Chapter 7 discusses a range of potential risks to be considered before establishing a greenfield agriculture or aquaculture development. These include the ecological implications of altered flow regimes, a range of biosecurity considerations, off-site impacts from sediments, nutrients and agropollutants, irrigation drainage and aquaculture discharge water and irrigation-induced salinity. The key components and concepts of Chapter 7 are shown in Figure 7-1.
7.1 Summary

This chapter provides information on the ecological, biosecurity, off-site and irrigation-induced salinity risks to the Mitchell catchment from greenfield agriculture or aquaculture development. It is principally concerned with the risks from these developments to the broader environment but also considers biosecurity risks to the enterprises themselves.

7.1.1 KEY FINDINGS

Ecological implications of altered flow regimes

Although irrigated agriculture in Australia typically occupies a small percentage of a catchment area (i.e. <3%), it can potentially use a large proportion of the water (i.e. greater than 30%). Consequently, it was important for the Assessment to consider ecological changes to near-shore marine, estuarine, freshwater and riparian ecosystems that may result from changes in streamflow following water resource development for irrigation and other uses. It should be noted, however, that several other human-related factors can also impact on these ecosystems, including grazing, fire, disease, invasive species and changes in water quality. These factors are discussed qualitatively in the companion technical reports on ecology (Pollino et al., 2018a, 2018b).

This section provides an overview of outcomes from the ecology asset analysis, considering impacts of potential dams and water harvesting. It was found that the sensitivities of ecological assets to changes in flow vary. Vulnerabilities of assets change depending on their location in the catchment, the type and size of the development, volumes of water extracted and timing of water extraction. The sensitivity of assets to impacts also varies according to their dependencies on different parts of the flow regime.

The Mitchell catchment is largely intact in terms of the continuity of its plant and animal communities and the ecological processes that support them. The highly seasonal flows underpin river-floodplain productivity and provide critical habitats for species, including freshwater sawfish, a barramundi fishery and the extensive Northern Prawn Fishery, one of the most valuable fisheries in the country. The catchment supports significant wetland habitats and discharges into the Gulf of Carpentaria.

In the Mitchell catchment, the introduction of the potential Pinnacles dam causes dramatic changes in flow, with localised catastrophic impacts on migratory fish, including sawfish and barramundi, and on waterhole habitats. Downstream, these changes are moderated by flows from joining tributaries, such that only minor changes occur to the habitat of species near the mouth of the Mitchell River. Other potential dams analysed as part of the Assessment have considerably less impact on the habitat of near-shore marine and freshwater species than the potential Pinnacles dam, although all dams result in dramatic changes in flow in the reach immediately below their dam wall. This in part highlights the ecological importance of the Mitchell River relative to its major tributaries, the Palmer, Walsh and Lynd, which are more ephemeral in nature than the Mitchell River. With both the potential Pinnacles and Rookwood dams in combination, moderate changes to the habitat of some species, including the freshwater sawfish, occur along the Mitchell River below its confluence with the Palmer River. Major changes to the habitat of species occur along the Walsh River near its confluence with the Mitchell River, and below the
Rookwood dam. Dams and re-regulating structures such as weirs are likely to impact on the passage of migratory fish, although potentially less so for sawfish where dams are above their mapped ranges.

Water harvesting typically causes less disruption to the high- and low-flow extremes than major instream dams and as a consequence the impact to habitat of species is generally lower, particularly if pumping/diversions only commence at a high minimum-flow threshold (i.e. 1800 ML/day). High minimum-flow thresholds, however, reduce the reliability of extracting a given volume of water. Under the water harvesting scenarios examined as part of the Assessment, changes to the habitat of near-shore marine and estuarine species are minor, except for whole-of-system extraction volumes of less than 600 GL/year with a high flow threshold, where no changes are evident. Thereafter, for extraction volumes between 600 and 6000 GL, minor changes to habitat occur. Elsewhere in the Mitchell catchment water harvesting with a high flow threshold results in no change or minor change to habitats of freshwater species, except in some reaches when the whole-of-system extraction equals or exceeds 3600 GL/year. Under a low flow threshold, moderate impacts to habitat important to freshwater species occurs with whole-of-system extraction volumes of 1200 GL and higher, with the impacts becoming major for extraction volumes greater than 4800 GL/year.

Biosecurity considerations

Compared with many other countries, Australia has many advantages in terms of the opportunity to mitigate risks from pests and disease due to its isolation and sound regulatory processes. However, there have been a number of recent disease outbreaks, such as the disease of bananas, Panama disease tropical race 4, which highlight the risks to enterprises and to industries. The recent discovery of white spot syndrome virus in south-east Queensland prawn farms similarly shows the potential for disease to damage whole industries and to have a negative impact on industries which depend on wild catch. Man-made pathways include road transport, ships and planes, and the ‘carriers’ (e.g. humans, animals, plants, machinery) that facilitate the movement and incursion of new pests, diseases and weeds. In the short to medium term, biosecurity risks to the Mitchell catchment are most likely to come from within Australia and, in particular, from adjacent or climatically similar parts of the country, although wind dispersal from countries to the north is also possible. The warmer, north Australian environment is more favourable than temperate climates for insects and pathogens to adapt and multiply with the introduction of a new food source (e.g. a crop). However, the environment also favours beneficial organisms that prey on pest species. A range of macro-pests also pose a risk, such as feral pigs. Pathogens pose a constant risk to aquaculture enterprises. In some cases management actions aim to prevent the introduction of the pathogen to the enterprise, in other cases the pathogen is present in the farmed population and needs to be managed through good husbandry practices. There is also potential for disease in farmed populations to escape into wild populations. Similarly, irrigated agriculture has the potential to introduce weeds into the broader environment.

Sediment, nutrients and agropollutant loads to receiving waters

Agriculture can affect the water quality of downstream freshwater, estuarine, and marine ecosystems. The principal pollutants from agriculture are nitrogen, phosphorus, total suspended solids, herbicides and pesticides. A relative-risk assessment approach was used to determine potential surpluses of these. Nitrogen use varied considerably but was high for bananas and
relatively high for sugarcane. Conversely, the nitrogen surplus was low or even negative in the case of Jarrah grass. Phosphorus surplus also varied substantially but was very high for crops such as bananas and watermelon that have high phosphorous inputs and a low quantity of phosphorous in the harvestable product. Conversely, the phosphorous surplus was low or even negative for crops such as chickpea, maize and sesame. Some crops, such as bananas, have high pesticide, herbicide and fungicide application rates while other crops such as rice or mangoes have much lower application rates. Determining herbicide and pesticide surplus is difficult, especially since technologies are changing rapidly and many newer chemicals have much lower application rates, but their active ingredients are relatively more potent than older chemicals. Nitrogen losses were also examined using a different approach, through simulation. Losses via runoff were highly dependent on the amount of runoff, which is highly dependent on rainfall amount and intensity. Simulated sediment loss showed that the use of a cover crop could minimise soil loss and therefore the amount of sediment released into water bodies downstream.

**Irrigation-induced salinity**

The gently undulating plains and rises with cracking clay soils (SGG 9) developed on fine-grained sediments of the Great Artesian Basin at Wrotham Park in the centre of the Mitchell catchment show considerable potential for irrigated agriculture. However, field work indicated considerable levels of salt in this landscape where salts may mobilise and cause secondary salinity if a watertable rises close to the surface, resulting in lost crop productivity. Other soils considered to be suitable for irrigated agriculture in the Mitchell catchment were considered to have a lower salinity risk.

Under irrigation, the soils on the upper slopes of these gently undulating plains are essentially non-saline in the rooting zone and salinity only becomes apparent at depths greater than 2 m. With application of good quality water these soils can be successfully irrigated with careful management of water rates to avoid waterlogging and ponding (and possible water erosion). The sites on lower slopes, however, contain higher levels of salt and there is potential for salt to become an environmental problem and cause production losses unless carefully managed.

The watertable level depends on the initial depth to the watertable, recharge from rain and irrigation, size of the irrigation area, management practices and distance to the river. The Assessment indicates that watertable levels under small neighbouring irrigation developments (less than 500 ha in area) are not likely to interact in the next 100 years if the developments are placed at least 1 km apart. The watertable level is most likely to rise with high recharge rates and in soils with low saturated hydraulic conductivity. Proximity to rivers considerably reduces irrigation-induced rise in watertable level by increasing groundwater discharge. It may take many decades for watertable level to respond fully to irrigation development, especially if the cultivated area is large or far from the river. It is important to note, however, these watertable level and groundwater discharge results are under ‘idealised’ conditions and that the risk of secondary salinisation and watertable rise at a specific location can only be properly assessed by undertaking detailed field investigation.

**7.1.2 INTRODUCTION**

The range of environmental changes that could potentially occur as a result of water and irrigation development is as varied as the number of developments that could be proposed. Furthermore,
Water and irrigation development can result in complex and in some cases unpredictable changes to the surrounding environment and communities. For instance, prior to the construction of the Burdekin Falls Dam, the Burdekin Project Committee (1977) and Burdekin Project Ecological Study (Fleming et al., 1981) concluded that the dam would improve water quality and clarity in the lower river and that para grass (*Brachiaria mutica*), an invasive weed from Africa that was then present at relatively low levels, could become a useful ecological element as a result of increased water delivery to the floodplain. However, the Burdekin Falls Dam has remained persistently turbid since construction in 1987, greatly altering the water quality and ecological processes of the river below the dam and the many streams and wetlands into which that water is pumped on the floodplain (Burrows and Butler, 2007). Para grass and more recently hymenachne (*Hymenachne amplexicaulis*), an ecologically similar plant from South America, have become serious weeds of the floodplain wetlands, rendering innumerable wetlands unviable as habitat for most aquatic biota that formerly occurred there (Tait and Perna, 2000; Perna, 2003, 2004).

Thus, there are limitations to the specific advice that can be provided in the absence of specific development proposals and for this reason this section provides general advice on those considerations or externalities that are most strongly affected by water resource and irrigation developments. It is not possible to discuss every potential change that could occur. For this reason, the chapter is structured as follows:

- **Section 7.2, ecological implications of altered flow regimes:** examines how river regulation affects inland and freshwater assets in the Mitchell catchment and marine assets in the near-shore marine environment.
- **Section 7.3, biosecurity considerations:** discusses the risks presented by disease, pests and weeds to an irrigation development and the risks new agriculture or aquaculture enterprise in the Mitchell catchment may present to the wider industry and broader catchment.
- **Section 7.4, sediment, nutrients and agropollutant loads to receiving waters:** examines water quality resulting from agriculture and aquaculture enterprises and discusses agricultural or other chemical containment risks to downstream aquaculture enterprises.
- **Section 7.5, irrigation-induced salinity:** examines the risk of irrigation-induced salinity to an irrigation development and the downstream environment in the Mitchell catchment.

Other externalities associated with water resource and irrigation development discussed elsewhere in this report include:

- The direct impacts of the development of a large dam and reservoir on:
  - Indigenous cultural heritage (Section 3.5)
  - the movement of aquatic species (Section 5.3)
  - terrestrial ecosystems and species within the reservoir inundation area (Section 5.3).

The externalities listed above are rarely factored into the ‘true costs’ of water resource or irrigation development, and the reality is that even in parts of southern Australia where data are abundant, it is very difficult to express these ‘costs’ in monetary terms as perceived changes are strongly driven by values, which can vary considerably within and between communities and fluctuate over time. For this reason, the material in this chapter is presented as a standalone analysis to help inform conversations and decisions between communities and government.
It is important to note that this chapter is primarily focused on key risks from irrigated agriculture and aquaculture, although the section on biosecurity considers both risks to the enterprise and risks emanating from the enterprise into the broader environment. Other risks to irrigated agriculture and aquaculture are discussed elsewhere in this report and include risks associated with:

- flooding (Section 2.5)
- regulatory delays (Section 3.6)
- erosion (captured in the land suitability analysis described in Section 4.3)
- sediment infill of large dams (Section 5.3)
- reliability of water supply (sections 5.3 and 6.3)
- timing of runs of failed years on the profitability of an enterprise (Section 6.3).

Material within this chapter is largely based on (and further information can be found in) the companion technical reports on agricultural viability (Ash et al., 2018), aquaculture viability (Irvin et al., 2018), hydrogeological assessment (Taylor et al., 2018) and two companion technical reports on ecology (Pollino et al., 2018a, 2018b).

7.2 Ecological implications of altered flow regimes

7.2.1 INTRODUCTION

As outlined in Chapter 2, the Mitchell catchment is characterised by distinct wet and dry seasons, with significant variability year to year in annual rainfall. Most rivers only flow during the wet season and freshwater and marine ecosystems of northern Australia have adapted to this seasonal variability. Changes in flow as a consequence of new water resource developments have the potential to impact on these ecosystems depending upon the location, scale and nature of regulation. This section assesses the potential ecological impacts arising from changes in flow as a result of:

- major instream dams (Section 5.3)
- water harvesting (Section 5.3), where water is pumped or diverted from a major watercourse into an offstream storage, usually a ringtank.

As described in Section 5.3, major instream dams are efficient at capturing water. However, they can dramatically change the flow patterns downstream of the dam wall. With distance downstream of the dam and the point of irrigation extraction, the seasonality of streamflow may increasingly resemble the natural pattern and the amount of water extracted as a proportion of total streamflow decreases. Water harvesting typically causes less disruption to the high- and low-flow extremes than major instream dams. This is because during high-flow events mechanical pumps can only physically extract so much water, which is usually a small volume in relation to the volume of the flows. At low flows, pump-take thresholds constrain water take, where pumping only commences when the flow in the river is above a certain ‘threshold’ discharge. While a lower threshold results in an irrigator being able to extract their full allocation of water at a higher degree of reliability, it reduces the protection of low flows for the environment.
This section evaluates the potential impact of changes in flow on freshwater and marine ecological assets that may result from these two water resource development options (Scenario B). Some analyses also include assessment under wet and dry extreme climate change projections (Scenario C), and assessment of climate change projections with potential developments (Scenario D). Assets are defined as important habitats, species or functional groups that are of conservation, cultural, commercial or recreational value or that support ecological function. The outputs are intended to examine the sensitivity of freshwater and marine ecological assets to potential changes at locations within the catchment where the asset is located and to assist any future decisions on environmental flows.

Environmental flows are used to describe the quantity, timing and quality of water flows required to sustain freshwater, estuarine and coastal ecosystems. While simple ‘rules of thumb’ approaches to defining environmental flows are desirable, they have no empirical basis and can put the integrity of ecosystems at risk (Arthington et al., 2006). Contemporary methods for defining environmental flows require the development of relationships between components of flow and ecological responses (Poff et al., 2010). This report develops such relationships to evaluate the impacts of flow alteration.

Unless specified elsewhere, the material presented in Section 7.2 has been summarised from the companion technical report on ecology (Pollino et al., 2018b).

**Contextual information**

Water-dependent ecological assets are sensitive to changes in flow, being sustained by either surface water or groundwater flows or a combination of these. Priority assets have been selected in the Mitchell catchment to represent potential impacts to ecology as a consequence of surface water developments. To assess impacts, knowledge of the distribution and drivers of change of assets was synthesised and potential impacts of altered flows to assets assessed using quantitative and qualitative methods. By using these scenarios, the sensitivity of assets to changes to flow as a consequence of different types of flow changes was evaluated.

A two-tiered approach was used for analysis.

- **The first tier evaluated changes in flow throughout the catchment where assets are known to be located.** The components of the flow that are important to the asset were identified and the potential for these to change were evaluated. This was done by comparing ‘important’ components of flow (referred to as flow metrics) under Scenario B to the same components of flow under Scenario A and calculating the mean change, with the scale of change ranging from 0 and 100%.

- **A subset of these assets proceeded to a second tier of analysis that provided a more detailed assessment of potential impacts.** This was only undertaken for those assets for which there was a sufficient body of literature available to develop quantitative models (for more information see the companion technical report on ecology (Pollino et al., 2018b)).

Several other factors can also impact ecological systems such as water quality, access to groundwater, soil characteristics, physical changes (e.g. grazing impacts), fire, disease and invasive species. However, this analysis solely focuses on ecological changes that may result from changes in flow regimes. Other factors are discussed in Pollino et al. (2018a) but not evaluated.
A subset of hydrological model nodes was selected for assessment, representing locations in the Mitchell catchment. These nodes were selected based on observational records of the location of assets within the catchment, and the likelihood that the nodes would experience changes in flow (Figure 7-2).

Figure 7-2 Map of the Mitchell catchment showing the location of potential water development sites and nodes used for the ecological assessment

Scenario A incorporates historical climate and current levels of development. This is the ‘baseline’ scenario against which other scenarios were compared.

Under Scenario B, water is extracted at different locations within the catchments and the Assessment examined how the reliability of extraction is affected by a variety of pump-related variables and other uses (Hughes et al., 2018). It also involved using major dams to store water within a catchment (Petheram et al., 2017).

Under Scenario B water harvesting (Scenario B-WH), the ecological analysis considered seven whole-of-system annual extraction volumes (300, 600, 1200, 2400, 3600, 4800 and 6000 GL) at a low pump-take threshold (LT; 200 ML/day) and at a high pump-take threshold (HT; 1800 ML/day).
Under Scenario B major dams (Scenario B-D), the impacts of seven potential dams on streamflow were either calculated individually (Scenario B-D-I) or in combination (Scenario B-D-C). These potential dams are:

- the Pinnacles dam site on the Mitchell River
- the Rookwood dam site on the Walsh River
- the Palmer dam site on the Palmer River
- the Lynd downstream dam site on the Lynd River
- the Chillagoe dam site on the Walsh River
- the Elizabeth Creek dam site
- the Nullinga dam site on the Walsh River.

Potential changes under scenarios Cwet and Cdry are examined (Scenario C), as are potential changes under a projected future climate and potential development (Scenario D). Scenarios are described in Section 1.4.

Categories of impact used in the scenario analysis are:

- no change – no changes are likely to be measurable (mean change in flow metrics is less than 10%)
- minor change – minor changes that are unlikely to be measurable (mean change in flow metrics is between 10 and 30%)
- moderate change – measurable changes but without major changes to ecosystem structure or function (mean change in flow metrics is between 30 and 60%)
- major change – significant changes to ecosystem structure or function, no longer supporting habitat or species (mean change in flow metrics is between 60 and 90%)
- extreme change – complete change of ecosystem structure and function (mean change in flow metrics is greater than 90%).

The remainder of this section is structured as follows:

- Section 7.2.2 provides a summary of changes to assets using the first-tier approach at selected locations across the study area.
- These results form the basis of interpretation and discussion in sections 7.2.3 to 7.2.5.
- Combined with the second-tier approach, further results are discussed in sections 7.2.4 and 7.2.5.

### 7.2.2 Change in Components of Flow Specific to Key Ecological Assets and Habitats in the Darwin Catchments

To analyse the potential for ecological change arising from potential water resource development, relative changes in streamflow under scenarios A, B and C were assessed at six locations in the Mitchell catchment. Data from select sites, corresponding with streamflow nodes, are presented here.

The locations for which results are presented in this section are shown in Figure 7-2. Results are presented for streamflow nodes 9190000, 9190090, 9190092, 9190111, 9190030 and 9193090.
from the coast (end-of-system) to inland (Figure 7-3, Figure 7-4, Figure 7-5, Figure 7-6, Figure 7-7, Figure 7-8, respectively). The figures in this section present a summary of information on aspects of change in volume and timing of streamflow specific for each species at each node under Scenario A. The greatest changes occur at nodes immediately downstream of dams. It should be noted that the modelled changes (i.e. the results of this analysis) are based on the output of hydrological model simulations, and that the modelled flows may vary from those reported here depending upon the specific details of a development (e.g. whether water is being released directly from the dam into a channel/pipe or released down the river), particularly in river reaches immediately downstream of a dam. Consequently, these results can be considered as indicative of the likely magnitude of change and should not be used as a substitute for a detailed ecological analysis of a specific development proposal.

Each figure in Section 7.2.2 corresponds to a specific node in the river model (Hughes et al., 2018). The proximity of assets to each river model node was determined, and only assets in close proximity to a node were assessed at that node. Scenarios for water harvesting (Scenario B-WH) and dams (Scenario B-D) are presented on the x-axis and assets are listed on the y-axis. Each number represents the percentage difference between Scenario B and Scenario A, ranging from 0 to 100%. The percentage difference is calculated as the mean of change of those flow components considered important to each asset relative to Scenario A. The larger the mean percentage difference between scenarios B and A, the larger the potential for ecological change to assets. The intensity of the colour in Figure 7-3 to Figure 7-8 illustrates the magnitude of the percentage difference in flow components.
Figure 7-3 Assessment of change in flow metrics for assets at assessed location 9190000 in the Mitchell catchment

The intensity of the colour represents the variability of the scenario evaluated relative to Scenario A. Water harvesting scenarios are designated by an ‘LT’ or ‘HT’ where ‘LT’ and ‘HT’ represent low (200 ML/day) and high (1800 ML/day) extraction thresholds for water harvesting, respectively. The pump rate was assigned such that the allocated volume could be extracted in 20 days. The values in the water harvest label correspond to the total extraction in the Mitchell catchment. B-D-I denotes the results are for a single dam. B-D-C denotes the results are for a combination of dams, starting with the Pinnacles dam site on the Mitchell River and incrementally adding each dam to the right. Numbers in cells represent mean percentage change in flow metrics for an asset (between Scenarios B and A). The bar graph at the top shows mean of the mean change across all assets. This node represents end-of-system.
Figure 7-4 Assessment of change in flow metrics for assets at assessed location 9190090 in the Mitchell catchment
Refer to caption note in Figure 7-4 for figure explanation.

Figure 7-5 Assessment of change in flow metrics for assets at assessed location 9190092 in the Mitchell catchment
Refer to caption note in Figure 7-4 for figure explanation.
Figure 7-6 Assessment of change in flow metrics for assets at assessed location 9190111 in the Mitchell catchment
Refer to caption note in Figure 7-4 for figure explanation.

Figure 7-7 Assessment of change in flow metrics for assets at assessed location 9190030 in the Mitchell catchment
Refer to caption note in Figure 7-4 for figure explanation.
Figure 7-8 Assessment of change in flow metrics for assets at assessed location 9193090 in the Mitchell catchment
Refer to caption note in Figure 7-4 for figure explanation.

7.2.3 CHANGE IN MARINE ASSETS

Marine fish

Grunter and threadfin are the most sensitive species of the marine fish to changes in flow arising from potential dam and water harvesting developments, followed by mullet (Figure 7-3). Changes are greatest under the low pump-take threshold (LT) pumping options. Under Scenario B-WH, for extraction of up to and including the 3600 GL extraction volume, impacts are minor for all marine fish. The 4800 GL and 6000 GL water harvest scenarios show moderate changes to grunter and threadfin.

Under Scenario B-D-I, the potential Pinnacles dam has a minor impact on the components of flow relevant to marine fish (Figure 7-3). Other individual dams result in no change to the components of flow relevant to marine fish at the river end-of-system. Under Scenario B-D-C impacts to grunter and threadfin are moderate with the addition of the potential Rookwood, Palmer and Lynd dams and moderate for mullet with the further addition of the potential Chillagoe and Elizabeth Creek dams.

Marine fish were assessed at the end-of-system node (9190000) (Figure 7-2). This part of the Mitchell catchment represents estuarine habitats, which have high primary productivity and dynamic food webs supporting abundant and diverse fish and crustacean species (Halliday et al., 2012). The annual monsoon season is a critical time period for many fish species, including estuarine-dwelling species. The late dry season to early wet season is a critical period for recruitment and survival, with catchment flows reducing salinity in the estuary allowing for optimal growth and feeding of adults and juveniles (Humphries et al., 1999). Below is an overview of changes to grunter, mullet and threadfin salmon.
Adult grunter spawn in the estuary. They prefer brackish habitats, which are sustained by late dry-season flows. Grunter reproduce in the late dry and wet season, and the estuary habitat becomes abundant with juvenile fish. Grunter are an important predator in estuary habitats, and they structure fish populations in estuaries as a result. Changes in the volume of wet-season flows to the estuary would restrict grunter breeding and feeding habitat. Changes in late dry-season and wet-season flows could potentially affect the population of grunter in the estuary if changes occurred on an annual basis.

Mullet use both estuary and freshwater environments. Adult mullet spawn in estuaries and juvenile mullet migrate to the river during the late dry-season flows; flows are needed to reconnect the estuary and river channel for migration to occur (De Silva, 1980; Grant and Spain, 1975; Halliday and Robins, 2005; Kailola et al., 1993). Given the reduction in late dry-season flows, it is possible that the timing of connection is delayed, reducing the opportunities for mullet to migrate when flows are not at a high velocity. This could strand juveniles in the estuary in less optimal habitats for growth.

Adult threadfin salmon spawn in the estuary (Halliday and Robins, 2005) with juveniles abundant in the estuary in the late dry season (Halliday et al., 2012). Prey species for fish are also abundant in the late dry period, with prey coming downstream from the river. With reduced-size pools in the estuary in the late dry season, predation of juvenile fish is higher. Reduced food availability and increased predation could decrease the population size of juvenile fish, including the threadfin salmon.

**Snubfin dolphin**

Changes to the flow metrics most relevant to snubfin dolphin range from minor up to 3600 GL and moderate up to and including 6000 GL under Scenario B-WH LT (Figure 7-3). Under Scenario B-WH HT, impacts range from no change to minor change. Under Scenario B-D, impacts to the flow metrics relevant to snubfin dolphin from the Pinnacles dam are minor, increasing as dams are added incrementally. Impacts increase to moderate considering the cumulative changes with the addition of the Rookwood, Palmer and the Lynd dams. End-of-system components of flow relevant to snubfin dolphin do not change for individual dams, other than the Pinnacles dam, which results in a minor change.

Snubfin dolphin were assessed at the end-of-system node (9190000) (Figure 7-2). The snubfin dolphin is an estuarine- and embayment-dwelling species of conservation significance and, until recently, considered a different species from the Irrawaddy dolphin (Palmer et al., 2011). Their occupation of marine habitats is exclusively near-shore and while they are found in upper estuary freshwaters, they do not venture into riverine habitats. The dependence of snubfin dolphin on river flows and habitats in the Mitchell catchment is via the dynamics of the annual monsoon driving a complex food web. There is very limited information available on snubfin dolphins.

In the dry season, waterhole habitats in the estuary are affected. Dry-season flows are important for maintaining the quality of habitat in waterholes and for the recruitment, growth and survival of key prey species for snubfin dolphin, which recruit in estuarine and riverine habitats during September to December annually. The cumulative impacts of new developments over consecutive years will likely result in habitat loss in the estuary for snubfin dolphin. These changes are based on modelled flows and changes may vary depending upon the specific details of a development.
Crocodiles

Changes to the flow metrics relevant to crocodiles range from minor to moderate under Scenario B-WH LT, with changes being moderate from extraction volumes exceeding 4800 GL (Figure 7-3). Impacts range from no change to minor change under Scenario B-WH HT. Under Scenario B-D, impacts are minor with the addition of the potential Pinnacles dam, increasing to moderate with the addition of the potential Rookwood and Palmer dams.

Crocodiles were assessed at the end-of-system node (9190000) (Figure 7-2). Flows are critical to support breeding and nesting of crocodiles, which span the wet season. Inundated areas provide suitable habitat for crocodiles. If the size or number of inundated areas increases, it could provide an opportunity for juveniles to access a greater range of habitats. This could result in a greater food supply that could boost the growth rate of crocodiles and enable them to outcompete predators. If inundated areas decrease in size or number, it may limit the growth rate of juveniles.

Change in flow characteristics could potentially reduce the available nesting sites available to crocodiles in the wet season. Over time, reduced breeding could decrease the population size of crocodiles in the catchment. These changes are based on modelled flows and may vary depending upon the specific details of a development.

White banana prawns

Past assessments have shown the relationship between flows at the end of a river system and the catch of white banana prawns (Bayliss et al., 2014). Analysis of white banana prawn data from the Mitchell catchment indicate that catches in this catchment are likely to be minimally affected by Scenario B-WH and Scenario B-D. It found that a reduction in annual streamflow of 20% from the Mitchell catchment may reduce the median annual prawn catch across the whole Northern Prawn Fishery by about 2.5%, though the median annual reduction in some regions could be as high as 11% with percentage reductions in some low-flow years higher (see companion technical report on modelling the influence of streamflow on prawn catch, Shao et al., 2018). However, river flows are only able to partially explain the catch of white banana prawns in the Mitchell catchment. This suggests that other variables that affect the distribution, growth and survival of white banana prawns (including temperature, salinity, food availability and habitat availability) are not captured by the methods of Bayliss et al. (2014). These factors are important for the recruitment of white banana prawns and should be considered in future assessments. These changes are based on modelled flows and may vary depending upon the specific details of a development. For more information on this analysis see the companion technical report on ecology (Pollino et al., 2018b).

Salt flats

Salt flats were assessed at the end-of-system node (9190000) (Figure 7-2). At the end-of-system, flood flows expand the available habitats. Overbank floods inundate salt flats and other coastal habitats adjacent to the estuary. These are colonised by a range of fish and crustacean species. When salt flats are inundated a spike in primary productivity occurs, and this contributes to the overall dynamic production in estuaries during the monsoon season (Burford et al., 2016). Overall, changes in flow metrics relevant to salt flats range from no change to minor impacts (Figure 7-3). These changes are based on modelled flows and changes may vary depending upon the specific details of a development.
7.2.4 CHANGE IN INLAND AND FRESHWATER ASSETS: HABITATS

Floodplain wetlands

Inputs to the wetlands analysis were from the inundation modelling undertaken in the Mitchell catchment as documented in the companion technical report on flood mapping and modelling (Karim et al., 2018).

The Mitchell catchment supports some of the most ecologically diverse aquatic systems in Australia (Close et al., 2012) with a mixture of ephemeral, semi-permanent and permanent wetlands (Australian Nature Conservation Agency, 1996). Riverine flows establish connections between the river channel and surrounding wetlands. These need to be of sufficient magnitude and duration to facilitate movement and recruitment of aquatic biota between the main channel habitat and the floodplain wetlands.

An analysis of the impact of future climate projections and potential dam developments on the connectivity of 65 wetlands on the floodplain of the Mitchell catchment was undertaken. This analysis considered (i) wetlands with no connection to the main river channel; (ii) wetlands that have one or more connections to the main river channel, with a mean connection period of less than 21 days; and (iii) wetlands that had a single connection event to the main river channel that generally persisted for more than 21 days. Connectivity was analysed for low and high annual exceedance probabilities (AEP), representing low and high flood frequencies, of whether a flood is likely to be exceeded in a given year.

Under Scenario A, for flood events that occur on average once every 2 years (AEP 1 in 2), 65% of wetlands in the Mitchell catchment have no connection to the main river channel, and only 6% have extended periods of connection (>21 days) (Table 7-1). Wetlands with extended periods of connection occur low on the floodplain, which allows connection to be maintained for longer as the flood peak passes. The remaining wetlands are intermittently connected for variable periods with a mean duration of less than 21 days (Table 7-1).

Compared to Scenario A, Scenario Cwet has a greater proportion of wetlands connected intermittently, while the number of wetlands with extended periods of connection is similar (Table 7-1). Under Scenario Cdry, the proportion of wetlands with no connection to the main river channel increases, with a corresponding decrease in wetlands with intermittent connectivity. Importantly those wetlands low on the floodplain that had extended periods of connection under Scenario A are likely to become disconnected from the floodplain under Scenario Cdry (Table 7-1).

Under Scenario B-D with the potential Pinnacles dam, 91% of the wetlands have no connection to the main river channel and only 2% are connected for more than 21 days (Table 7-1). Under Scenario Dwet the percentage of wetlands connected to the main river channel is similar to that under Scenario A (Table 7-1).
Table 7-1 Percentage of wetlands connected to the main river channel in the Mitchell catchment for a flood event of AEP 1 in 2

<table>
<thead>
<tr>
<th>SCENARIO</th>
<th>% WETLANDS WITH NO CONNECTION</th>
<th>% WETLANDS CONNECTED &lt;21 DAYS</th>
<th>% WETLANDS CONNECTED &gt;21 DAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario A</td>
<td>65</td>
<td>29</td>
<td>6</td>
</tr>
<tr>
<td>Scenario B-D</td>
<td>91</td>
<td>8</td>
<td>2</td>
</tr>
<tr>
<td>Scenario Cwet</td>
<td>42</td>
<td>51</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Cdry</td>
<td>63</td>
<td>37</td>
<td>0</td>
</tr>
<tr>
<td>Scenario Dwet</td>
<td>63</td>
<td>31</td>
<td>6</td>
</tr>
<tr>
<td>Scenario Ddry</td>
<td>94</td>
<td>5</td>
<td>2</td>
</tr>
</tbody>
</table>

AEP = annual exceedance probability

Under Scenario A, for flood events that occur on average once every 10 years (AEP 1 in 10), 69% of wetlands have either intermittent connectivity (<21 day) or extended periods of connectivity (>21 days). The remaining wetlands (31%) have no connection (Table 7-2). A similar pattern of connectivity occurs under Scenario Cwet (Table 7-2). In contrast, under Scenario Cdry 63% of wetlands are not connected and 8% of wetlands have periods of connectivity longer than 21 days (Table 7-2).

Under Scenario B-D, the potential Pinnacles dam has a similar influence on wetland connectivity as Scenario Cdry with 68% of wetlands not connected to the main river channel and only 8% of wetlands having connectivity longer than 21 days (Table 7-2). Under Scenario Dwet this pattern was the same (Table 7-2). In contrast, under Scenario Ddry the percentage of wetlands no longer connected (86%) increases with losses of those wetlands connected intermittently (Table 7-2).

Table 7-2 Percentage of wetlands connected to the main river channel in the Mitchell catchment for a flood event of AEP 1 in 10

<table>
<thead>
<tr>
<th>SCENARIO</th>
<th>% WETLANDS WITH NO CONNECTION</th>
<th>% WETLANDS CONNECTED &lt;21 DAYS</th>
<th>% WETLANDS CONNECTED &gt;21 DAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario A</td>
<td>31</td>
<td>46</td>
<td>23</td>
</tr>
<tr>
<td>Scenario B-D</td>
<td>68</td>
<td>25</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Cwet</td>
<td>29</td>
<td>45</td>
<td>26</td>
</tr>
<tr>
<td>Scenario Cdry</td>
<td>63</td>
<td>29</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Dwet</td>
<td>68</td>
<td>25</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Ddry</td>
<td>86</td>
<td>6</td>
<td>8</td>
</tr>
</tbody>
</table>

AEP = annual exceedance probability

Under Scenario A, for flood events that occur once every 26 years (AEP 1 in 26), all wetlands are connected with most wetlands having intermittent connections <21 days and those wetlands lower on the floodplain having extend periods of connection (Table 7-3). Under Scenario Cwet, this pattern remains the same (Table 7-3). In contrast, under Scenario Cdry the number of intermittently connected wetlands decreases, with 31% of these wetlands no longer connected. Wetlands with extended periods of connection under Scenario A experience little change under future climate and/or development scenarios (Table 7-3).

Under Scenario B-D, the potential Pinnacles dam reduces the percentage of wetlands with an intermittent connection, with 43% of wetlands having no connectivity (Table 7-3). Under Scenario Dwet, 28% of wetlands have no connection (Table 7-3). In contrast, under Scenario Ddry, the
percentage of wetlands with no connection is 57% (Table 7-3). Importantly the number of wetlands with extended periods of connection remains relatively unchanged (Table 7-3).

**Table 7-3 Percentage of wetlands connected to the main river channel in the Mitchell catchment under a flood event of AEP 1 in 26**

<table>
<thead>
<tr>
<th>SCENARIO</th>
<th>% WETLANDS WITH NO CONNECTION</th>
<th>% WETLANDS CONNECTED &lt;21 DAYS</th>
<th>% WETLANDS CONNECTED &gt;21 DAYS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scenario A</td>
<td>0</td>
<td>92</td>
<td>8</td>
</tr>
<tr>
<td>Scenario B-D</td>
<td>43</td>
<td>49</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Cwet</td>
<td>0</td>
<td>92</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Cdry</td>
<td>31</td>
<td>63</td>
<td>6</td>
</tr>
<tr>
<td>Scenario Dwet</td>
<td>28</td>
<td>65</td>
<td>8</td>
</tr>
<tr>
<td>Scenario Ddry</td>
<td>57</td>
<td>37</td>
<td>6</td>
</tr>
</tbody>
</table>

AEP = annual exceedance probability

The potential Pinnacles dam has the potential to alter wetlands becoming disconnected and permanently dry. The disconnection of wetlands from the main river channel under Scenario B-D (potential Pinnacles dam) is amplified under the Scenario Ddry but is less pronounced under Scenario Dwet.

Under small, moderate and large flood events, wetlands that are closest to the river remain connected and wetlands further from the river become permanently disconnected and remain dry (Walker and Thoms, 1993; Ward and Stanford, 1995). Increasing dryness is likely to lead to loss of aquatic biota and result in a more terrestrial landscape. If the period a wetland remains dry exceeds 10 years, there will be a substantial loss of resilience with many biota adapted to periodic wetting and drying by surviving as dormant propagules becoming lost (Nielsen et al., 2013).

**Inchannel waterholes**

The Mitchell catchment is highly seasonal with high wet-season and low dry-season flows. During the dry season, waterholes are important habitat that provide refuge for aquatic species and sources of water for other flora and fauna. As the dry season progresses, a reduction in the total area of waterholes occurs with the loss of smaller waterholes (Close et al., 2012; Pollino et al., 2018b) with many aquatic species, including a high diversity of fish species, turtles and sawfish, finding refuge within these waterholes. While waterholes have a relatively small contribution by area, it is from these refuge habitats that recolonisation of the river habitat occurs during the wet season (Lymburner and Burrows, 2008).

Maintaining the quality of waterhole habitats during periods of low flow is crucial for the local persistence of many of aquatic species (Department of Environment and Resource Management, 2010). Lower dry-season flows resulting in longer periods of low flows due to water resource development threaten to reduce the habitat value of waterholes. This can occur due to the loss of waterholes within the landscape and also decreases in the habitat quality and condition of the waterholes that remain (Department of Environment and Resource Management, 2010). In highly seasonal catchments such as the Mitchell catchment, increases in flow or longer water persistence due to dam releases or weirs during the dry season can also have impacts on waterholes as the fauna of the region are adapted to the annual cycle of low-flow and high-flow periods. Over the dry season, flow pulses within a river, groundwater input and local rainfall can replenish waterholes.
Modification of the duration or timing of low-flow or cease-to-flow periods threatens to change the ecological character of waterholes in the Mitchell catchment. During cease-to-flow events, where no surface water enters waterholes, species lose migration and movement pathways, and water quality often deteriorates. During periods of low flow, waterhole area is reduced, resulting in the loss of important ‘glide’ and riffle habitat, or potential loss of entire waterholes. The location of individual waterholes within the catchment is an important contributing factor to the duration of the cease-to-flow period, with waterholes in the upper Mitchell River more likely to undergo prolonged periods of disconnection under current conditions.

In the Mitchell catchment, changes to waterholes by water harvesting (Scenario B-WH) and dams (Scenario B-D) are considered. Water harvesting is found to reduce flows at downstream nodes, although there are no changes in the mean number of cease-to-flow days across scenarios. The impact to low flows, and hence waterholes, is minimised by the high pump-take threshold (HT), with minimal impact to low-flow periods.

Although water harvesting causes minimal changes to the duration of the cease-to-flow periods, it was associated with a small reduction in the total area of waterholes, due to a reduction in dry-season flows (Table 7-4). This would result in some loss of important glide or riffle habitat on the edge of or between waterholes and in some locations where the disconnection from habitat structure occurs, such as bank overhangs and tree roots. This impact is reduced under Scenario B-WH HT. Limiting water harvesting leading up to and during periods of low flow and maintaining dry-season pulses and first wet-season flows would provide protection to waterholes within the Mitchell catchment. The water harvest volume and pump rate (Section 5.3) has little influence on waterhole area in the Mitchell catchment.

Table 7-4 Waterhole area by scenario, shown as mean minimum dry-season waterhole area
Node 9190111 is located just downstream of water harvesting and node 9190000 is located near the river mouth.

<table>
<thead>
<tr>
<th>NODE</th>
<th>UNIT</th>
<th>SCENARIO A</th>
<th>SCENARIO B-WH LT</th>
<th>SCENARIO B-WH HT</th>
<th>SCENARIO Cdry</th>
<th>SCENARIO Cmid</th>
<th>SCENARIO Cwet</th>
</tr>
</thead>
<tbody>
<tr>
<td>9190111</td>
<td>Area (ha)</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
<td>0.6</td>
<td>0.8</td>
<td>1.3</td>
</tr>
<tr>
<td></td>
<td>% change</td>
<td>0</td>
<td>−1.9%</td>
<td>0.0%</td>
<td>−41.2%</td>
<td>−26.4%</td>
<td>28.2%</td>
</tr>
<tr>
<td>9190000</td>
<td>Area (ha)</td>
<td>39.9</td>
<td>39.1</td>
<td>39.9</td>
<td>37.1</td>
<td>38.0</td>
<td>41.8</td>
</tr>
<tr>
<td></td>
<td>% change</td>
<td>0</td>
<td>−2.0%</td>
<td>0.0%</td>
<td>−7.0%</td>
<td>−4.7%</td>
<td>4.9%</td>
</tr>
</tbody>
</table>

Potential dams (Scenario B-D) in the Mitchell catchment result in substantial changes in cease-to-flow days, with increases from no cease-to-flow days to a mean of 230 days/year (Table 7-5). Such changes would likely have severe impacts on waterhole ecology, including affecting migration and movement pathways in affected reaches, and have impacts on water quality and other habitat characteristics. In the cumulative dam assessment, changes to cease-to-flow periods in the Mitchell catchment are greatest in reaches close to the upstream dams (e.g. node 9190111) and have comparatively reduced but noticeable effects further downstream towards the river mouth (e.g. node 9190091). The end-of-system reach is unaffected by cease-to-flow periods from dams (Table 7-5) but is still affected by changes to important low flows during the seasonal dry period.
Table 7-5 Annual mean number of cease-to-flow days at selected nodes in the Mitchell catchment
Node 9190111 is located just downstream of a dam and node 9190000 is located near the river mouth.

<table>
<thead>
<tr>
<th>NODE</th>
<th>SCENARIO A</th>
<th>PINNACLES</th>
<th>ROOKWOOD</th>
<th>PALMER</th>
<th>LYND D/S</th>
<th>CHILLAGOE</th>
<th>ELIZABETH CK</th>
<th>NULLINGA</th>
</tr>
</thead>
<tbody>
<tr>
<td>9190111</td>
<td>0</td>
<td>232</td>
<td>232</td>
<td>232</td>
<td>232</td>
<td>232</td>
<td>232</td>
<td>232</td>
</tr>
<tr>
<td>9190000</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>

The cumulative dam scenario (Scenario B-D-C) results in the greatest impact on waterholes with dry-season waterhole area severely reduced, with impacts extending downstream. At the most downstream reach (9190000) the dry-season waterhole area is reduced by over 65% by the initial dam and by over 75% when all seven dams are considered (Table 7-6). Such large losses of habitat area would result in substantial impact to plants and animals that depend upon waterholes as refuge habitat during the annual dry period.

Table 7-6 Waterhole area at the end-of-system reach by cumulative dams shown as annual mean dry-season minimum waterhole extent

<table>
<thead>
<tr>
<th>NODE</th>
<th>UNIT</th>
<th>SCENARIO A</th>
<th>PINNACLES</th>
<th>ROOKWOOD</th>
<th>PALMER</th>
<th>LYND D/S</th>
<th>CHILLAGOE</th>
<th>ELIZABETH CK</th>
<th>NULLINGA</th>
</tr>
</thead>
<tbody>
<tr>
<td>9190000</td>
<td>Area (ha)</td>
<td>39.9</td>
<td>12.9</td>
<td>12.0</td>
<td>10.9</td>
<td>10.1</td>
<td>10.1</td>
<td>9.8</td>
<td>9.8</td>
</tr>
<tr>
<td></td>
<td>% change</td>
<td>0</td>
<td>–67.7%</td>
<td>–69.9%</td>
<td>–72.6%</td>
<td>–74.6%</td>
<td>–74.6%</td>
<td>–75.5%</td>
<td>–75.5%</td>
</tr>
</tbody>
</table>

Riparian vegetation zones

Riparian vegetation is adjacent to watercourses. Riparian zones are highly diverse, dynamic and complex, and because they depend on flood processes they are highly vulnerable to disturbances. 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., 2016). The inputs to this analysis are from inundation modelling within the Mitchell catchment (Karim et al., 2018).

A major riparian vegetation type in the Mitchell catchment is the ‘Melaleuca species woodland-open forest on sands in channels and on levees’ which is dominated by *Melaleuca leucadendra* and/or *Melaleuca argentea* fringing forests and woodlands and *Eucalyptus camaldulensis* (regional ecosystem (RE) class 2.3.24). It has a biodiversity status listed as ‘of concern’ and is a valuable refuge for some fauna and flora, including providing habitat for sawfish (Queensland Government, 2017). This RE class is subject to invasion by rubber vine (*Cryptostegia grandiflora*) and water hyacinth (*Eichhornia crassipes*) and grazing pressure (Queensland Government, 2017).

The vegetation RE class 2.3.24 was selected as representative of riparian vegetation to determine the likely impact of each of the scenarios by analysing inundation patterns. The larger floods (Figure 7-9c) show a smaller impact between the scenarios compared to the smaller floods (Figure 7-9a). The Scenario Ddry shows the largest decrease in area inundated from Scenario A for floods of all magnitudes, while Scenario Cwet shows an increase in inundated vegetation area for the smaller floods. The development scenarios (scenarios B, Ddry and Dwet) generally show a decrease in the inundated area compared to Scenario A (except for ‘Scenario Cwet’ for the smaller flood; Figure 7-9).

The alteration of the flow regime caused by development and/or a future drier climate, which in turn leads to a reduced area of inundation, will likely increase the success of invasive species, including rubber vine and water hyacinth, identified as threats to this RE class.
7.2.5 CHANGE IN INLAND AND FRESHWATER ASSETS: FISH

Stable flow spawners

Stable flow spawners are a functional group of fish that spawn in the dry season, in association with stable flows (low flow, baseflow and cease-to-flow). The species in the group include the freshwater longtom (*Strongylura krefftii*); mouth almighty (*Glossamia aprion*); bony herring (*Nematalosa erebi*); barred grunter (*Amniataba percoidei*); flyspecked (*Craterocephalus stercusmuscarum stercusmuscarum*) and freckleheaded (*Craterocephalus lentiginosus*) hardyhead; and the eastern (*Melanotaenia splendida splendida*), chequered (*Melanotaenia splendida inornata*) and western (*Melanotaenia australis*) rainbowfish. Stable flow spawners were assessed at nodes 9190090, 9190092, 9190111, 9193090 and 9190030 (Figure 7-2). Stable flow spawners are an important fish species group in river ecosystems. They are typically small and sedentary,
and prey for a range of larger species, such as barramundi. Despite stable flow spawners not requiring high flows to initiate spawning, recruits do benefit from exploiting the nutrient-rich floodplains during the wet season and they require flows for habitat maintenance and condition.

A reduction in the extent of inundation would increase competition between species because of the decrease of available resources and space. Overbank floods in the wet season provide dispersal routes for fish species to migrate to exploit the nutrient-rich inundated floodplains (Karim et al., 2012). Wet-season floods also inundate important offchannel habitats that function as refugia habitat in the dry season (Karim et al., 2012; King et al., 2015; Pusey et al., 2004). Changes in ‘condition’ are described as changes in the area, quality and persistence of habitat for species. Changes in habitat condition can affect recruitment rates, dispersal and range, body condition and persistence of stable flow spawners.

Tier 1 screening analysis indicates that under Scenario B-WH, for important flow metrics to stable flow spawners at node 9190090 there is no change for extraction volumes up to 1200 GL and minor change for extraction volumes greater than 1200 GL. At 9193090, for the LT, changes are minor for extraction volumes between 2400 and 6000 GL, and for the HT are moderate for extraction volumes between 3600 and 6000 GL. Changes in flow of concern are the number of zero-flow days, timing of minimum flows and duration of overbank flows.

Under Scenario B-D-C, for the combined potential Pinnacles, Rookwood, Palmer and Lynd dams, the components of flow relevant to stable flow spawners at nodes 9190090 and 9190092 show minor changes. No change is observed for any of the individual dams (Scenario B-D-I). Under Scenario B-D-I, for the Pinnacles dam the components of flow at node 9190111 show only minor changes. Under Scenario B-D-I at node 9193090, there are major changes in components of flow relevant to stable flow spawners, with the potential Rookwood dam on the Walsh River. The changes are extreme under Scenario B-D-C, which includes the potential Rookwood, Chillagoe and Elizabeth Creek dams, which would cause the habitat for stable flow spawners to no longer be suitable. Under Scenario B-D-I for the potential Pinnacles dam, the components of flow relevant to stable flow spawners at node 9190030 show moderate change.

Further analysis (Tier 2) was undertaken at nodes 9190090, 9190110 and 9193090 for water harvest scenarios and 9190090, 9190092, 9190111 and 9190030 (Figure 7-2) for dam scenarios, given the frequency of mapped occurrences of stable flow spawners in the Mitchell catchment. This analysis shows a substantial decline in condition of the stable flow spawners, longtom and barred grunter as a consequence of Scenario B-WH. Increasing volumes of flow extraction in Scenario B-WH result in the greatest change. Scenario B-WH for the HT shows little change for longtom and barred grunter. The mouth almighty, hardyheads and rainbowfish also show a loss in condition with increased flow extractions, although the response was not as dramatic.

Stable flow spawners show some decline under select dam scenarios, with changes being greater than under Scenario B-WH or Scenario C. This is evident at node 9190030, immediately downstream of the potential Pinnacles dam. At node 9190111, which is downstream of node 9190030, there is a decline in habitat for the potential Pinnacles and Rookwood dams.

Changes downstream of dams would negatively affect the habitat maintenance and persistence of stable flow spawners, altering the connection and disconnection periods of offchannel and main channel habitat. A reduced connection period could hinder the ability of stable flow spawners to migrate to suitable dry-season refugia habitat. Stable flow spawners would remain in main
channel habitats where competition and predation effects would be greater. In contrast, an increased connection period (permanency of water in a previously ephemeral reach) could facilitate the dispersal of species that would normally not occupy offchannel habitats. This could alter fish assemblage structures and increase competition and predation.

### Migratory fish

A fish group vulnerable to inchannel barriers and changes to flows is freshwater migratory fishes. Migratory fish are distributed throughout the Mitchell catchment. While there are many species in this group, barramundi (*Lates calcarifer*), bull shark (*Carcharhinus leucas*), black catfish (*Neosilurus ater*) and Hyrtl’s tandan (*Neosilurus hyrtlii*), sooty grunter (*Hephaestus fugilinosus* and *H. jenkinsi*), freshwater longtom (*Strongylura kreffti*) and spangled perch (*Leiopotherapon unicolor*) are used here for analysis.

Migratory fish were assessed at nodes 9190090, 9190092, 9190111, 9193090 and 9190030 (Figure 7-2). Tier 1 screening analysis found that under Scenario B-WH some changes in habitat for migratory fish occur. Changes in important flow metrics for migratory fish at node 9190090 are minor up to and including extraction volumes of 1200 GL for the LT, becoming moderate up to 3600 GL and major at 6000 GL. Changes are minor, becoming moderate at extraction volumes greater than 3600 GL for the HT. Changes in important flow metrics range from no change to minor changes at nodes 9190092, 9190111 and 9190030. Changes at node 9193090 are minor between 600 GL to 2400 GL for the LT and moderate in remaining extraction volumes. Changes range from no change to minor change for the HT.

Under Scenario B-D-I at node 9190090, minor changes to habitat of migratory fish occur with the introduction of Pinnacles dam, increasing to moderate changes under Scenario B-D-C with the addition of the Rookwood dam on the Walsh River and for the remaining potential dams. Other potential dams when assessed individually (i.e. Scenario B-D-I) result in no change to important flow metrics for migratory fish at this location. At node 9190092, under Scenario B-D-I there are moderate changes in flow metrics with the introduction of a potential dam on the Lynd River (Figure 7-5).

Under Scenario B-D-I at node 9190111, major changes to flow metrics occur with the introduction of the potential Pinnacles dam (slightly upstream of Pinnacles dam site). Major changes would have substantial impacts on the habitat for migratory fish. Immediately below the potential Pinnacles dam (i.e. below the spillway), at node 9190030, extreme changes in flow metrics relevant to migratory fish are observed, which would result in the habitat no longer being suitable for migratory fish. However, it should be noted that under this particular model configuration it was assumed that all water released from the dam was released into a channel not down the river. Had water from the dam been released down the river the impact would be moderated to some extent.

### Detailed flow analysis

Further analysis (Tier 2) was undertaken at nodes 9190090, 9190110 and 9193090 under Scenario B-WH and 9190090, 9190092, 9190111 and 9190030 under Scenario B-D. Analysis shows a decline in habitat of migratory fish. Increasing volumes of flow extraction under Scenario B-WH show increasing change in habitat with increased volumes of water extracted. Under Scenario B-D, the introduction of the potential Pinnacles dam results in a decline in habitat for freshwater fish.
species Hyrtl’s tandan, black catfish, sooty grunter and spangled perch at nodes 9190030 and 9190111.

Dams compromise the movement of fishes, preventing access to distant or adjacent habitats, and their breeding cues (Roscoe and Hinch, 2010). Migratory fish are actively moving on the decreasing flows of the wet season and a subset would be using early wet-season flows to spawn and recruit. The potential dams in the Mitchell catchment will have localised impacts by limiting movement within the river channel. These changes could influence the structure of fish communities.

The location corresponding to node 9190090 is thought to be particularly important for migratory fish, as it connects to the floodplain and estuarine habitats. With the cumulative addition of dams, there is potential for extensive changes in habitats for migratory fish. At the end-of-system node (9190000) there are major shifts relevant to large-bodied species, such as sawfish, barramundi and bullsharks.

Overbank floods in the wet season provide dispersal routes for fish species to migrate and exploit the nutrient-rich inundated floodplains (Karim et al., 2012). Reduced dry-season flows may limit the movement of migratory fish into high primary-productivity estuarine habitats that support productive food webs and high growth rates as river connectivity is required to access these habitats (King et al., 2015). Reduced duration of high flows would also restrict the migration distance and window for migratory fish. In addition to changes in habitat, cues and productivity, dams are also a physical barrier. This would reduce the movement of migratory fish through the catchment.

**Barramundi**

The barramundi is a large fish that occurs throughout northern Australia in rivers, lagoons, swamps and estuaries and is arguably the most important fish species to cultural, recreational and commercial fisheries. Barramundi is found extensively throughout the Mitchell catchment. Barramundi has the potential to be affected by barriers to movement and changes in the flow regime. Spawning occurs in estuaries, juveniles migrate up rivers and the first few years of a barramundi’s life are spent in freshwater habitats. In the dry season, barramundi use permanent waterholes as a refuge.

Barramundi were assessed at six nodes: 9190000, 9190090, 9190092, 9190111, 9193090 and 9190030 (Figure 7-2). Tier 1 screening analysis indicates that Scenario B-WH results in some changes to the habitat for barramundi. Changes in flow metrics of importance to barramundi at the end-of-system node (9190000) range from no changes to minor changes. Changes at node 9190090 are minor up to and including 1200 GL, becoming moderate up to 6000 GL for the LT. Changes are moderate between 600 and 2400 GL, becoming moderate for remaining extraction volumes under the HT. Changes range from no change to minor changes at nodes 9190092, 9190111 and 9190030. Changes at node 9193090 are minor up to 3600 GL and moderate in remaining scenarios for the LT, with changes ranging from no change to minor for the HT.

Under Scenario B-D-C at the end-of-system node (9190000), changes are minor up to the addition of the fourth potential dam, the Lynd dam. The remaining potential dams result in a moderate change. Under Scenario B-D-I, Pinnacles dam results in a minor change to flow metrics of importance to barramundi at the end-of-system.
Under Scenario B-D-I at node 9190090, minor changes to the habitat of barramundi occur with the introduction of the potential Pinnacles dam, increasing to moderate under Scenario B-D-C with the addition of the Rookwood dam on the Walsh River and the remaining dams. At node 9190092, moderate changes occur with the introduction of the potential dam on the Lynd River.

Under Scenario B-D-I at node 9190111, major changes to flow metrics important to barramundi occur with the introduction of the potential Pinnacles dam. Under scenario B-D-I and B-D-C at node 9193090, moderate changes to flow metrics occur with the introduction of the Rookwood dam on the Walsh River, increasing to major changes with the addition of the Elizabeth Creek dam. Major changes would have substantial impacts on the habitat for barramundi.

Immediately below the potential Pinnacles dam (i.e. below the spillway), at node 9190030, extreme changes in flow metrics relevant to barramundi occur, which would result in the habitat no longer being suitable for barramundi. However, it should be noted that under this particular model configuration it was assumed that all water released from the dam was released for irrigation into a channel not run down the river. Had water from the dam been released down the river the impact would be moderated to some extent.

**Detailed flow analysis**

Further analysis (Tier 2) was undertaken at nodes 9190090, 9190110 and 9193090 under Scenario B-WH and 9190090, 9190092, 9190111 and 9190030 under Scenario B-D. The decrease in the condition scores of barramundi with increasing water extraction indicates the potential to negatively affect the populations of barramundi over time (Figure 7-10). The introduction of the Pinnacles dam in the Mitchell catchment results in a substantial modelled decline in habitat at nodes 9190030 and 9190111 (Figure 7-11). However, it should be noted that under this particular model configuration it was assumed that all water released from the dam was released for irrigation into a channel not run down the river. Had water from the dam been released down the river the impact would be moderated to some extent.

Water harvest scenarios have the potential to reduce dry-season flows and the duration of wet-season flows. Dam scenarios increase the duration of low-flow periods in the dry season. In the dry season, adult barramundi in near-shore marine habitats spawn and the juveniles migrate to estuaries and freshwater river reaches where they forage and grow for several years. Barramundi spawned during the late dry season move upstream to inchannel waterhole habitats as juveniles. Connectivity via river flows is required to access these habitats. Late dry-season flows are important to reconnect the estuary to the river channel and allow juvenile barramundi to move to riverine habitats. For inland juvenile barramundi, the loss of freshwater habitats may represent a critical bottleneck for barramundi populations in the catchment. Decreases in the extent, depth and productivity of waterholes would reduce the extent of dry-season habitat and food abundance. In addition to changes in habitat, cues and productivity, dams are also a physical barrier. Flow changes as a consequence of the potential dams are likely to have an impact on barramundi, with changes throughout the year, impacting on habitat. This is more evident in nodes close to potential dams. Dams compromise the movement of barramundi, preventing access to distant or adjacent habitats (Roscoe and Hinch, 2010).
Figure 7-10 Maximum condition scores of barramundi, considering Scenario B-WH at nodes 9190090, 9190110 and 9193090, showing LT and HT scenarios.
Figure 7-11 Maximum condition scores of barramundi, considering Scenario B-D-C at nodes 9190090, 9190092, 9190111 and 9190030

Sawfish

The freshwater sawfish (Pristis pristis) is distributed throughout the Mitchell catchment. Inchannel barriers along migration routes of the Mitchell catchment pose a threat to the freshwater sawfish. The freshwater sawfish has a marine adult phase while the juvenile phase is in estuaries and rivers, and juveniles and adults occupy large pools and waterholes of large rivers.

Sawfish were assessed at five nodes in the Mitchell catchment (nodes 9190000, 9190090, 9190092, 9190111 and 9193090) (Figure 7-2). Tier 1 screening analysis in the Mitchell catchment
under Scenario B-WH results in some changes in habitat for sawfish. Changes to important flow metrics for sawfish at the end-of-system node (9190000) are minor up to and including an extraction volume of 3600 GL, increasing to moderate up to 6000 GL for the LT. Changes range from no change to minor change at the HT. Changes at nodes 9190090 are minor up to and including 600 GL, becoming moderate up to 4800 GL and becoming major at 6000 GL for the LT. For the HT, there are minor changes for 2400 and 3600 GL, ranging up to moderate changes for 4800 and 6000 GL. At 9110092, changes range between no change and minor change for the LT and HT. Changes range between no change and minor change at nodes 9190111 and 9190030. Changes at node 9195090 are minor up to 3600 GL and moderate in remaining scenarios.

Under Scenario B-D-I, the potential Pinnacles dam results in minor changes at nodes 9190000 (end-of-system) and 9190090. Under Scenario B-D-C these increase to moderate with the addition of the potential Rookwood dam on the Walsh River (Figure 7-2). Under Scenario B-D-I the introduction of individual dams other than Pinnacles dam results in no change to important flow metrics for sawfish at nodes 9190000 and 919090. Changes at node 9190092 are moderate with the introduction of the Lynd River dam.

Under Scenario B-D-I at node 9190111 (i.e. in the reach below the potential Pinnacles dam), major changes occur with the introduction of the potential Pinnacles dam. At node 9195090 major changes occur with the addition of the Rookwood dam on the Walsh River, increasing to extreme changes with the addition of the Elizabeth Creek dam. With the introduction of the Pinnacles dam at node 9190030, extreme changes occur immediately downstream. Major changes would have substantial impacts on the habitat for sawfish. Extreme changes would result in the habitat for sawfish no longer being suitable. However, it should be noted that under this particular model configuration it was assumed that all water released from the dam was released for irrigation into a channel not run down the river. Had water from the dam been released down the river the impact would be moderated to some extent.

**Detailed flow analysis**

Further analysis (Tier 2) was undertaken at nodes 9190090, 9190110 and 9195090 under Scenario B-WH and 9190090, 9190092, 9190111 and 9190030 (Figure 7-2) under Scenario B-D. Under Scenario B-WH at the lower-volume extractions there are minimal changes to the condition of sawfish. In contrast, high extraction would potentially have a substantial impact on the condition of sawfish. The decrease in condition for sawfish is proportional to the annual flow extracted (Figure 7-12). These changes have the potential to reduce sawfish populations over time.

Under Scenario B-D, there are substantial changes at nodes 9190111 and 9190030, with minimal changes at other assessment nodes (Figure 7-13).

Water harvest scenarios and potential dam scenarios led to reduced dry-season flows. Dam scenarios also increased the duration of low-flow periods. In the dry season, juvenile sawfish use inchannel waterholes in the lower to mid-reaches. With reduced flows, inchannel waterholes decrease in extent and depth, reducing habitat diversity and prey abundance within the waterholes. Inchannel waterholes act as refuge habitats that sustain freshwater sawfish during the dry season. Sawfish rely on the perennial nature and diversity of the instream pool habitats to survive, shelter and forage. Waterholes that have both deep-water pools and shallow bank and glide habitats are preferred as they support sheltering and feeding. In the late dry season, the water quality in waterholes declines and they are replenished by late dry-season flows that are
also critical for maintaining habitat and production in estuaries. These late dry-season flows are at risk of change in the scenarios.

Under dam scenarios, flow changes also occur for wet-season flows. This has the potential to reduce sawfish reproduction and migration. Sawfish juveniles would be limited in their ability to migrate upstream to freshwater channel–pool and billabong habitats. Movement of sawfish downstream may be reduced due to decreased flood duration.

Figure 7-12 Maximum condition scores for sawfish, considering Scenario B-WH at nodes 9190090, 9190110 and 9193090
Figure 7.13 Maximum condition scores for sawfish, considering Scenario B-D-C at nodes 9190090, 9190092, 9190111 and 9190030.


7.3 Biosecurity considerations

7.3.1 INTRODUCTION

Diseases and pests, due either to pathogens endemic to farming regions or introduced through translocations of animals from other regions, have had, and continue to have, major impacts on global agriculture and aquaculture production. For example, the resultant economic loss to global aquaculture industries alone was estimated to be US$6 billion/year (World Bank, 2014).

Compared with many other countries, Australia has many advantages in terms of the opportunity to mitigate risks from pests and disease through sound regulatory processes controlling translocation, technological knowledge and capability, and the greater geographic spread between farming operations in many regions. However, the recent discovery of the highly pathogenic white spot syndrome virus (WSSV) in south-east Queensland prawn farms has had a devastating effect on parts of the industry. There have also been serious outbreaks of diseases in agricultural crops in recent years across northern Australia, including green cucumber mottle mosaic virus on melons in the Katherine region in the NT; the fungal rice blast affecting rice production in the Ord, WA; Panama disease tropical race 4 affecting bananas in northern Queensland and the NT; and banana freckle in the NT. These serve as contemporary reminders of the impact that a pathogen can have at the point of infection as well as on an entire industry.

From a biosecurity point of view, risk is defined as the product of the likelihood of an invasion by a pest or pathogen and the impact that species will have. With both likelihood and impact there is a great deal of uncertainty and difficulty in making clear predictions.

This section examines biosecurity risk from an agriculture and aquaculture perspective, with the discussion structured around:

- the biosecurity risk to new farming enterprises in the Mitchell catchment
- the biosecurity risk that new farming enterprises in the Mitchell catchment may present to the broader industry across Australia.

Agriculture and aquaculture production systems can be threatened by generalist or specialised pests. The relative isolation of small areas of agriculture and aquaculture may, under some circumstances, provide some protection from certain pests and diseases but the opposite may also be the case; extensive areas of natural or less intensively managed vegetation may provide refuge for pests and diseases as well as beneficial organisms. For example, the Mitchell catchment is far removed from the WSSV outbreaks in prawn production systems in south-east Queensland but such isolation does not preclude the very real prospect of a biosecurity risk should such production systems be established there.

Generally, both dryland and irrigated cropping systems have relatively well-developed pest management protocols and the economics of such systems is such that they can bear the cost of controlling the pests that are of concern to them. This is especially the case for high-value crops. Less-intensive agricultural industries and environmental interests are likely to be in a less-favourable economic position when it comes to pest management. Irrigation and other intensive agricultural industries can thus increase the risks from pests faced by less-intensive industries and the natural environment.
Unless stated otherwise, material in the agriculture biosecurity (Section 7.3.2) and aquaculture biosecurity (Section 7.3.3) sections have been summarised from the companion technical reports on agricultural viability (Ash et al., 2018) and aquaculture viability (Irvin et al., 2018), respectively.

7.3.2 AGRICULTURAL BIOSECURITY

Irrigated agriculture in the Mitchell catchment will be exposed to existing and new diseases, pests and weeds. Although a tropical environment is conducive to a wide range of diseases, pests and weeds, the long dry season and the loss of green vegetation that characterises the Mitchell catchment provide an unfavourable environment for many diseases, pests and weeds and act as a natural break to their year-round persistence. However, areas of irrigation with year-round green foliage may increase the risk of insect pests and diseases persisting throughout the year. Further, new incursions are more likely because of increased human activity and the transport of new vectors on equipment, people or through seeds.

In the short to medium term, biosecurity risks to the Mitchell catchment are most likely to come from within the country and, in particular, from adjacent or climatically similar parts of Australia. The catchment already experiences impacts from introduced plants that are widespread in northern Australia. Not all parts of the catchment are equally vulnerable. Important pathways for dispersal of invasive species include existing road and river corridors. In northern Australia, riparian zones are vulnerable to invasion by many different non-native plants because these zones are relatively moist, have generally higher nutrient levels and experience naturally high levels of disturbance.

New risks will mostly be associated with insect pests and diseases. Understanding and managing this increased risk will be important. For this catchment, the types of pests and diseases that provide a risk to the various agricultural industries that could be established are many and varied. Annual weeds are a risk to crop production but they can generally be controlled through herbicides, use of cover crops and stubbles, and cultivation. They tend to not be an acute problem like pests and diseases and thus represent an ongoing management challenge rather than a threat to viability. For example, ongoing weed management in sugarcane costs around $338/ha.

Identifying potential pest and disease problems that can occur in greenfield agricultural areas can be problematic. The warmer, north Australian environment is more favourable than temperate climates for insects and pathogens to adapt and multiply with the introduction of a new food source (i.e. a crop). However, the environment also favours beneficial organisms that prey on pest species. Production systems that recognise the ecological realities of the natural environment are recommended; the collapse of the cotton industry in the Ord in WA during the 1970s is one example of a failure to do this. Irrigating a number of crops each year in rotation can provide a year-round food source for pests and carry-over of pathogens between crops.

**Biosecurity risks to new agricultural enterprises and the risks from these enterprises to the broader environment**

This discussion examines pathways of entry to the Mitchell catchment and then specifically discusses disease, pests and weeds in turn.
**Pathways of entry**

Pathogens, pests and weeds can enter a catchment via man-made pathways or natural pathways. Man-made pathways include road transport, ships and planes, and the ‘carriers’ (e.g. humans, animals, plants, machinery) that facilitate the movement and incursion of new pests, diseases and weeds. Published work has shown that the most likely human-facilitated pathway for bringing invasive species is either general trade, or live plant or animal trades. While it is not currently possible to determine the actual or relative risk to the Mitchell catchment from these forms of human-facilitated trade, it is possible to look at the historical rate of invasions (also referred to as incursions) in Australia. Studies have shown that the incursion rate in Australia for four orders of insects (beetles, bugs, flies, and moths and butterflies) has been about 15 species per year. Given the low human population in the Mitchell catchment it is not likely that this catchment will be exposed to regular incursions facilitated by humans. However, they do occur and can be very damaging, for example the green cucumber mottle mosaic virus in the NT.

Natural pathways include wind dispersal and river flow. Of these two modes, river flow is probably the higher risk as the headwaters of the Mitchell catchment are relatively densely populated and intensively farmed.

Another mode of entry to the Mitchell catchment is from pathogens and insects that arrive via the wind from Papua New Guinea or South-East Asia. Wind dispersal modelling undertaken in this Assessment shows that there is no risk of a fungal pathogen or insect pest landing in the Mitchell catchment from Sumatra, Java, or Bali (Table 7-7). The greatest risk of a fungal pathogen arriving to this region is from Papua, with Papua New Guinea and Timor-Leste presenting a lower risk. For insect pests, the only threat is from the coastal area of Papua. It should also be noted that these risks are only present for part of the year.

From mid-April to the beginning of September, the prevailing winds are generally southerlies through to easterlies (i.e. either blowing offshore or back towards Papua New Guinea). As a result, there are no arrivals from any points in South-East Asia to the Mitchell catchment during this period. In summary, the risk of arrival of fungal pathogens and insect pests directly from South-East Asia into the Mitchell catchment is small and restricted to a relatively narrow period.

**Table 7-7 Proportion of simulations (1 per week = 52), in which a fungal pathogen or insect pest could have been transported from a location in South-East Asia to the Mitchell catchment**

<table>
<thead>
<tr>
<th>LOCATION</th>
<th>FUNGAL PATHGEN</th>
<th>INSECT PEST</th>
</tr>
</thead>
<tbody>
<tr>
<td>PNG (south coast)</td>
<td>0.04</td>
<td>0.00</td>
</tr>
<tr>
<td>PNG (mid-and north coast)</td>
<td>0.06</td>
<td>0.00</td>
</tr>
<tr>
<td>Indonesia (Papua – south coast)</td>
<td>0.15</td>
<td>0.02</td>
</tr>
<tr>
<td>Indonesia (Papua – mid- and north coast)</td>
<td>0.08</td>
<td>0.00</td>
</tr>
<tr>
<td>Indonesia (Java and Bali)</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Indonesia (Sumatra)</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Timor-Leste</td>
<td>0.06</td>
<td>0.00</td>
</tr>
</tbody>
</table>

PNG = Papua New Guinea

Threats to agriculture in the Mitchell catchment are not just from new arrivals from overseas but also from human-mediated dispersal from other areas within Australia. There are many different human-mediated vectors within Australia that have been shown to spread invasive species. These
include tourists, livestock, vehicles, machinery, trains and transport containers. For example, the movement of cattle has been shown to spread the prickly acacia weed (*Acacia nilotica*) in western Queensland and parthenium weed (*Parthenium hysterophorus*) is spread by the movement of vehicles, machinery, livestock and stock feed. Those two examples are woody, perennial weeds that negatively affect extensive, pastoral operations as there is limited documentation of weed spread into irrigated cropping areas in northern Australia.

**Pests**

Pests are not limited to insects and pathogens, with macro-pests such as birds (cockatoo; galah, *Eolophus roseicapilla*; brolga, *Grus rubicunda*) and macropods (kangaroos and wallabies) considered a risk to introduced irrigated crops, particularly during the drier winter months when native food sources may be scarce. Locally developed and adaptive integrated pest and pesticide resistance management plans are an essential component of best practice and must be implemented pre-emptively.

More intensive land uses, such as irrigated agriculture, are associated with higher likelihoods of pest introduction and greater prospects for dispersal once pests are introduced. Aquaculture and irrigated agriculture entail more movement of materials and people and require more infrastructure than, for example, extensive, rangeland-based pastoralism. Roads and other transport corridors, pipelines, irrigation channels and powerlines, along with river corridors, are important transport routes and foci for invasions of pests. More vehicular traffic provides greater prospects for long-distance dispersal and agricultural machinery in particular is notorious for moving seeds around the landscape. The higher levels of disturbance that are almost inevitably associated with more intensive land uses will also promote invasion by certain kinds of species.

Pest animals are already present in the Mitchell catchment. Feral pigs (*Sus scrofa*) are perhaps the most apparent among terrestrial vertebrate pests and present some threat to agricultural enterprises. They are also destructive in natural environments, for example, through predation on nesting marine turtles on the beaches of the Gulf of Carpentaria, and through extensive damage to wetlands and associated vegetation. Pigs are a major problem in Queensland, and some 75% of their estimated population of 4 to 6 million is found in tropical north Queensland. Pigs can cause indirect damage, for example by carrying weed seed from watercourses to open country, and can cause direct and major physical damage to a wide range of crops and even to cultivated ground.

Pigs have a daily water requirement, which means that during the dry season their range is generally restricted to watercourses and man-made water supplies, precisely the areas where crops are most prospective. Pig control is expensive and so selection of crops not attractive to pigs (e.g. cotton) is desirable where their numbers are high. Freshwater systems are also prone to invasions by non-native species including plants; vertebrates, such as the invasive fish tilapia; and invertebrates.

**Weeds**

The Mitchell catchment, along with other parts of northern Australia, is subject to invasion by a wide variety of weeds, many of them deliberately introduced (Cook and Dias, 2006). Some are more generally problematic while others are problematic for specific industries, sectors or land users. Riparian zones and other more mesic parts of the landscape are prone to a greater variety of weeds than elsewhere, but even drier parts of the landscape provide niches for some invasive
species. Greater levels of disturbance, such as those that occur in association with any cropping system, provide opportunity for particular types of weeds.

Some plants associated with agricultural or more intensive pastoral developments can themselves become problematic for other land users. These could be the crop or pasture species themselves or commensals of the cropping and grazing systems. Such plants and other pests present particular threats to environmental assets. Olive hymenachne (*Hymenachne amplexicaulis*), a Weed of National Significance, is one example. It is present in Southedge Dam (Lake Mitchell), in the upper reaches of catchment, presenting the very real threat of downstream spread into riparian zones and coastal and other wetlands.

### 7.3.3 AQUACULTURE BIOSECURITY

Aquatic diseases are the main biosecurity risk to aquaculture and are caused by a range of pathogenic agents such as viruses, bacteria, fungi and parasites that have varying impacts across species, geographies, rearing systems and life stages.

**Biosecurity risk to new aquaculture farming enterprises in the Mitchell catchment**

This section briefly examines pathways by which pathogens could enter an operation in the Mitchell catchment. It then discusses disease and other biosecurity issues relating to the two main tropical farmed species, black tiger prawns (*Penaeus monodon*) and barramundi (*Lates calcarifer*). However, it should be noted that there are significant, and often different, experiences and issues posed by pathogens and disease for all Australian aquaculture industries.

**Pathways of entry**

The introduction of pathogens into aquatic farming systems comes from two main routes, the first being vertical transmission from parent to progeny, and the second being horizontal transmission from an infected environment, equipment, worker, or animal coming into contact with an uninfected animal during the rearing process. Horizontal transmission can occur through many vectors that harbour the pathogen such as the rearing water, other animals, dead tissues that are consumed, or animal faeces which are consumed or touched. Understanding the primary mode of transmission for each pathogen is critical to understanding how to mitigate disease risks. Applying preventative biosecurity measures that mitigate risks of all routes of transmission for all likely problematic pathogens is key to managing disease. Importantly, the existence of pathogens in the farming system does not necessarily equate to disease, and so disease management in aquaculture needs to both exclude those pathogens that can be excluded, and manage those pathogens that cannot be excluded.

**Pathogens**

As with all agricultural industries, there are a range of pathogens that pose risks to and may impact aquaculture. Fortunately, Australia is free of many of the aquatic pathogens that affect other aquaculture farming regions of the world.

For prawns, the disease agents that have most affected farming have been viruses. There are a large number of different viruses that can infect prawns, which vary significantly in their ability to cause disease and affect production. Fortunately, most of the highly pathogenic viruses are exotic to Australia. However, the recent discovery of the highly pathogenic WSSV in south-east...
Queensland farms has led to investigations to determine whether WSSV is in fact endemic, or the result of an aberrant localised introduction (QDAF, 2017). Several endemic viruses can also have an effect on Australian prawn production, particularly when detrimental pond conditions, such as poor water quality, inflict environmental stress on the prawns and trigger disease episodes (QDPIF, 2006).

Bacteria pathogens can also reduce production, but are also often believed secondary to other stressors (QDPIF, 2006). Recently, syndromes caused by toxicity associated with bacteria have had significant impacts on prawn production both in Australia (Penaeus monodon mortality syndrome (QDAF, 2016)) and even more so overseas (acute hepatopancreatic necrosis disease (NACA, 2016)). Fungi and a range of other microbial and parasitic agents can also cause disease at various life stages and have a negative effect on the appearance of harvested prawn products, but have rarely affected Australian farming in recent decades due to better health and pond management practices.

For barramundi, a range of viral, bacterial, fungal and parasitic pathogens can also affect hatcheries and grow-out. The predominant viral pathogens of concern for barramundi farming in Australia are the nodaviruses, which can cause major mortalities in larval and juvenile barramundi. Bacterial diseases, such as streptococcosis, can also cause high mortalities in both fresh and marine farming systems. Vibriosis and other bacterial pathogens, which infect the gut (causing ‘bloat’) and the gills, also reduce production in fresh and marine waters but are typically secondary to other environmental and dietary stresses. Fungal diseases causing ulceration also periodically affect production in the freshwater and estuarine phases, and typically cause fish to become lethargic and prone to cannibalism. Parasitic protozoans residing in the skin and gills can increase in numbers at times and cause disease, and a blood protozoan has also been associated with major mortalities in sea-caged barramundi. In addition to these non-infectious diseases, particular deformities can reduce production, typically due to nutritional inadequacies in the diet.

A comprehensive knowledge of pathogen agents is essential for developing and implementing risk-based biosecurity measures to mitigate against disease impacts in aquaculture. Understanding of the diseases and disease agents that are likely present in various jurisdictions, or through the process of acquiring animal stocks, and which may have adverse effects, is also important in developing a biosecurity plan. Government departments have important roles in the ongoing surveillance of pathogens, in controlling translocation of stocks based on pathogen risks, and in undertaking investigations where potential disease episodes have been identified (Department of Agriculture and Fisheries Queensland, 2013).

Due to increasing awareness of pathogen risks and the need for biosecurity, and the increasing professionalism of farming operations, it is becoming more common for individual farms to undertake their own pathogen monitoring to minimise the disease risks to their operations. The key elements to effective biosecurity and disease management at the farm level are to access clean and healthy stock; to provide a clean and healthy rearing environment (e.g. good quality water); to provide an adequate quality and quantity of diet; and to control access to water, equipment and people that may introduce pathogens into the farming system.

Treatment actions once diseases are present typically provide few options, particularly for viral pathogens, and are also costly to implement and rarely as effective as prevention. Consequently, for aquaculture, the most important component of disease management is prevention. Important components of prevention are hygiene and biosecurity in the earliest hatchery stages of
production, as well as decontamination processes between crops to ensure the environment is clean before the next crop is commenced. Another very important aspect of disease management is to maintain a quality rearing environment, as both the introduction of pathogens, and more importantly the increase of pathogens in the environment and their manifestation to a disease episode, is typically triggered by increased stress on the animals caused by a poor rearing environment.

Due to the rudimentary immune system of crustaceans, there is limited ability to manage the most serious diseases once established, and so pathogen management has typically focused on exclusion through pathogen screening of broodstock and postlarvae prior to stocking ponds. Some treatments for external bacterial and fungal pathogens are employed, particularly for broodstock, eggs and larvae within the hatchery (FAO, 2007). During the rearing of larvae, control of bacterial pathogens is typically focused on maintaining a good environment and through pre- and probiotics, with antibiotics used only in exceptional circumstances.

Biosecurity risks that new aquaculture farming enterprises in the Mitchell catchment may present to the broader catchment or industry

Risk to other aquaculture enterprises

A major risk pathway with the development of new and established marine aquaculture enterprises is associated with sharing of a water source (usually a river). The risk of contamination between enterprises is highest when there is limited distance (<2 km) between the location of the discharge point from one farm and the supply point of another farm.

Risk to wild populations

There is potential for disease transfer between aquaculture species (e.g. prawns and barramundi) and their respective wild populations. The main transfer routes are discharge waters containing disease or from infected animals (escapees) in discharge water or transferred by predatory vectors (e.g. seabirds). The potential impact of disease on wild populations will depend on pathogen volume, ability of the pathogen to survive without a host, proximity of a significant susceptible host population and the health and tolerance of the host to the disease. In general, susceptible animals in the wild occur only in low-density populations adjacent to land-based aquaculture operations.

The effect that exotic or endemic disease outbreaks from aquaculture have on wild stocks is difficult to evaluate. In Australia, impacts of disease transferred from aquaculture to wild populations have not been widely reported and are difficult to detect. In overseas countries where WSSV is endemic there is little evidence that the disease has any effect on wild prawn populations. In Australia, the response to a suspected outbreak of exotic disease (e.g. WSSV) involves the farm notifying the relevant authorities, isolation of affected ponds and preventing water flow from the ponds to the surrounding environment. The authority (e.g. Biosecurity Queensland) provides advice, which depending on the diagnosis may include destruction and disposal of stock and decontamination of the site. In the case of the recent WSSV discovery in south-east Queensland, a surveillance program commenced (post decontamination) that requires 24 months of no detection of infection in the wild before farming can recommence (DAF, 2018). Since the introduction of the surveillance program in Queensland only very small numbers of infected wild crustaceans have been detected, the vast majority sampled in the vicinity of the original discovery.
The discovery of exotic disease may have a larger impact on the fisher than the wild fishery. For example, in the case of WSSV in Queensland, local commercial and recreation prawn fishers have been constrained by a ban on the movement of uncooked prawns within a restriction zone, which stretches from Caloundra to the NSW border (DAF, 2018). If an exotic disease (e.g. WSSV) was to be identified in aquaculture enterprises located in the Mitchell catchment, commercial fishers operating in waters adjacent to the catchment would likely face similar restrictions in the movement of prawns.

Biosecurity regulatory impacts on development
The aquaculture industry is managed by numerous agencies at the local, state and federal levels of government. To date, significant development of marine aquaculture in northern Australia has in part been constrained by complex legislation and the absence of aquaculture-specific policy, particularly relating to biosecurity issues. A parliamentary inquiry into the development of northern Australia identified the regulatory environment as a serious impediment to major expansion of the prawn farming industry (Parliament of Australia, 2014).

In general, large areas of low-value land located away from populated coastal areas are likely to be suitable for freshwater ponds. In contrast, marine ponds require higher value coastal land, often located in close proximity to towns or regional centres. Compared to marine pond aquaculture, freshwater pond aquaculture is ranked as a low environmental risk option for development. There is no difference in the probability of marine or fresh pond water escaping containment and seeping into the groundwater or surrounding environment. However, marine water discharged into groundwater or freshwater bodies has greater potential to cause negative environmental and ecological impacts.

The approval process for an aquaculture licence is simple for freshwater pond-based farming. This is reflected in the number of licence approvals. For example, in Queensland in 2014–15 there were 158 development approvals for freshwater red claw production compared to 58 approvals for marine prawn production. The disconnection between the number of licence approvals and value of the respective industries ($1 million for red claw and $86 million for prawns) is due to the greater difficulty in obtaining an aquaculture licence for marine pond-based farming (Savage, 2016).

The approval process will vary depending on the state or territory and jurisdiction of water source. Specific details on the approvals required for a land-based aquaculture operation can be found at the website of the relevant authority. Two reviews undertaken in 2013 and 2014 by the Centre for International Economics (CIE) provide a good overview of the regulatory framework for aquaculture in Queensland (CIE, 2013, 2014). The 2014 CIE review provides a comparative assessment of Queensland with three southern jurisdictions, highlighting the degree of difference in regulatory approaches across jurisdictions (CIE, 2014).
7.4 Sediment, nutrients and agropollutant loads to receiving waters

7.4.1 INTRODUCTION

Agriculture can affect the water quality of downstream freshwater, estuarine and marine ecosystems. The principal pollutants from agriculture are nitrogen, phosphorus, total suspended solids, herbicides and pesticides (Lewis et al., 2009; Kroon et al., 2016; Davis et al., 2017). Losses via runoff or deep drainage are the main pathways by which agricultural pollutants enter water bodies. The climate, location (e.g. soils and topography), land use (e.g. cropping system) and management (e.g. conservation and irrigation practices) influence the type and quantity of pollutants lost from an agricultural system.

The development of agriculture in northern Australia has been associated with declining water quality (Lewis et al., 2009; Mitchell et al., 2009; De’ath et al., 2012; Waterhouse et al., 2012; Thorburn et al., 2013; Kroon et al., 2016). Since the 1850s it has been estimated that pollutant loads in north-eastern Australian rivers (typically those in which agriculture as a land use dominates) have increased considerably for nitrogen (2 to 9 times baseline levels), phosphorus (3 to 9 times), suspended sediment (3 to 6 times) and pesticides (~17,000 kg) (Kroon et al., 2016). Degraded water quality can cause a loss of aquatic habitat, biodiversity, productivity and ecosystem services. Increased nitrogen and phosphorus can cause planktonic blooms and weed infestation, increased hypoxia, and result in fish deaths. Suspended sediment can smother habitat and aquatic organisms, reduce light penetration and reduce dissolved oxygen levels. Pesticides may be toxic to habitats and aquatic organisms (Pearson and Stork, 2009; Brodie et al., 2013; Davis et al., 2017).

Water quality monitoring has been undertaken in specific areas of the Mitchell catchment. There is evidence that agricultural development within the Mitchell catchment has resulted in reduced quality of the water discharged from the Mareeba–Dimbulah Water Supply Scheme (MDWSS). Nutrient concentrations have been recorded to be two to ten times higher than the acceptable level in Cattle Creek and Two Mile Creek, with ammonia concentrations high enough to be acutely toxic to aquatic animals (Butler et al., 2008).

Northern Australian river systems are distinctive as they may have highly variable flow regimes, unique species composition, low human population densities and, in some cases, naturally high turbidity (Brodie and Mitchell, 2005). Primary influences on water quality include increased sediment loads associated with land clearing, grazing, agriculture and late dry-season fires, and nutrient pollution from agricultural and pastoral land use (Dixon et al., 2011).

Unless specified otherwise, material in sections 7.4.2 and 7.4.3 are summarised from the companion technical report on agriculture viability (Ash et al., 2018) and sections 7.4.4 and 7.4.5 are summarised from the companion technical report on aquaculture viability (Irvin et al., 2018).

7.4.2 AGRICULTURE POLLUTANT LOSSES AT THE PADDOCK SCALE

Two approaches were used to quantify the likely losses of key pollutants (nitrogen, phosphorus, total suspended solids, and chemicals such as herbicides, pesticides and fungicides) from potential agricultural development in the Mitchell catchment:
1. The first approach was a relative-risk assessment of the crops using potential nutrient surpluses arising from recommended tillage management and herbicide, pesticide and fungicide application rates in these cropping systems as indicators of risk of losses of the key pollutants.

2. The second approach involved a more detailed estimate of pollutant losses for some specific crops for agricultural development based on Agricultural Production Systems Simulator (APSIM) simulations of the cropping systems for those crops.

**Relative-risk assessment of pollutant losses**

To estimate the risk of losses of nitrogen, phosphorus, total suspended solids and chemicals from a range of crops and cropping systems, information on a wide range of factors was collated or calculated including information on nutrient surpluses, number of tillage operations, and application rates of herbicides, pesticides and fungicides (Ash et al., 2018).

Central to the relative-risk assessment is nutrient surpluses, which occur when a greater amount of nutrient (e.g. nitrogen, phosphorus) is applied to the crop than is removed from the field in the harvested product. Nutrient surpluses are an indicator of potential nutrient losses from fields and have been used to assess the risk of nutrient discharges from agricultural areas in other parts of northern Australia (Thorburn and Wilkinson, 2013). The risk of herbicide, pesticide, fungicide and sediment losses was assessed from the amount of chemical applied and the number of tillage operations, respectively.

Nitrogen use varied considerably for crops assessed in the Mitchell catchment (Table 7-8). The nitrogen surplus was high for bananas and relatively high for sugarcane, which have high nitrogen inputs and a low quantity of nitrogen removed in the harvestable product. Conversely, nitrogen surplus was low for a number of crops or even negative in the case of Jarrah grass (*Digitaria milanjiana* cv Jarrah). A negative nitrogen surplus indicates that nitrogen inputs are less than the nitrogen in the harvestable product. In these instances, nitrogen from mineralisation of soil nitrogen would be required to meet crop nitrogen demands (Angus, 2001). It should be noted that efficiencies of use will be higher with split applications of nitrogen and where crop rotations or cover crops are used as part of the cropping system.

**Table 7-8 Nitrogen (N) surplus for multiple crops grown in the Mitchell catchment based on risk assessment**

Data calculated using APSIM output and values from literature (see the companion technical report on agricultural viability [Ash et al., 2018]).

<table>
<thead>
<tr>
<th>CROP</th>
<th>N APPLIED TO CROP (kg/ha)</th>
<th>N FIXED (kg/ha)</th>
<th>TOTAL N INPUTS (kg/ha)</th>
<th>N IN HARVESTABLE PRODUCT (kg/ha)</th>
<th>N SURPLUS (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Banana</td>
<td>400</td>
<td>0</td>
<td>400</td>
<td>73</td>
<td>327</td>
</tr>
<tr>
<td>Chickpeas</td>
<td>6</td>
<td>87</td>
<td>93</td>
<td>92</td>
<td>1</td>
</tr>
<tr>
<td>Cotton</td>
<td>180</td>
<td>0</td>
<td>180</td>
<td>110</td>
<td>70</td>
</tr>
<tr>
<td>Jarrah grass (hay)</td>
<td>71</td>
<td>0</td>
<td>71</td>
<td>122</td>
<td>–51</td>
</tr>
<tr>
<td>Maize</td>
<td>180</td>
<td>0</td>
<td>180</td>
<td>142</td>
<td>38</td>
</tr>
<tr>
<td>Mungbean</td>
<td>23</td>
<td>65</td>
<td>88</td>
<td>85</td>
<td>3</td>
</tr>
<tr>
<td>Peanut</td>
<td>15</td>
<td>124</td>
<td>139</td>
<td>131</td>
<td>8</td>
</tr>
<tr>
<td>Rice</td>
<td>200</td>
<td>0</td>
<td>200</td>
<td>98</td>
<td>102</td>
</tr>
<tr>
<td>Rockmelon</td>
<td>107</td>
<td>0</td>
<td>107</td>
<td>37</td>
<td>70</td>
</tr>
</tbody>
</table>
Phosphorus surplus varied substantially for crops assessed in the Mitchell catchment (Table 7-9). The phosphorus surplus was very high for crops such as bananas and watermelon that have high phosphorus inputs and a low quantity of phosphorus in the harvestable product. Conversely, the phosphorus surplus was low or even negative for crops such as chickpea, maize and sesame. A negative phosphorus surplus indicates that phosphorus inputs are less than the phosphorus in the harvestable product. In these instances, phosphorus reserves in the soil are being depleted (Stewart and Tiessen, 1987).

**Table 7-9 Phosphorus (P) surplus for multiple crops grown in the Mitchell catchment based on risk assessment**

Data calculated using APSIM output and values from literature (see the companion technical report on agricultural viability (Ash et al., 2018)).

<table>
<thead>
<tr>
<th>CROP</th>
<th>TOTAL P INPUTS (kg/ha)</th>
<th>P IN HARVESTABLE PRODUCT (kg/ha)</th>
<th>P SURPLUS (kg/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Banana</td>
<td>101</td>
<td>9</td>
<td>92</td>
</tr>
<tr>
<td>Chickpea</td>
<td>11</td>
<td>9</td>
<td>2</td>
</tr>
<tr>
<td>Cotton</td>
<td>22</td>
<td>14</td>
<td>8</td>
</tr>
<tr>
<td>Jarrah grass (hay)</td>
<td>17</td>
<td>18</td>
<td>–1</td>
</tr>
<tr>
<td>Maize</td>
<td>19</td>
<td>27</td>
<td>–8</td>
</tr>
<tr>
<td>Mungbean</td>
<td>25</td>
<td>8</td>
<td>17</td>
</tr>
<tr>
<td>Peanut</td>
<td>33</td>
<td>11</td>
<td>22</td>
</tr>
<tr>
<td>Rice</td>
<td>28</td>
<td>20</td>
<td>8</td>
</tr>
<tr>
<td>Rockmelon</td>
<td>60</td>
<td>9</td>
<td>51</td>
</tr>
<tr>
<td>Sesame</td>
<td>3</td>
<td>18</td>
<td>–15</td>
</tr>
<tr>
<td>Sorghum (grain)</td>
<td>48</td>
<td>24</td>
<td>24</td>
</tr>
<tr>
<td>Soybean</td>
<td>22</td>
<td>18</td>
<td>4</td>
</tr>
<tr>
<td>Sugarcane</td>
<td>38</td>
<td>16</td>
<td>22</td>
</tr>
<tr>
<td>Sunflower</td>
<td>60</td>
<td>17</td>
<td>43</td>
</tr>
<tr>
<td>Watermelon</td>
<td>60</td>
<td>5</td>
<td>55</td>
</tr>
</tbody>
</table>

The total herbicide, pesticide and fungicide application rates per crop varied substantially for crops grown in the Mitchell catchment (Table 7-10). Some crops, such as bananas, have high pesticide, herbicide and fungicide application rates while other crops, such as rice or mangoes, have much lower application rates. It should be noted that this assessment is quite limited as it is simply reporting total amounts applied rather than the impact of the active ingredients. Many newer herbicides and pesticides have much lower application rates but their active ingredients are relatively more potent than older chemicals. For example, the pesticide chlorantraniliprole...
(Alatacor, Dupont Chemicals) is a recent pesticide that is highly effective against caterpillars in pulse crops and is applied at a rate of just 70 g/ha.

Table 7-10 Herbicide, pesticide and fungicide application rates for multiple crops grown in the Mitchell catchment

<table>
<thead>
<tr>
<th>CROP</th>
<th>TOTAL HERBICIDE APPLICATION (L/ha/CROP)</th>
<th>TOTAL PESTICIDE APPLICATION (L/ha/CROP)</th>
<th>TOTAL FUNGICIDE APPLICATION (L/ha/CROP)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sorghum (grain)</td>
<td>5.5</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Avocado</td>
<td>5.5</td>
<td>13.7</td>
<td>13</td>
</tr>
<tr>
<td>Banana</td>
<td>25.3</td>
<td>90</td>
<td>3.7</td>
</tr>
<tr>
<td>Cashew</td>
<td>2</td>
<td>6.4</td>
<td>0</td>
</tr>
<tr>
<td>Chickpea</td>
<td>8.4</td>
<td>0.1</td>
<td>2</td>
</tr>
<tr>
<td>Cotton</td>
<td>9.1</td>
<td>2.2</td>
<td>0</td>
</tr>
<tr>
<td>Jarrah grass (hay)</td>
<td>6</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Maize</td>
<td>5.5</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>Mango Calypso</td>
<td>3</td>
<td>0.5</td>
<td>3.4</td>
</tr>
<tr>
<td>Mango KP</td>
<td>3</td>
<td>0.5</td>
<td>3.4</td>
</tr>
<tr>
<td>Mungbean</td>
<td>4</td>
<td>3.1</td>
<td>70.1</td>
</tr>
<tr>
<td>Peanut</td>
<td>4.1</td>
<td>0</td>
<td>13.2</td>
</tr>
<tr>
<td>Rice</td>
<td>3.5</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>Rockmelon</td>
<td>1.5</td>
<td>1.5</td>
<td>4.9</td>
</tr>
<tr>
<td>Sesame</td>
<td>3.8</td>
<td>0.14</td>
<td>0</td>
</tr>
<tr>
<td>Sugarcane</td>
<td>9.1</td>
<td>1.5</td>
<td>0</td>
</tr>
<tr>
<td>Sunflower</td>
<td>NA</td>
<td>0.8</td>
<td>0</td>
</tr>
<tr>
<td>Soybean</td>
<td>2.4</td>
<td>5</td>
<td>0</td>
</tr>
<tr>
<td>Watermelon</td>
<td>1.5</td>
<td>1.5</td>
<td>4.9</td>
</tr>
</tbody>
</table>

NA = data not available

The number of tillage operations varied substantially for crops grown in the Mitchell catchment. The greater the number of tillage operations, the greater the risk of loss of soil to the environment, which has the potential to end up as suspended sediment in waterways. As well, intensive tillage operations are likely to be more damaging than low-impact tillage operations. Thus, crops such as melons and bananas that have a high number of both total and intensive tillage operations pose the greatest risk. Crops such as rice and maize that have a low number of both total and intensive tillage operations pose the least risk.

Simulated pollutant losses

Simulated annual loss of nitrogen via runoff varied depending on crop, climate and soil (Figure 7-14a–d). Mean annual simulated loss of nitrogen was 1 kg/ha and the maximum annual loss was 16 kg/ha from a cotton crop on a Brown Sodosol (Figure 7-14c). In general, losses of nitrogen via runoff were higher in cotton than in other crops and tended to be higher for the Brown Sodosol than for the Grey Vertosol (Figure 7-14a–d).

Simulated annual losses of leached nitrogen were higher than losses via runoff and varied depending on crop, climate and soil (Figure 7-14e–h). Mean annual simulated nitrogen loss was 8 kg/ha and the maximum annual loss was 108 kg/ha from a sugarcane crop on a Brown Sodosol.
In general, losses of leached nitrogen were higher in sugarcane than in other crops and tended to be higher for the Vertosol than for the Brown Sodosol (Figure 7-14e–h).

Simulated annual soil losses also varied according to crop, climate and soil (Figure 7-14i–l). Mean simulated annual loss was 6 t/ha and the maximum annual loss was 42 t/ha from a mungbean crop on a Brown Sodosol (Figure 7-14k). In general, soil loss from cotton, mungbean and sugarcane was higher than for forage sorghum and rice and soil loss tended to be higher for the Brown Sodosol than for the Vertosol.

Pollutant losses varied considerably based on rainfall as nitrogen loss via runoff and soil erosion are driven by frequency and intensity of rainfall, while nitrogen leaching is driven by water drainage. An example of these differences is a cotton crop simulated at Dunbar Station on a Grey Vertosol: annual rainfall was 559 mm in 2005 and 1440 mm in 2006. Simulated annual nitrogen losses through runoff were 0.05 kg/ha in 2005 and 0.9 kg/ha in 2006, while N leaching was 3 kg/ha in 2005 and 39 kg/ha in 2006, and soil losses were 1.1 t/ha in 2005 and 5.1 t/ha in 2006 (data not shown). These results are based on a single annual crop and do not include a cover crop or other form of rotational cropping system. Cropping system implications are explored in the next section in the context of reducing pollutant losses.

Climate is also a driver of pollutant losses. Comparison of years with considerably different annual rainfall found nitrogen losses via runoff or leaching were greater in years with high rainfall than with low rainfall. Soil texture also plays a role in driving pollutant losses, with sandy soils more prone to nitrogen losses via leaching than clay soils (Gaines and Gaines, 1994). Future agricultural development could minimise pollutant losses by prioritising development on soils with lower potential for pollutant losses.
Reducing pollutant losses

There is a large body of literature that has investigated approaches to minimise pollutant losses from farming systems in Australia. Refining application rates of fertiliser to better match crop requirements and improving irrigation management are effective ways to minimise nitrogen losses (Brodie et al., 2008; Thorburn et al., 2008, 2011a, 2011b; Webster et al., 2012; Biggs et al., 2013; Thorburn and Wilkinson, 2013). Lower fertiliser application rates has reduced losses of nitrogen via leaching from banana crops Armour et al. (2013). The use of ‘best management practices’ including controlled traffic and banded application of herbicides can substantially reduce the loss of herbicides (Masters et al., 2013; Silburn et al., 2013). Furthermore, crop rotation, particularly the use of a cover crop, can minimise soil loss (Carroll et al., 1997; Dabney et al., 2001). In a simulated example of a cropping rotation that includes a summer cover crop, simulated annual soil loss was reduced from 6.5 t/ha for a single cotton crop to between 2.2 and 2.4 t/ha for a cotton–sorghum or cotton–soybean rotation (Figure 7-15). Nitrogen losses were little affected by the cropping system rotation but for all scenarios these losses were very low.

Figure 7-15 Simulated annual N losses via runoff or leaching and soil loss from Chillagoe climate station and a Brown Sodosol for a cotton crop, a cotton–sorghum crop rotation, and a cotton–soybean crop rotation for the Mitchell catchment

Simulation duration was 125 years (1890 to 2015).

7.4.3 MANAGING IRRIGATION DRAINAGE

Surface drainage water is water that runs off irrigation developments as a result of over-irrigation or rainfall. This excess water can potentially affect the surrounding environment by modifying flow regimes and changing water quality. Hence, management of irrigation or agricultural drainage waters is a key consideration when evaluating and developing new irrigation systems and should be given careful consideration during the planning and design process. Regulatory constraints on
the disposal of agricultural drainage water from irrigated lands are being made more stringent as this disposal can potentially have significant off-site environmental effects (Tanji and Kielen, 2002). Hence, minimising drainage water through the use of best-practice irrigation design and management should be a priority in any new irrigation development in northern Australia. This involves integrating sound irrigation systems, drainage networks and disposal options so as to minimise off-site impacts.

Surface drainage networks need to be designed to cope with the runoff associated with irrigation, and also the runoff induced by rainfall events on irrigated lands. Drainage must be adequate to remove excess water from irrigated fields in a timely manner, and hence reduce waterlogging and salinisation, which can seriously limit crop yields. In best-practice design, surface drainage water is generally reused through a surface drainage recycling system where runoff tailwater is returned to an on-farm storage or used to irrigate subsequent fields within an irrigation cycle.

The quality of drainage water will vary depending upon a range of factors including water management and method of application, soil properties, method and timing of fertiliser and pesticide application, hydrogeology, climate and drainage system (Tanji and Kielen, 2002). These factors need to be taken into consideration when implementing drainage system water recycling and also when disposing of drainage water to natural environments.

A major concern with tailwater drainage is the agropollutants derived from pesticides and fertilisers that are generally associated with intensive cropping and are found in the tailwater from irrigated fields. Crop chemicals can enter surface drainage water if poor water application practices or significant rainfall events occur, after pesticide or fertiliser application (Tanji and Kielen, 2002). Tailwater runoff from pesticides and fertilisers can contain phosphate, organic nitrogen and pesticides that have the potential to adversely affect flora and fauna and ecosystem health, on land waterways, estuaries or marine environments. Tailwater runoff may also contain elevated levels of salts, particularly if the runoff has been generated on saline surface soils.

Training of irrigators in responsible application of both water and agrochemicals is therefore an essential component of sustainable management of irrigation.

As tailwater runoff is either discharged from the catchment or captured and recycled it can result in a build-up of agropollutants that may ultimately require disposal from the irrigation fields. In externally-draining basins, the highly seasonal nature of flows in northern Australia does offer possibilities to dispose of poor-quality tailwater during high-flow events. However, downstream consequences are possible and no scientific evidence is available to recommend such disposal as good practice. Hence, consideration should be given to providing an adequate understanding of downstream consequences of disposing of drainage effluent and options must be provided for managing disposal that minimise impacts on natural systems.

### 7.4.4 CHEMICAL CONTAMINANT RISKS TO AQUACULTURE

Hundreds of different chemicals, including oils, metals, pharmaceuticals, fertilisers and pesticides (e.g. insecticides, herbicides, fungicides) are used in different agricultural, horticultural and mining sectors, and in industrial and domestic settings, throughout Australia. The release of these chemical contaminants beyond the area of target application can lead to the contamination of soils, sediments and waters in nearby environments. In aquatic environments, including aquaculture environments, fertilisers have the potential to cause nonpoint source pollution. This
eutrophication is caused by nutrients that trigger excessive growth of plant and algal species, which then form hypoxic (low oxygen) ‘dead zones’ and potentially elevated levels of toxic un-ionised ammonia (Kremser and Schnug, 2002). This can have a significant impact on the health and growth of animals in aquaculture operations, as well as in the broader environment. For example, health indicators are lower in barramundi collected from agriculturally affected rivers in Queensland relative to those collected at more pristine sites.

Of concern to aquaculture in northern Australia are the risks posed to crustaceans (e.g. prawns and crabs) by some of the insecticides in current use. These insecticides can be classified based on their specific chemical properties and modes of action. The different classes of insecticides have broad and overlapping applications across these different settings.

The first class is organophosphate insecticides of which toxicity is not specific to target insects, raising concerns about the impacts on non-target organisms, such as crustaceans and fish. Despite concerns about human health impacts and potential carcinogenic risks, organophosphates are still one of the most broadly used insecticides globally and are still used in Australia for domestic pest control (Weston and Lydy, 2014; Zhao and Chen, 2016). Pyrethroid insecticides have low toxicity to birds and mammals, but higher toxicity to fish and arthropods. Phenylpyrazole insecticides are another class that also pose risks to non-target crustaceans (Stevens et al., 2011). Neonicotinoid insecticides are a class being used in increasing amounts because they are very effective at eliminating insect pests, yet pose low risks to mammals and fish (Sánchez-Bayo and Hyne, 2014). Monitoring data from the Great Barrier Reef catchments indicate that concentration of neonicotinoid insecticides in marine water samples is rapidly increasing with widespread use. One significant concern for aquaculture is the risk that different insecticides, when exposed to non-target organisms, may interact to cause additive or greater than additive toxicity.

An awareness and knowledge of the potential exposures, risks, and impacts that chemical contaminants may pose in a location is valuable when establishing and operating a commercial aquaculture enterprise, to ensure exposures are best mitigated. For the most vulnerable life cycle stages of production, such as the larval stages of rearing within the controlled hatchery environment, water treatment systems can be employed to mitigate risks of exposure to contaminants. However, for broadacre pond systems, the best approach is to understand the risks of exposure in an area, and ideally to establish farms in areas of lower exposure.

### 7.4.5 AQUACULTURE DISCHARGE WATER AND OFF-SITE IMPACTS

Discharge water is effluent from land-based aquaculture production. Discharge water is water that has been used (culture water) and is no longer required in a production system. In most operations (particularly marine), bioremediation is used to ensure that water discharged off farm into the environment contains low amounts of nutrients and other contaminants. The aim is for discharge waters to have similar physiochemical parameters to the source water.

Discharge water from freshwater aquaculture can be easily managed and provides a water resource suitable for general or agriculture-specific irrigation. Marine discharge water is comparatively difficult to manage, with limited reuse applications. The key difference in discharge management is that marine (salty) water must be discharged at the source, whereas location for freshwater discharge is less restrictive and potential applications numerous (e.g. irrigation).

Specific water discharge guidelines vary with species and jurisdiction. For example in Queensland,
water discharge policy minimum standards for prawn farming include minimum standards for physiochemical indicators (e.g. oxygen and pH) and nutrients (e.g. nitrogen, phosphorus and suspended solids) and total volume (EHP, 2013).

Management of water quality in ponds is a key component of aquaculture production. A consequence of water quality management is the requirement to discharge effluent water into the surrounding environment.

A large multidisciplinary study on intensive Australian prawn farming, which assessed the impact of effluent on downstream environments (CSIRO, 2013), found that Australian farms operate under world best practice in regards to the management of discharge water. The study found that discharge water had no adverse ecological impact on receiving water and that nutrients could not be detected 2 km downstream from the discharge point.

While Australian prawn farms are reported as being among the most environmentally sustainable in the world (CSIRO, 2013), location of the industry adjacent to the listed Great Barrier Reef and related strict policy on discharge has been a major constraint to expansion of the industry. Strict discharge regulation, which require zero net addition of nutrients in waters adjacent to the Great Barrier Reef, has all but halted expansion in the last decade. An example of the regulatory complexity in this region is the 14-year period taken to obtain approval to develop a site in the Burdekin shire in north Queensland (APFA, 2016). Over the last decade, increases in production have been due to improvements in production efficiency rather than any expansion of the industry footprint.

In a report to the Queensland Government (Department of Agriculture and Fisheries Queensland, 2013) it was suggested that less-populated areas in northern Australia, which have less conflict for the marine resource, may have potential as areas for aquaculture development. The complex regulatory environment in Queensland was a factor in the decision by Project Sea Dragon to investigate greenfield development in WA and NT as an alternative location for what would be Australia’s largest prawn farm (Seafarms, 2016).

Today most farms (particularly marine) use bioremediation ponds to ensure that water discharged off-farm into the environment contains low amounts of nutrients and other contaminants. The prawn farming industry in Queensland has adopted a code of practice to ensure that discharge waters do not result in irreversible or long-term impacts to the receiving environment (Donovan, 2011).

Pump stations are used to distribute water around the farm (Figure 7-16). Marine water is pumped from a primary pump station located near the water source (usually a river) to a raised supply channel engineered to gravity-deliver water to the ponds. During production and at final harvest water is discharged from production ponds via gravity into a waste water channel. A secondary pump station is then used to pump the water from the waste water channel to the bioremediation pond. The role of the bioremediation process is to reduce suspended solids and nutrients (nitrogen and phosphorus) in the water to meet discharge water quality standards set by regulators. Water treated in the bioremediation pond is either recirculated to the production pond via a third pump station and the supply channel or discharged by gravity to the river. The specifications of each pump station are in keeping with the volume of water required to fill the ponds and to service water exchange requirements. Farm layout should be designed to minimise the chance of reintroducing discharged water to the ponds via the primary pump station. The
location of the primary pump station and the discharge channel should be separated by as large a physical distance as practical. In general, best practice involves access of source water at high tide and discharge of water at low tide.

The volume of water required to be discharged or possibly diverted to a secondary application (e.g. agriculture) is equivalent to the total pond water use for the season, minus total evaporative losses and the volume of recycled water used during production.

7.5 Irrigation-induced salinity

7.5.1 INTRODUCTION

Salts occur naturally in all soils and landscapes depending on climate; the salt store in the geology, soils and watertables; and landscape hydrology. Naturally-occurring areas of salinity or ‘primary salinity’ occur in the landscape with ecosystems adapted to these conditions. Any change to landscape hydrology, including clearing and irrigation, can mobilise salts resulting in environmental harm and agricultural productivity losses, a process referred to as ‘secondary salinity’. Secondary salinity manifests itself in two main forms: that which occurs in irrigation regions and salinity occurring in dryland regions. The Assessment is concerned with irrigation-induced ‘secondary salinity’.

Figure 7-16 Cross-section of a marine aquaculture farm detailing optimal land elevation, water flow and discharge

The volume of water required to be discharged or possibly diverted to a secondary application (e.g. agriculture) is equivalent to the total pond water use for the season, minus total evaporative losses and the volume of recycled water used during production.
In the case of irrigation-induced salinity, an increase in root-zone drainage following applications of irrigation water can provide a source of water to mobilise soluble salts stored in the soil. Root-zone drainage rates tend to be higher under coarser-textured soils (Petheram et al., 2002) and poor irrigation practices. In Australia, excessive root-zone drainage through poor irrigation practices, together with leakage of water from irrigation distribution networks and drainage channels, has caused the watertable level to rise under many intensive irrigated areas. Significant parts of all major intensive irrigation areas in Australia are currently either in a shallow watertable equilibrium condition or approaching it (Christen and Ayars, 2001). Where shallow watertables containing salts approach the land surface (in the vicinity of 2 to 3 m from the land surface), salts can concentrate in the root zone over time through evaporation. The process by which salts accumulate in the root zone is accelerated if the groundwater also has high salt concentrations.

The Mitchell catchment has moderate natural surface salinity occurring in drainage lines and at lower slopes of deeply weathered geologies in the south of the centre of the catchment. These areas are not considered suitable for irrigation development. Soils suitable for irrigation development in the Mitchell catchment tend to be either deep, free-draining loams (SGG 4.1, 4.2), sand or loam over sodic clay subsoils (SGG 8), or clays (SGG 9) (Section 2.3). Of the more suitable soils for irrigated agriculture the clay soils at Wrotham Park, at the centre of the catchment, were identified as having the highest salinity risk in the Mitchell catchment.

The survey undertaken for this reconnaissance salinity appraisal used an EM34 instrument able to measure to approximately 40 m in depth under favourable conditions (Reynolds, 2000). Through electromagnetic induction (EM), the electrical conductivity (ECa) patterns are used to differentiate between conductive/resistive layers in the profile, which equate to mineralogy (clay type), salinity, soil water content and rock.

This section is structured around the three basic requirements for salt to become an environmental problem: (i) a source of salt (Section 7.5.2), (ii) a source of water in which to mobilise the salt (Section 7.5.3), and (iii) mechanisms by which the salt is redistributed to locations in the landscape where it causes damage (Section 7.5.3).

### 7.5.2 POTENTIAL SOURCES OF SALT

**Salt stores in the Mitchell catchment**

The salts in the landscape are derived from salts delivered through rainfall, weathering of primary minerals and origin of the geology such as marine sediments. The amount of salts in the landscape (salt store) depends on the origin of salts, degree of geology weathering, climate (particularly rainfall), position in the landscape, landscape permeability (soils and rock) and watertable dynamics.

It should be noted that this section presents generalised results on soils suitable for irrigation development. The risk of secondary salinisation at a specific location in the Mitchell catchment can only be properly assessed by undertaking detailed field investigation.

The gently undulating plains and rises with cracking clay soils (SGG 9) developed on fine-grained sediments of the Great Artesian Basin at Wrotham Park in the centre of the catchment show considerable potential for irrigated agriculture (Figure 7-17). However, field work indicated a considerable salt store in the landscape where salts may mobilise and cause secondary salinity if a watertable rises close to the surface, resulting in lost crop productivity.
Five sites were selected to represent the various landscape positions, as shown in Figure 7-18. Core samples were collected at the five sites for analysis while a geophysical survey using an EM34 instrument to measure EM was conducted at three of the sites (SAL1, SAL2 and SAL5).

A representation of the land surface is presented in Figure 7-19, which as a conceptual model highlights the subtle landscape features and positions of the sites. Stylised profile logs comprising distributions of saline and sodic properties, clay %, rock depth (if encountered) and main water flows (and relative strength) as recorded or interpreted from field and laboratory data are presented in the model. Upper slopes and plains (SAL1 to SAL3) are essentially non-saline with the EM measurements showing moderately conductive layers with low EM readings to approximately 5 m over high readings, which is consistent with the depth that bedrock was encountered. Lower slope positions (SAL4 and SAL5) have naturally very high salt levels at less than 1 m.
Figure 7-18 Study area site locations (red dots) and EM34 transects (blue dots)

Figure 7-19 Study area cross-sectional conceptual model showing site positions, land surface elevation and landform, profile soil properties (EC, ESP, clay %, bedrock depth) and key water flows
The EM trace at SAL5 (Figure 7-20) shows distinctive conductivity lobes at depths of approximately 20 m and 30 m that may correspond to saline groundwater and/or lithological structure which is consistent with the EM traces of other sites.

![Figure 7-20 EC trace for site SAL5](image)

Under irrigation, the soils on the upper slopes are essentially non-saline in the rooting zone and salinity only becomes apparent at depths greater than 2 m. With application of good quality water these soils can be successfully irrigated with careful management of water rates to avoid waterlogging and ponding (and possible water erosion). The sites are likely to be hydraulically connected by water throughflows, although rates are likely to be extremely slow.

The sites on lower slopes contain more salts. Given the landscape position and gradient, surface flow rates are likely to be high and throughflow rates moderate, which are likely to be sufficient over time to accumulate salts in lower landscape positions. This explains the saline-sodic conditions of SAL4 downslope. Here the throughflow rates (very slowly permeable) in the low-slope gradients are insufficient to off-set salt supplies from upslope with leaching rates downslope, hence salts accumulate in this zone. Down slope of this at SAL5, the loamy textures deeper than 2 m act as an internal drain for upslope salts due to the moderately permeable conditions and the maximum watertable level decreases as the distance to the river decreases because groundwater can be discharged to the river at a greater rate.

It is likely that the watertables on the lower slopes would take a relatively long period to rise to close to the surface. For additional information on groundwater in the Mitchell catchment see the companion technical report on hydrogeological assessment (Taylor et al., 2018).

**Irrigation water as a potential source of salt**

In many irrigation developments around the world, poor-quality irrigation water is the source of salt in salinisation. In the Mitchell catchment, however, the river water is relatively fresh and aquifers with potential for groundwater resource development are also relatively fresh, so are unlikely to be a source of salt. This is because the low levels of salt in the river water or groundwater would be leached through the soil profile before they could accumulate in the root zone to levels that adversely affect crop development. However, in some cases, certain levels of localised groundwater extraction from aquifers may result in the entrainment of poorer quality water from surrounding aquifers or aquitards over time, thereby reducing the overall quality of groundwater applied to crops. The potential for this to occur would require a site-specific investigation.
7.5.3 RISE IN WATERTABLE LEVEL AND CHANGES IN GROUNDWATER DISCHARGE DUE TO IRRIGATION DEVELOPMENT

The extent to which the watertable level rises close to the surface depends on:

- the initial depth to the watertable
- the amount of recharge (originating from root-zone drainage)
- the size of the irrigation area (thus dictating the total volume added to the landscape)
- the lateral distance to the river (which acts as a drainage boundary, thus reducing the height of the groundwater mound under irrigation)
- aquifer parameters, including the saturated hydraulic conductivity, aquifer thickness and specific yield.

The hydraulic conductivity and specific yield are hydraulic properties of a soil’s ability to transmit water when submitted to a hydraulic gradient (e.g. difference in watertable level between two locations). The specific yield is the volume of water that could be allowed to drain from an aquifer under the forces of gravity and expressed as a proportion of the total volume of material in the aquifer.

In the Mitchell catchment, there is limited shallow groundwater data, and aquifer parameters typically need to be estimated from bore log information and generic relationships in the literature. The use of such relationships is made particularly challenging by the fact that saturated hydraulic conductivity is the most variable environmental parameter, its range varying by over 11 orders of magnitude. Typical values of saturated hydraulic conductivity and specific yield are provided in Table 7-11.

Table 7-11 Typical values of specific yield and saturated hydraulic conductivity

<table>
<thead>
<tr>
<th>SOIL TEXTURE</th>
<th>SPECIFIC YIELD</th>
<th>SATURATED HYDRAULIC CONDUCTIVITY (m/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gravel</td>
<td>0.25</td>
<td>3 to 30,000</td>
</tr>
<tr>
<td>Sand</td>
<td>0.20</td>
<td>0.3 to 300</td>
</tr>
<tr>
<td>Silt</td>
<td>0.18</td>
<td>0.00003 to 3</td>
</tr>
<tr>
<td>Clay</td>
<td>0.02</td>
<td>0.0000003 to 0.00003</td>
</tr>
</tbody>
</table>

*Adapted from Johnson (1967) and Carsel and Parrish (1988).
‡Adapted from Freeze and Cherry (1979).

An analytical modelling approach (Jolly et al., 2013) was used to evaluate the maximum (steady-state) rise in watertable level likely as a result of introducing new irrigation developments of varying areas situated at various distances from a river. A separate analysis was undertaken to investigate the time it takes the watertable to rise to its maximum level and how changes in groundwater discharge occur. It is important to note that these results are under ‘idealised’ conditions. The risk of secondary salinisation and watertable rise at a specific location in the Mitchell catchment can only be properly assessed by undertaking detailed field investigation.

**Farm-scale developments**

To investigate the sensitivity of the results to these parameters a range of likely values was selected (Table 7-12). Irrigation developments between 100 and 1000 ha in size are representative...
of irrigation developments on individual properties. Web-based applications are provided on the Assessment website that enable the user to assess the maximum rise and change in watertable level using parameter values specific to their area of interest. The results in this section are presented to illustrate general concepts.

Table 7-12 Likely range of values for parameters in farm-scale development in the Mitchell catchment

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>SYMBOL</th>
<th>UNIT</th>
<th>VALUES</th>
<th>COMMENT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Distance from centre of irrigation area to river</td>
<td>d</td>
<td>km</td>
<td>0.5, 1.0, 2.0, 5.0, 10.0</td>
<td>River was assumed to be straight.</td>
</tr>
<tr>
<td>Circular irrigation area</td>
<td>A</td>
<td>ha</td>
<td>100, 250, 500, 1000</td>
<td>For radii of 564, 892, 1262 and 1784 m.</td>
</tr>
<tr>
<td>Recharge rate</td>
<td>R</td>
<td>mm/y</td>
<td>1, 10, 20, 50, 100, 200, 500</td>
<td>Recharge rate is related to the amount of water applied and the permeability of the soil. A recharge rate of 500 mm/y (or more) could occur under a ringtank.</td>
</tr>
<tr>
<td>Aquifer transmissivity (saturated hydraulic conductivity multiplied by aquifer thickness)</td>
<td>T</td>
<td>m²/day</td>
<td>200, 500, 2000</td>
<td>Represents a constant saturated aquifer thickness (h = 10 m), and hydraulic conductivities (K) of 20, 50 and 200 m/day.</td>
</tr>
<tr>
<td>Specific yield</td>
<td>Sy</td>
<td></td>
<td>0.10 to 0.20</td>
<td>Specific yield does not alter the maximum height of the watertable. It affects the time over which the watertable rise occurs.</td>
</tr>
</tbody>
</table>

Maximum rise in watertable level

The maximum rise in watertable level increases with higher recharge rates and decreases with higher saturated hydraulic conductivity. Figure 7-21a shows that the effects of saturated hydraulic conductivity (and hence aquifer transmissivity) and recharge rates are linear but opposite and perfectly correlated. Hence, to simplify the presentation of results and reduce the number of variables, it is possible to report watertable level against recharge rate divided by the aquifer transmissivity.

Figure 7-21b shows the maximum watertable level expected for an irrigation area of 100 ha. This maximum level decreases as the distance to the river decreases. This is because, with the irrigation development located closer to the river, groundwater can be discharged to the river at a greater rate.

Figure 7-22 shows that the maximum watertable level increases in a non-linear manner as the distance to the river increases.

Figure 7-23 and Figure 7-24a show that the maximum watertable level increases as the irrigation area increases, and also as the recharge rate increases. For the combination of parameters considered for the Mitchell catchment, the highest point on the red line in Figure 7-24a shows the upper bound for watertable rise (h_{max} = 41.8 – 10 = 31.8 m, where 10 m is the initial watertable level), which represents the largest irrigation area (A = 1000 ha) located furthest from the river (d), with an aquifer having the lowest drainage capacity (highest R/T). Figure 7-24b presents the same results in a different way to highlight the effect of increasing the recharge area and distance to the river on watertable level.
Figure 7-21 Steady-state watertable level for (a) various recharge rates and hydraulic conductivities ($K$) and (b) an irrigation area of 100 ha, at varying distances to the river

Figure 7-22 Steady-state watertable level for an irrigation area of 1000 ha, plotted against distance to the river

Figure 7-23 Steady-state watertable level at varying distances ($d$) to the river for an irrigation area of (a) 250 ha and (b) 500 ha
Changes in rise in watertable level over time

One of the challenges in managing groundwater is the time lag between a change in management and the response of the groundwater system. This analysis demonstrates how the watertable level rises over time until it achieves its maximum height. A key parameter for undertaking this analysis is the specific yield of the groundwater system. Figure 7-25 provides an example where the irrigation area is 100 ha and the recharge rate is 100 mm/year. For a given aquifer diffusivity ($D$) (i.e. aquifer transmissivity divided by specific yield), Figure 7-25 shows that after a change in recharge, the initial response of the groundwater system is identical regardless of the distance to the river ($d$). This is because the groundwater mound under an irrigation development forms before groundwater discharge to the river increases. When the groundwater mound reaches the river, the rate of the rise in watertable level starts to decline until the level reaches its maximum (i.e. under steady-state conditions). The watertable level takes longer to reach its maximum when the irrigation development is further from the river (Figure 7-25).

In Figure 7-25, for an aquifer diffusivity ($D$) with a high value of 200,000 m$^2$/day, the maximum watertable level is reached within about 13 years, whereas for the low value of 20,000 m$^2$/day, the maximum watertable level is reached in 30 to 100 years. The watertable level takes longer to reach its maximum when irrigation areas are larger and are located a greater distance from the river. With the most extreme combination of parameters from Table 7-12 ($d = 10$ km, $A = 1000$ ha, $R = 500$ mm/year, and $D = 2000$ m$^2$/day), it takes the watertable level about 270 years to reach approximately 90% of its rise.
Groundwater mounds under irrigation developments can result in increased groundwater discharge to nearby rivers. This can have important ecological implications. The time taken for a groundwater mound to discharge to a nearby river depends on the aquifer diffusivity and the distance to the river. In Figure 7-26, the groundwater discharge to the river (i.e. flux response) is expressed as a fraction of the recharge. The increase in groundwater discharge to a river following an irrigation development can take many years to occur, particularly where the irrigation development is located a long distance from the river.

The flux response in Figure 7-26 is the groundwater discharge to the river, expressed as a fraction of the recharge. Because it is a fraction, it is unitless.

Interactions of groundwater as a result of neighbouring irrigation developments
The groundwater mounds that form under neighbouring irrigation developments have the potential to superimpose upon each other, resulting in higher groundwater levels than may otherwise occur. Figure 7-27 illustrates the variation in groundwater level beneath two small (500 ha) neighbouring irrigation developments at different distances of separation. Two sets of parameters are examined. The first assumes a saturated hydraulic conductivity of 1 m/day and a
recharge rate of 65 mm/year (Figure 7-27a). The second assumes a saturated hydraulic conductivity of 20 m/day and a recharge rate of 130 mm/year (Figure 7-27b). Both assume the irrigation developments are 1 km from a river. The former is considered to be more representative of mosaic irrigation developments associated with offstream storages or groundwater extraction from an underlying aquifer in the Mitchell catchment. The results indicate that small-size irrigation developments (i.e. 500 ha) exhibit very little interaction during the first 10 years after development. Small interactions occur within a 100-year time frame, but interactions can be avoided when the developments are placed 1 km apart. Placing 500-ha irrigation developments at least 5 km apart excludes any interaction (i.e. under steady-state conditions).

![Figure 7-27 Variation in watertable level beneath two neighbouring 500-ha irrigation developments at different distances of separation](image)

(a) Saturated hydraulic conductivity of 1 m/day, recharge of 65 mm/year and 1 km from river. (b) Saturated hydraulic conductivity of 20 m/day, recharge of 130 mm/year and 1 km from river.

**Scheme-scale developments**

This section illustrates how water levels may rise under a hypothetical scheme-scale irrigation development with parameters as shown in Table 7-13.

<table>
<thead>
<tr>
<th>AQUIFER PARAMETER</th>
<th>UNIT</th>
<th>VALUE</th>
<th>COMMENT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquifer thickness</td>
<td>m</td>
<td>12 na</td>
<td></td>
</tr>
<tr>
<td>Depth to groundwater</td>
<td>m</td>
<td>9</td>
<td>na</td>
</tr>
<tr>
<td>Distance of irrigation development boundary to river</td>
<td>km</td>
<td>1</td>
<td>na</td>
</tr>
<tr>
<td>Recharge rate</td>
<td>mm/y</td>
<td>67,118</td>
<td>Lower and higher estimate. Recharge as a result of both irrigation and rainfall</td>
</tr>
<tr>
<td>Saturated hydraulic conductivity (K)</td>
<td>m/day</td>
<td>1, 10, 100</td>
<td>Lower, middle and higher estimate, respectively</td>
</tr>
<tr>
<td>Specific yield</td>
<td>0.2</td>
<td>Only has bearing on rate of rise, not maximum rise</td>
<td></td>
</tr>
</tbody>
</table>

na = not applicable
Figure 7-28 indicates that under the low (1 m/day) and middle (10 m/day) values for saturated hydraulic conductivity, and under conditions where the depth to groundwater is 9 m below the ground surface, the watertable level under a 12,000-ha irrigation development reaches within 2 m of the ground surface in 10 to 25 years, depending on the recharge rate.

For the higher estimate of saturated hydraulic conductivity (100 m/day), the drainage capacity of the aquifer is higher, which results in a slower rise in watertable level. For the lower and higher recharge rates, the watertable level would approach to within 2 m of the ground surface in about 20 to 50 years.

In this hypothetical example any rise in watertable level is likely to mobilise soluble salts in the substrate and clay subsoils. This could potentially cause secondary salinisation when watertable level rises to within 2 m of the ground surface.

![Figure 7-28 Change in depth to watertable for different values of saturated hydraulic conductivity (K)](image)

(a) Low recharge rate of 67 mm/year and (b) high recharge rate of 118 mm/year.

7.6 References


Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.


Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.


