co-generic species \((P. kaakan, P. argenteus\) and \(P. maculatus)\) in commercial and recreational catches. This review describes \(P. kaakan\).

Grunter range from Shark Bay, WA to the north coast of NSW. Grunter are a demersal species that complete their entire life cycle in estuaries and inshore marine waters. They are able to tolerate a wide range of salinities and do not use freshwater habitats (Welch et al., 2014). In Australia, they live to approximately 14 years and are a slower-growing fish that matures early (3 years and 50% mature at <250 mm FL) (Welch et al., 2014). They are an important ecological species: a top predator in the fish community, consuming a range of crustacean (in particular) and fish prey in the coastal ecosystem (Bade, 1989; Blaber et al., 1995; Salini et al., 1990). Their body form and quality of flesh makes them a prized table fish.

Grunter is an important by-product species in commercial, inshore gill-net fisheries. Moreover, it is a prized target species for recreational fisheries throughout wet–dry tropical Australia (Newman et al., 2004; Welch et al., 2014). Grunter typify an estuarine-dependent fish whose habitat, and the habitat of their prey, could be significantly affected by extraction of natural river flows, given the role of flows in maintaining the turbid, brackish ecotone of the rivers in tropical northern Australia.

Distribution in Assessment study areas

Figure 4-40, Figure 4-41, and Figure 4-42 show the recorded sightings of the barred grunter in the Assessment catchments and marine study areas.
Figure 4-40 Distribution of barred javelin (grunter) in the Fitzroy catchment and marine study area
Figure 4-41 Distribution of barred javelin (grunter) in the Darwin catchments and marine study area
Figure 4-42 Distribution of Barred javelin (grunter) in the Mitchell catchment and marine study area

**Habitat use**

Grunter are widely distributed throughout tropical Australian inshore marine and estuarine habitats. They grow to a maximum length of 80 cm TL (Froese and Pauly, 2014). The life history, habitats and population biology of grunter has been little studied, with only three species-specific studies being undertaken on the species in Australian waters (Cleveland Bay, Townsville, (Bade, 1989); south-east Gulf of Carpentaria, (Garrett et al., 1997); Hinchinbrook Channel, (Szczecinski, 2012). Few studies of their stock delineation have been undertaken (Welch et al., 2014).

Research that has targeted the general fish community and predators of penaeid prawns in the Gulf of Carpentaria provides detail on their distribution and habitat use at a broad scale. Grunter are abundant in inshore shallow waters and estuaries, though they occur Gulf-wide (Blaber et al., 1994; Blaber et al., 1995). Both juveniles and adults are abundant in coastal embayments and estuaries. In estuaries they are more abundant in the lower and middle reaches, but also occur upstream (Blaber et al., 1989; Blaber et al., 1995).
Grunter are a euryhaline species. They can tolerate a range of salinities (0 to 39.0 PSU) and turbidities (1.2 to 21.0 NTU), though in estuaries their catch rate declines as salinity increases (Cyrus and Blaber, 1992). They are widely distributed throughout the Kimberley region of WA and the Northern Territory. They are a target recreational fishery species and also comprise a significant portion of the by-product of commercial catch (Newman et al., 2004; Welch et al., 2014). Grunter are mangrove, mud habitat and estuarine-associated fish; they are not abundant in littoral seagrass habitats in the north-west Gulf of Carpentaria (Blaber et al., 1992; Brewer et al., 1995).

Juvenile grunter recruit to a range of shallow near-shore habitats, including estuaries, seagrass beds and coastal embayment prawn trawl grounds (Blaber et al., 1989; Blaber et al., 1990; Blaber et al., 1992). Wet-season rainfall and flood flows promote the recruitment of juvenile grunter through the connectivity of estuarine habitats to ephemeral supra-littoral wetlands on the estuarine margin, expanding their habitat extents (Robins and Ye, 2007). Robins and Ye (2007) also suggest flood flows may cue migration of grunter, and that aggregation in response to high abundances of prey species may also be cued by salinity declines and resultant shifts within estuaries.

In the Cleveland Bay region (Townsville, Queensland), juvenile grunter recruit to inshore waters in October to December and February to April (Bade, 1989). Throughout the year, juvenile and adult fish in the size range from about 130 to 330 mm TL were found in the shallow coastal Cleveland Bay, though fish greater than approximately 250 mm TL were only seasonally abundant (February and September to October). In shallowest beachside waters of the bay, grunter ranged from 20 to about 140 mm TL during those months of recruitment (Bade, 1989). In a nearby estuary (Barramundi Creek), juvenile and adult grunter in the size range of 130 to 400 mm TL were abundant, with smaller fish being more abundant from September to December and February to April (Bade, 1989). The data suggest that grunter spawn in coastal bays. Their larvae and earliest juvenile phases occupy the bay, moving inshore to shallow beachside waters as they metamorphose to small fish, after which juvenile fish in the 100 to 150 mm size range move into the estuaries. Likewise, both juvenile and adult grunter were found in Albatross Bay (80 to 555 mm) and the nearby estuary of the Embley River (53 to 510 mm), Gulf of Carpentaria (Blaber et al., 1989). Grunter grow much faster in spring, summer and early autumn than in winter (May to July).

Little is known of the movements or population genetic structure of grunter in Australian waters (Welch et al., 2014). However, given its widespread distribution, occupation of a range of habitats and high salinity and turbidity tolerance, it is likely that grunter exists as a single stock throughout the Gulf of Carpentaria. However, Garrett et al. (1997) suggest that there might be more than one stock in the Gulf of Carpentaria, in contrast to the single stock of barramundi.

Floods can modify the distribution of likely prey within an estuary. Therefore, they can also modify the distribution of dependent fish (Duggan et al., 2014; Vance et al., 1998; Welch et al., 2014). The grunter is a generalist feeder, consuming a variety of fish and benthic invertebrates, including penaeid prawns, in the Gulf of Carpentaria estuaries (Salini et al.,
Bade (1989) found that decapod crustaceans make up the dominant portion of their prey for all fish, but particularly fish greater than 150 mm FL. It has been suggested that aggregations of grunter around river mouths in winter months form in response to seasonal concentrations of prey, possibly white banana prawns (Robins and Ye, 2007). In South Africa, the estuarine distribution of a co-generic grunter, *P. commersonnii*, has been shown to correlate with the distribution of its prey. Their distribution in an estuary is sympatric with the distribution of epibenthic prey species in the lower estuary (Childs et al., 2008a; Childs et al., 2008b; Childs et al., 2008c; Maree et al., 2016). In addition, the African grunter has a home range that increases with fish size. As suggested for Australian grunter, South African spotted grunter tolerate a range of salinities, and move to and from estuaries in ventures to the marine environment for period of 1 day to about 9 days. No strong salinity cue occurred during these studies (Childs et al., 2008a; Childs et al., 2008b).

**Physiology**

Much of the biological knowledge of grunter comes from the waters of Iran (Fakhri et al., 2011; Falahatimarvast et al., 2012), Pakistan (Majid and Imdad, 1991) and Kuwait (Al-Husaini et al., 2001; Al-Husaini et al., 2002a). In the northern Persian Gulf, grunter live to 12.5 years and grow to 600 to 700 mm TL; females are 50% more numerous than males. They mature at about 393 mm TL for males and 477 mm for female fish, and Von Bertalanffy growth parameters were estimated for the stock (Fakhri et al., 2011; Falahatimarvast et al., 2012). In the waters of Kuwait, large fish were a similar size (600 to 700 mm TL, and 10 to 15 years old), while the maximum size and age for males was 750 mm TL and 36 years, and was 730 mm TL and 27 years for females (Al-Husaini et al., 2002b). Females grow faster than males and are larger at maturity; 50% of female grunter were mature at 390 mm TL and 2.3 years old. The sex ratio of fish is significantly skewed towards females, although there is no evidence of ontogenetic sex inversion from macroscopic staging of gonads. In the Middle East, grunter exhibit a prolonged spawning season with multiple spawnings. They are mature and spawn through the northern hemisphere spring and early summer (Falahatimarvast et al., 2012).

In north-eastern Australia (Hinchinbrook Channel), grunter are slow-growing, reaching 50% of maximum length by about 4 years, and living to at least 15 years (Szczecinski, 2012). Growth parameters (K=0.177) in the Hinchinbrook Channel are lower than in the Middle East. However, in the Gulf of Carpentaria growth parameters are higher than estimates from the Middle East (K=0.35, Garrett et al., 1997). In the Gulf of Carpentaria and Queensland’s east coast, large fish are in the range of 650 to 700 mm TL, males having a larger size-mode than females. In the Hinchinbrook Channel, grunter mature early in life, with 50% of females mature by age 2 to 3 years (280 to 319 mm TL) (Szczecinski, 2012). This is significantly smaller than the length at 50% maturity estimate of 477 mm TL – based on histology – for female fish in the waters of Iran (Falahatimarvast et al., 2012). In the Gulf of Carpentaria, grunter are mature at 366 mm for males and 466 mm TL for females, lengths similar to the Middle East (Garrett et al., 1997). As in the Middle East, the sex ratio is skewed to females; with greater than twice the number of females caught than males.
Grunter spawn in north-eastern Australia between August to March, peaking in September and October (Szczecinski, 2012) and August to November (Bade, 1989). They have a prolonged spawning period, and about 50% of females are maturing or in spawning condition from August to February (Szczecinski, 2012). In the Gulf of Carpentaria, the Gonad Somatic Index of both male and female grunter rose from autumn lows through June and July and peaked in August, remaining high until October (Garrett et al., 1997). Large fish spawn approximately 2 million eggs (Bade, 1989). Spawning is thought to occur outside of estuaries in marine waters, since eggs require high salinity for survival (Robins and Ye, 2007).

**Fishery**

Grunter is a highly important species to commercial, recreational and Indigenous fisheries in the Gulf of Carpentaria, the Northern Territory and the Kimberley. Although catch data from Queensland fisheries is not available by fishery, the majority of the commercial catch comes from the commercial N3 gill-net fishery. The commercial catch of grunter in the Gulf of Carpentaria has been consistently above 10 t since the early 1990s, with the catch peaking at 52 t in 2004. In the Gulf of Carpentaria Inshore Fin Fish Fishery, the mean catch over the seven years to 2009 was 27 t (Welch et al., 2014). In the 4 years to 2013, the annual catch has averaged 14 t (Griffiths et al., 2014). The annual charter catch peaked at 3.6 t in 1997 but has only averaged 350 kg per year over recent years.

Grunter was the third most important species (by number) to recreational fishers in the Normanton region in the 2010 statewide survey. Recreational fishers caught an estimated 37,657 fish, of which an estimated 54% were released. The total catch, assuming a mean weight of 1.5 kg, equates to around 56 t. This is one of the few species where the recreational catch exceeds the commercial catch. Because grunter inhabit inshore waters and estuaries, they are also likely to be caught by Indigenous fishers (Zeller and Snape, 2006).

In the Kimberley and Pilbara region of WA, grunter is one of the 20 most popular fish taken by the recreational sector; they rank mid-range of the top 20 in terms of catch (Newman et al., 2004). Grunter are more common in Kimberley than Pilbara waters. They are taken as by-product in inshore commercial gill-net fisheries.

**Hydrological indicators**

Extraction of water from rivers and subsequent reduced flow levels reduce local populations of grunter and ontogenetic habitat access. There is no documented correlation between catch or year-class strength of grunter and spring and summer flood flows. Grunter spawn in the near-shore marine habitat and move to estuaries as juveniles. Baseflow in the spring and early season low flows likely are used by grunter larvae in marine habitat as cues to access estuaries. Monsoon flows create a brackish ecotone within estuaries that suit grunter and their prey as prime habitat. Flood flows deliver nutrients, increase turbidity and cause shifts in the distribution of the prey of grunter in estuaries, likely determining fish
distribution and movement to the lower estuary during floods. They may become more available to exploitation when accumulating in the lower estuary and close to near-shore.

Conceptual model

Figure 4-43 shows the conceptual model for grunter, with the accompanying narrative given in Table 4-14.
## Conceptual model narrative

### Table 4-14 Knowledge underpinning the conceptual model for grunter in northern Australia

<table>
<thead>
<tr>
<th>1 HABITAT AND THREATS</th>
<th>2 DRIVER/STRESSOR</th>
<th>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</th>
<th>4. EFFECT</th>
<th>5. OUTCOME</th>
<th>6. EVIDENCE AND PATHWAYS</th>
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</thead>
<tbody>
<tr>
<td>Estuarine mangrove, seagrass and mudbank habitats</td>
<td>Reduction in baseflow due to either water extraction/regulation and/or decrease in rainfall</td>
<td>Maintenance of baseflow Timing and amplitude of monsoonal rainfall and flows is important in maintaining estuarine habitats (including a brackish ecotone) and possibly juvenile access to estuarine habitats via physical/chemical immigration cues</td>
<td>Disruption/cessation of baseflow causes the loss of freshwater inputs to the estuary. Thus the salinity of the estuary increases and may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost to the estuary. Optimal non-marine salinity for juvenile and adult grunter is lost (Cyrus and Blaber, 1992). Simultaneously, the optimal brackish water habitats of key grunter prey species (e.g. banana prawns) also declines or is lost (Vance et al., 1998). Weak/ out-of-sync monsoon low rainfall, low runoff and reduced groundwater maintenance. Reduction in baseflow and freshwater inputs to estuary. Baseflow occurs out of sync with the recruitment of juvenile grunter to estuaries. The horizontal extent of brackish waters decreases or is lost to the estuary. Low baseflow may impact the ability of early juveniles to access estuaries as little to no flow negates the presence of immigration cues to estuaries (either physical such as declining salinity, or chemical cues from freshwater loads).</td>
<td>The reduction of brackish water habitat reduces the ability of the grunter population to forage and grow in the estuary. Grunter populations decline. Low or nil flows during juvenile immigration may impede the movement of small juveniles from early marine habitats to the estuary, reducing the recruitment strength of grunter.</td>
<td>Greater abundance of grunter in salinities lower that 33 ppt marine salinity (Cyrus and Blaber, 1992). Higher growth rates and lower mortality in brackish waters by grunter prey; e.g. juvenile and banana prawns (Staples and Heales, 1991).</td>
</tr>
<tr>
<td>1 HABITAT AND THREATS</td>
<td>2 DRIVER/STRESSOR</td>
<td>3 SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
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<tr>
<td>Reduction in flood flow inundation/flooding regime</td>
<td>Maintenance of amplitude, volume and duration of peak flows Timing of monsoonal rainfall and flooding important in maintaining brackish water habitats and flooding of supra-littoral habitats</td>
<td>Reduction in flood flow volume and duration reduces brackish ecotone and the inundation of supra-littoral habitats adjacent to the estuary and in seasonal coastal wetlands. As well, a brackish ecotone is optimal for prey species such as decapod crustaceans. The persistence of the ecotone during the post-wet season (autumn) when the grunter recruit to estuaries. Optimal brackish water habitat is lost during these crucial months. Reduction in extent and quality of juvenile grunter seasonally ephemeral habitat (Robins and Ye, 2007).</td>
<td>Reduction in brackish water habitat limits the ability of grunter to forage and grow in the estuary. Grunter populations decline. Reduction in possible intra-estuarine movement cue for adult grunter residents in estuaries but requiring suitable habitats for foraging and spawning.</td>
<td>Preference for brackish waters by juvenile and adult grunter (Cyrus and Blaber, 1992; Robins and Ye, 2007).</td>
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<tr>
<td>Reduction in flood flow volume</td>
<td>Maintenance of volume of water leaving the estuary to the plume zone; and duration of peak flows</td>
<td>Reduction in flood flow volume and duration reduces export of nutrients to inshore coastal waters within the floodplume.</td>
<td>Limitation of primary production in the floodplume and near-shore waters due to nutrient limitation. Export of terrestrial-sourced nutrients through the estuary. Reduction in primary production within estuaries during floods (Burford et al., 2012); perhaps stimulating estuarine to near-shore emigration.</td>
<td>During a 1-in-50 year flood, an estimated 4300 tonne of nitrogen and 800 tonne of phosphorus was exported from the Norman River estuary (south-east GOC) and deposited in the near-shore zone (Burford et al., 2012).</td>
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<tr>
<td>Change in inter-annual sequencing</td>
<td>Inter-annual expansion of estuarine habitats during high flood flows</td>
<td>Juvenile grunter that access supra-littoral habitats during the wet season benefit from access to extensive and seasonally-productive habitat.</td>
<td>Loss of seasonal expansion of habitats adjacent to the estuary or within the coastal zone causes mortality of juvenile grunter seeking a niche in inshore habitats.</td>
<td>Use of brackish waters in supra-littoral habitats by juvenile and sub-adult grunter (Cyrus and Blaber, 1992; Robins and Ye, 2007).</td>
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<tr>
<td>1 HABITAT AND THREATS</td>
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<tr>
<td>Estuarine/saltpan wetland and floodplain</td>
<td>Saltwater intrusion</td>
<td>Expansion of saline habitats in the upper littoral and supra-littoral zone on the outer margins of estuaries</td>
<td>Climate change, storm intensity increase, sea-level rise, reduced flows resulting in increased saltwater intrusion. Saltwater intrusion may flood and expand supra-littoral habitats and increase the extent of shallow-tidal habitats available to grunter. In contrast, saline intrusion in the main estuary reduces the extent or presence of a brackish ecotone which is optimal habitat for grunter.</td>
<td>Contrasting effects of expanded upper tidal habitats offset by a reduction in the extent of optimal salinity in estuarine waters. The resultant impact on grunter population requires investigation.</td>
<td>Use of brackish waters in supra-littoral, littoral and sub-littoral habitats by juvenile and adult grunter (Cyrus and Blaber, 1992; Robins and Ye, 2007).</td>
</tr>
<tr>
<td>Change in inundation frequency, depth and duration Change in land use and management</td>
<td>Loss of habitat extent Habitat degradation/Homogenisation</td>
<td>Annual flooding of salt flats and wetlands mark an inundation expansion of habitat available to grunter. Desiccated algal crust on salt pans invigorates and begins to photosynthesise. Carbon and nitrogen production occur with input to the estuarine ecosystem and the food chain (Burford et al., 2016)</td>
<td>Disruption to the frequency and inundation depth (extent) of large areas of low elevation salt pan and wetland adjacent to tropical estuaries reduces key habitat that becomes available to estuarine fauna during the monsoon season. Reduction in habitat available to juvenile and sub-adult grunter during monsoon season large flood flows.</td>
<td>Reduction in population size of juvenile and sub-adult grunter and changes in distribution/habitat use. Reduction in nutrients transferred within and exported from tropical estuaries with subsequent reduced coastal productivity for resident and emigrant species (Burford et al., 2016; Burford et al., 2012)</td>
<td>Reduction in nutrient production from flooded salt pans during monsoon season large flood flows (Burford et al., 2016). Reduction in nutrients transferred within tropical estuaries with subsequent reduced coastal productivity and flow-on impacts for likely prey species for estuarine and near-shore demersal predatory fish (Burford et al., 2011).</td>
</tr>
<tr>
<td>Longstream habitat connectivity Loss of estuarine habitats</td>
<td>Loss of continuation and extent of juvenile and adult habitats (marine waters and estuarine reaches of rivers) due to the placement of barriers in a reach of the estuary</td>
<td>Juvenile and adult grunter move upstream within estuaries and inhabit lower, mid- and upper reaches of estuaries. Growth and survival likely is optimal in brackish habitats supported by baseflow entering the estuary year-round or seasonally</td>
<td>Physical barriers truncate the estuary and may remove a considerable extent of the available estuarine habitat for grunter. The salinity of the estuary below the barrier increases due to the pondage of baseflows by the barrier. Estuary may become hypersaline due to evaporation. The horizontal extent of brackish waters decreases or is lost</td>
<td>Reduced habitat extent for juvenile and adult grunter. Reduction in population size of grunter.</td>
<td>The optimal salinity for the growth and survival of juvenile and adult grunter has not been explored under laboratory experimental conditions. Field-based distribution observations of their estuarine distribution suggest bared javelin prefer brackish waters and that their distribution in...</td>
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<tr>
<td>HABITAT AND THREATS</td>
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<td>SUPPORTING SYSTEM PROCESSES AND/OR EXPOSURE</td>
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<tr>
<td>Water column habitats all waters</td>
<td>Sediment loads in flows creating high turbidity and downstream sedimentation</td>
<td>Loss of water clarity, light penetration Loss of water column productivity/ algal productivity Loss of epibenthic productivity</td>
<td>Riverine and estuarine production is sustained by microalgae, macroalgae, mangroves and seagrass. High turbidity may cause loss of photosynthetic capacity of the algal community on both the substrate and in the water column, destabilising riverine and estuarine food webs (Burford et al., 2012; Faggotter et al., 2013).</td>
<td>Loss of planktonic and benthic food-web within the estuary. Reduction in dependent higher trophic level species that are prey items for grunter.</td>
<td>Synthesis of literature.</td>
</tr>
</tbody>
</table>

Either low or nil baseflow causing the fluvial separation of riverine reaches and the estuary to the estuary. Optimal salinity for grunter is lost, thus critical habitat is lost. An estuary varies between the wet season, the early and the late dry seasons (Cyrus and Blaber, 1992).
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