Review of Water Resource (Warrego, Paroo, Bulloo and Nebine) Plan 2003 and Resource Operations Plan

Environmental risk assessment for selected ecological assets

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Introduction

This report examines the impact of water resource development associated with the Water Resource (Warrego, Paroo, Bulloo and Nebine) Plan 2003 and Resource Operations Plan on the ecological assets identified in the Environmental Assessment–Stage 1, Appendix A–Ecological Asset Selection Report (DSITIA 2013a). A comprehensive review of information and knowledge relating to the critical flow requirements of ecological assets (expressed in terms of facets of the flow regime) forms the basis for the assessment. The analysis uses an ecological risk assessment approach based on daily time series flow outputs from the Integrated Quantity Quality Model (IQQM) for different water resource development scenarios.

This document describes the critical flow requirements, assessment and measurement end-points, and thresholds of concern identified for each of the prioritised ecological assets in the plan area, along with the methods and supporting information used to derive these. For each catchment, results of the ecological risk assessment, identifying changes in the provision of critical flow requirements under pre-development and full development flow scenarios, are presented.

General method for the environmental assessment

This environmental assessment uses an eco-hydraulic modelling approach, based on the principles of ecological risk assessment (ERA), to assess the risk to aquatic ecosystem components, processes, and services from the plan (Figure 1). In summary the assessment focuses on ecological assets that:

i. represent the ecological values of the plan area;
ii. are dependent on aspects of the flow regime; and
iii. are vulnerable to the types of flow alteration reflected in the water resource plan (WRP).

The assessment uses a desktop modelling approach drawing on existing information and knowledge on the ecological values of the plan area as well as relevant flow-ecology information in the broader scientific domain. The approach used is consistent with the Framework for assessing the Environmental Water Requirements of Groundwater Dependent Ecosystems Report 1 Assessment Toolbox (Clifton et al. 2007) and the principals outlined in Ecological Risk Assessment of Water Resource Plans (Marshall & McGregor 2006).

Representation of flow-dependent ecosystem components, processes and services using ecological assets

Predicting potential ecological responses to altered flow regimes is complicated by interactions between the flow and ecosystem components and processes at multiple scales. This is further confounded by effects of non-flow related stressors present in the system (i.e. land use, toxicants, etc.). Consequently general measures of ecological responses to managed flow regimes are rarely observed (Kennard et al. 2009; Poff & Zimmerman 2010). To deal with this uncertainty, a practical approach for managing flow regimes for specific ecological outcomes requires identifying and partitioning the critical flow dependencies of ecosystem components and processes, and consideration of their specific water requirements over time.

These components, processes, and services are effectively indicators of flow modification and therefore broadly representative of the ecosystem response. Known as ecological assets, they are highly valued components of the ecosystem for which aspects of the flow regime (i.e. duration, timing, variability, predictability, magnitude, rate of rise and fall) are critical to support their long term viability. Ecological assets may be a species, a group of species, a biological function, an ecosystem or a place of natural value. They occur in the area of interest, have an aspect(s) of life history or process requirement critically linked to the flow regime, and are sensitive to the nature of flow regime alteration relevant to the area of interest. Each WRP area contains a unique set of ecological assets and related ecological outcomes. Consequently ecological assets selected for each WRP area will differ across the state. Additionally, the flow requirements of a specific ecological asset may also vary between WRP areas due to the different eco-hydrologic settings that characterise each basin. The risk to ecological assets from water resource development represented by the draft WRP is the focus of the environmental assessment process.

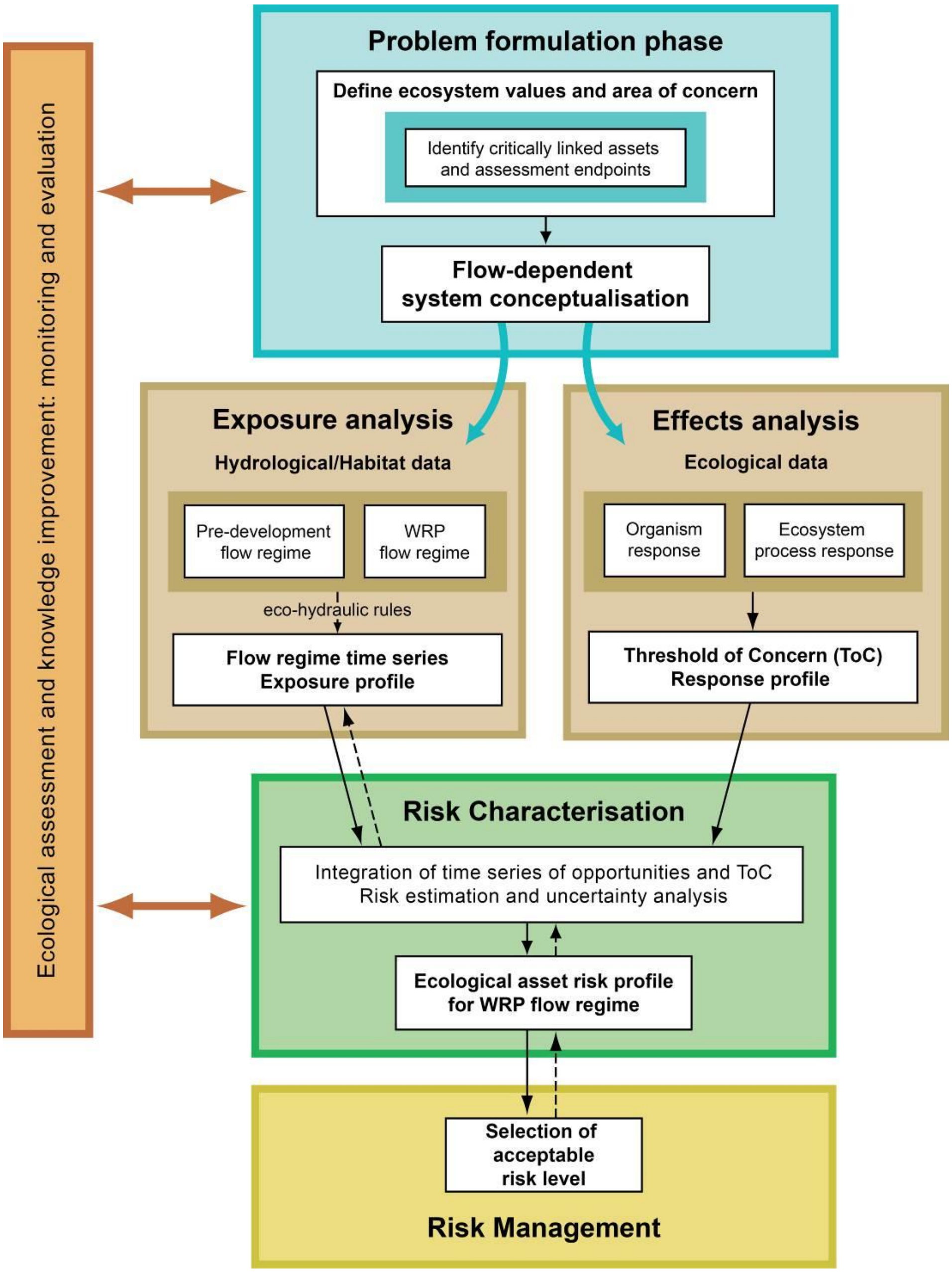

Figure 1: Ecological risk assessment framework for flow/groundwater dependent ecosystem components, processes, and services

Ecological asset identification

Ecological assets were selected following a comprehensive review of available data and information in the peer reviewed scientific literature, grey literature sources, government databases, and through consultation with relevant technical experts (see Water Resource (Warrego, Paroo, Bulloo and Nebine) Plan 2003, Environmental Assessment–Stage 1, Appendix A–Ecological Asset Selection Report, DSITIA 2013a). Ecological assets were categorised as either surface water or groundwater dependent.

Determining specific flow requirements

The environmental assessment uses an eco-hydraulic modelling approach to assess the risk to ecological assets from water resource development. Therefore, only those ecological assets with sufficiently detailed knowledge on their flow-ecology requirements (expressed in terms of facets of the flow regime) and supporting habitat data (in terms of waterhole bathymetry, stream cross sectional areas, etc.), are candidates for detailed quantitative risk analysis. A detailed review of the scientific literature and consultation with relevant technical experts was conducted for each of the candidate ecological assets identified in Water Resource (Warrego, Paroo, Bulloo and Nebine) Plan 2003, Environmental Assessment–Stage 1, Appendix A–Ecological Asset Selection Report (DSITIA 2013a). This information was distilled into discrete aspects of the flow regime with respect to location, timing, magnitude, duration, frequency, habitat features and associated water quality attributes where relevant. This expression of flow regime facets that support critical life history or process events, forms the basis for determining how the managed flow regime alters the provision of these opportunities over time, and hence represents a risk to the assets' long term viability.

Defining the assessment endpoints

Assessment endpoints are used to explicitly define the environmental values of concern and provide the focus for analysis and characterisation in ERAs. They can include species and life stages, multiple levels of organisation and numerous structural and functional attributes. Assessment endpoints are those characteristics/attributes of the valued ecological entity which are believed to be at risk (i.e. vulnerable). In this context the assessment endpoints are the flow-dependent ecological components, processes, and services of the Warrego, Paroo, Bulloo and Nebine WRP area. The assessment endpoint has two components: (i) the entity-which is the valued aspect of the ecosystem (i.e. ecological asset such as a fish, plant, turtle, waterhole, etc.); and (ii) the attribute of the entity-such as abundance, fecundity, recruitment, extirpation, persistence, etc. Assessment endpoints are generally estimated using measurement endpoints.

Measurement endpoints (also known as measures of effect) are expressions of observed or measured responses to the stressor, a measurable characteristic that is related to the assessment endpoint. Examples include; measures of fecundity, recruitment and survival. Measurement endpoints are derived via laboratory or field based observational studies that are used to estimate the effects on an assessment endpoint or exposure to a stressor. Measurement endpoints are typically the focus of the risk assessment and link the assessment endpoints to the risk assessment. When an assessment endpoint can be directly measured, the measurement and assessment endpoints are the same. In most cases, however, the assessment endpoint cannot be directly measured, so a measurement endpoint (or a suite of measurement endpoints) is selected that can be related, either qualitatively or quantitatively, to the assessment endpoint (USEPA 1992). For most of the species based ecological assets considered in this assessment, the measurement endpoints relate to spawning and recruitment opportunities linked to aspects of the flow regime. Measurement endpoints for ecological process- and service-related ecological assets vary; however typically they relate to the provision of critical habitat, or the conditions which support ecosystem structure and/or function.

Establishing critical flow requirements (eco-hydraulic rules)

The exposure analysis phase of the ERA (Figure 1) uses information on the flow requirements of the prioritised ecological assets to develop a time series of opportunities for each water resource development scenario as it relates to a specific ecological response defined as the measurement endpoint. The requirements of ecological assets in terms of aspects of the flow regime (e.g. magnitude, duration, timing, rate of change) are defined and expressed as eco-hydraulic rules. For each ecological asset, best current scientific understanding is used to describe the nature of the flow dependency by defining the flow related conditions needed to trigger an ecological response. This understanding is used to formulate eco-hydraulic rules which define an opportunity for the ecological response in terms of facets of the flow regime. For example:

spawning trigger flow = X magnitude for Y period between T_1–T_2 time of the year

These eco-hydraulic rules are then applied to a daily flow time series representing a water resource development scenario to generate a time series of opportunities for the ecological response. This likelihood or exposure data represents the probability of an ecological asset/indicator experiencing the critical conditions required, when and where they are needed over the assessment period.

Defining thresholds of concern (ToCs)

The effects analysis phase of the ERA (Figure 1) uses information on the consequence of altering the provision of the critically-linked response to the long term viability of the ecological asset. Defining what constitutes sufficient opportunities for an ecological response in order to maintain the viability of an ecological asset remains a global knowledge gap in the scientific literature for a great many flow dependent species and processes. The application of coarse hydrological metrics such as percentage of pre-development flow regime oversimplifies the temporal hydrological sequence experienced by the ecology. These statistics ignore the obvious relevance of timing, spell durations, and do not adequately represent the complex interactions between the hydrology and ecology. In this ERA flow context, consequence or effects data is the characterisation of an adverse ecological effect or response. Consequence is the impact on the valued attributes of an ecological asset/indicator of not providing the conditions it critically requires.

Ideally, defining sufficient opportunities is informed by response functions derived from controlled observational or manipulation-based studies of flow-ecology interactions. In the absence of widely applicable general response functions, best available science can be used to derive step functions or thresholds which represent critical change or failure points along a response gradient. In this WRP application, Thresholds of Concern (ToC) (*sensu* Rogers & Biggs 1999) are defined to represent the frequency of opportunities required to protect asset viability. ToCs represent failure points for the ecological asset and as such can be considered minimum water requirements. Therefore, the probability of achieving a desired ecological outcome is directly related to meeting a ToC over time. Where possible, ToCs are based on the biology or process knowledge of the asset. In most cases, ToCs represent the known time species-based ecological assets will survive without experiencing a flow-based opportunity (for responses related to maintenance and persistence dependencies) or the reproductive life time of the asset (for responses related to regeneration and recruitment dependencies). For those ecological assets without a clear life history basis for setting a ToC, thresholds can be related to the frequency of opportunity provision modelled to occur under the pre-development flow regime. Because even natural flow regimes are not without risk to ecological assets, the risk from management scenarios will be considered relative to the risk from the pre-development flow regime.

The process outlined above requires both a sound conceptual understanding of the flow dependent ecological assets and detailed biological and/or process knowledge relating to their critical flow dependencies. The synthesis of this knowledge is presented in this report for each ecological asset, or group of assets (guild) and forms the basis for the ecological model development and setting of ToCs, which will subsequently be used for the quantification of risk to those assets from various water resource development scenarios. The risk characterisation phase of the ERA (Figure 1) is presented in this report.

Assessing the relative risk to ecological assets from water resource development scenarios for surface water ecological assets

Two surface water development scenarios were assessed across the four plan area basins:

1. pre-development–assumes no water resource development in each catchment
2. full entitlement–reflects the full use of existing entitlements with current ROP operating rules–this scenario does not reflect the current utilisation of water entitlements.

Modelled daily flow time series

Simulated daily flow time series (as ML/day) were modelled for the pre-development and full entitlement scenarios at a series of environmental assessment nodes, representing stream flow gauging stations in the plan area, using the water resources management model–Integrated Quantity Quality Model (IQQM). The IQQM is a hydrological system simulation model which operates on a daily time step. A 122 year simulation period was modelled for the period 1889–2011 inclusive (DSITIA 2013a, b, c).

Time series of flow-related opportunities

The eco-hydraulic rules for each ecological asset were applied to the modelled daily flow time series (EcoModeller V 2.0.6, eWater CRC) to generate a time series of flow-related opportunities for the ecological response over the simulation period. This was undertaken for each of the two IQQM scenarios at each relevant node. Opportunities in these time series were represented by days in which all daily flow requirements defined by the eco-hydraulic rules were met.

Assessment of risk using ToC

At the node scale the risk to ecological assets was expressed as the percentage of years in the simulation period which were in either a low or high risk category as defined by the ToC for the asset being assessed (Table 1). Low risk events were periods in the time series of opportunities where the ToC was met by the flow scenario; whereas high risk events were represented by periods when the ToC was not met (otherwise referred to as a node failure). Where appropriate, for assets with multiple ToCs, node failures were refined into moderate or high risk events depending on which of the ToCs were not met.

For ecological assets where no ToC could be derived, because insufficient knowledge was available to do so, hazard rather than risk posed by the development scenarios was identified and discussed in relation to the proportional change in opportunities from the pre-development scenario. Because the modelling approach used for Yellowbelly incorporated meta-population dynamics at two spatial scales, risk to this asset was assessed at: (i) the environmental assessment node, and (ii) the catchment–scale.

The process outlined above requires both a sound conceptual understanding of the flow dependent ecological assets and detailed biological and/or process knowledge relating to their critical flow dependencies. Further details on the approach used for each asset are given in the relevant sections of this report.

Table 1: Summary of ecological asset indicators used in the environmental assessment, their link to hydrology and the ecological outcomes of the plan

Ecological asset indicator	Indicator measurement endpoint	Assessment nodes	WRP ecological outcome
Flow spawning fish	Annual and long-term abundance of Yellowbelly (*Macquaria ambigua*)	Warrego: 423004, 423005, 423201A, 423202C, 423203A, 423204A, 423206A Paroo: 424202A, 424201A Bulloo: 011203A, 011202A, 011201A Nebine: 422501A, 422502A	9f (i, iii, vi, vii)
Migratory fish species	Frequency of longitudinal dispersal opportunities	Warrego: 423004, 423005, 423201A, 423202C, 423203A, 423204A, 423206A Nebine: 422502A	9f (i, iii, vi, vii)
Eastern snake-necked turtle (*Chelodina longicollis*)	Frequency of high stress events	Warrego: 423004, 423005, 423202C, 423203A, 423204A	9f (i, iii, vi, vii), 9j
Absence of exotic fish species	Frequency of strong recruitment opportunities for the European carp (*Cyprinus carpio*)	Warrego: 423004, 423005, 423202C, 423203A, 423204A Nebine: 422502A	9f (iii, vii)
Floodplain vegetation	Length of spells between floodplain vegetation inundation events	Warrego: 423004, 423005, 423202C, 423203A, 423204A, 423206A Nebine: 422502A	9f (iii, vi, vii), 9j
Floodplain wetlands	Length of spells between floodplain wetland inundation events	Warrego: 423004, 423005, 423202C, 423203A, 423204A, 423206A Nebine: 422502A	9f (ii, iii, v, vii), 9g, 9j
Unique genetic diversity of aquatic plants and animals within the Bulloo basin	Movement of water between river catchments	Bulloo: catchment scale assessment	9f (vi)
Waterholes as refugia	No-flow spells Distance between waterholes	Warrego: 423004, 423005, 423201A, 423202C, 423203A, 423204A, 423206A Paroo: 424202A, 424201A Bulloo: 011203A, 011202A, 011201A Nebine: 422502A	9f (i, ii, iii, iv, vii), 9g

Ecological asset indicator	Indicator measurement endpoint	Assessment nodes	WRP ecological outcome
Fluvial geomorphology and river forming processes	Frequency of bankfull flow events	Warrego: 423004, 423005, 423201A, 423202C, 423203A, 423204A, 423206A	9f (iv)

Assessment assumptions and limitations

The assessment was conducted as a desktop study using existing information and scientific knowledge on the flow-dependent and groundwater dependent ecological assets of the Warrego, Paroo, Bulloo, and Nebine catchments and their likely response to water resource development. Given these terms of reference, the assessment was underpinned by a core set of assumptions and limitations:

1. the suite of ecological assets being assessed broadly represents the ecological components, processes and services present in the plan area that are potentially vulnerable to water resource development;
2. risk is expressed in relative terms between development scenarios and not in absolute terms.
3. risk to assets as presented here is the risk from water-resource development only and not total risk from all sources; because non-water resource development pressures (i.e. land use, contaminants, instream modification, etc.) also contribute to the risk profile for any given ecological asset, yet the plan manages only water-resource development pressures;
4. interactions between ecological assets and their response to the water resource development scenarios are not explicitly considered;
5. the environmental assessment nodes are representative of the stream network which is influenced by water resource development; and
6. the modelled flow scenarios accurately represent the hydrological regime produced by management options as framed in a water resource plan or resource operation plan.

Assets selected for the assessment

Of the potential surface water dependent ecological assets identified for the plan area (DSITIA 2013a), nine had sufficient information available concerning their specific ecological flow requirements for quantitative modelling as described above for this environmental assessment (Table 2). They included ecosystem components such as flow spawning, migratory and exotic fish species, snake-necked turtle, and floodplain vegetation and wetlands and ecosystem processes including the function of waterholes as refugia, river forming processes and the genetic diversity of the aquatic biota of the Bulloo catchment.

Table 2: Surface water ecological assets used in the environmental assessment, their link to hydrology, assessment endpoints, and their distribution in the plan area

Ecological asset	Assessment endpoint	Link to hydrology				Catchment			
		No-flow	Low flows	Medium flows	High flows	Warrego	Paroo	Bulloo	Nebine
Ecosystem components									
Flow spawning fish	Population viability of Yellowbelly (*Macquaria ambigua*)	✓		✓	✓	✓	✓	✓	✓
Migratory fish species	Maintenance of movement opportunities for migratory fish species	✓		✓		✓			✓
Eastern snake-necked turtle (*Chelodina longicollis*)	Population viability of Eastern snake-necked turtle (*Chelodina longicollis*)	✓			✓	✓			✓
Absence of exotic fish species	Minimised abundance and distribution of European carp (*Cyprinus carpio*)			✓	✓	✓			✓
Floodplain vegetation	Viability of floodplain vegetation communities				✓	✓			✓
Floodplain wetlands	Maintenance of wetting regime to support floodplain wetlands				✓	✓			✓
Genetic diversity of aquatic biota in the Bulloo	Absence of translocated genotypes of Yellowbelly in the Bulloo							✓	
Ecosystem processes									
Waterholes as refugia	Maintenance of appropriate spatial distribution and connectivity of permanent waterholes	✓	✓			✓	✓	✓	✓
Fluvial geomorphology and river forming processes	Maintenance of river forming processes			✓	✓	✓			

Risk assessment methods for ecological assets

Flow spawning fish species–Yellowbelly (*Macquaria ambigua*)

Background

Yellowbelly (*Macquaria ambigua*) is a representative of a guild of flow spawning fish species in the plan area (DSITIA 2013a) and has been selected as an indicator of this guild for ecological assessment. It is highly regarded by recreational fishers and is thus one of the most highly valued fish species of the northern Murray-Darling Basin and Bulloo River. It is a moderate to large fish growing up to 760 mm length and 23 kg in weight, but more commonly 400–500 mm long, and less than 5 kg (Pusey et al. 2004).

Distribution

Genetic studies indicate that there are four subspecies of Yellowbelly occupying respectively the Fitzroy, Murray-Darling Basin (MDB), Bulloo Basin and Lake Eyre Basin (LEB) (Faulks et al. 2010). The sub-species inhabiting the Bulloo basin has a closer affinity to the LEB sub-species than to the sub-species from the MDB, a finding which is corroborated by comparative studies with other fish species and evolutionary boundaries, and suggests bigger and older barriers to migration between the Bulloo and Paroo and Warrego, than with the LEB (Faulks et al. 2010). The Paroo and Warrego populations are connected during flood events through the Cuttaburra Creek channel and other links between Cunnamulla and Eulo, and Cunnamulla and Caiwarro. The Warrego and Nebine populations are also connected when rivers draining into the Darling River experience large floods, and may also be connected more directly via flood channels, but there is a lack of direct observation of such connections. Although Yellowbelly is currently common throughout its natural range, it is reported to have declined in abundance in the Murray-Darling over recent decades (Allen et al. 2002; Moffatt & Voller 2002).

Habitat

Mature fish of both subspecies in the plan area occur in a variety of riverine habitats but prefer warm, slow-moving, turbid sections of rivers (Table 3). They are also found in flooded lakes, backwaters and impoundments (Allen et al. 2002). They have strong associations with woody debris, deep pools of low velocity and sandy substrates (Pusey et al. 2004).

Rises in water level and temperature associated with first post-winter flow events trigger annual spawning migrations. Nutrient transfers from floodplains and overland runoff may also be important components of this reproductive cue. The species is not a fully facultative flow spawner, and offspring have been found in the absence of flood events (Balcombe et al. 2006). However, monitoring program results indicate increased abundances following small and medium-sized flood events, while large floods tend to lead to marked decreases in abundance (Hagedoorn & Smallwood 2007).

Juveniles are associated with shallow, inundated floodplain habitat, and have been collected from flooded backwaters (Pusey et al. 2004), but the strength of this association still has to be determined. Like many other species of fish, Yellowbelly is cannibalistic, which may explain juvenile habitat preferences for backwaters, and a tendency for juveniles to avoid deep water frequented by adults and other piscivores.

In the channels of the northern MDB, lagoon-like, large, terminal depressions associated with flood break-out channels also provide floodplain habitat. Due to the often large distances of these features from main channels, it is difficult for juvenile fish to return to the main channels, so much so that a reduction in recruitment in the Warrego River has been observed for the years when breakout channels (like the Cuttaburra Creek channel linking to the Yantabulla lagoon, and other shallow features) are flooding (Queensland Government, unpublished data).

Table 3: Physicochemical preferences of the MDB and LEB sub-species of *Macquaria ambigua*. Information on the Bulloo sub-species is scant, so it is represented by the LEB population on the basis of their close genetic similarity (after Pusey et al. 2004).

Parameter	Min	Max
MDB (n=100)		
Temperature (°C)	4.0	35.0
Dissolved oxygen (mg/L)	3.0	15.0
pH	7.1	7.8
Conductivity (µS/cm)	224	3000
Secchi depth (cm)	12	240
LEB (n=228)		
Temperature (°C)	25.0	33.0
Dissolved oxygen (mg/L)	1.1	6.8
Conductivity (µS/cm)	90	144
Secchi depth (cm)	1.5	15

Growth and reproduction

Life history attributes of Yellowbelly are summarised in Table 4.

Table 4: Life history attributes of *Macquaria ambigua*, the sub-species occupying the Murray-Darling Basin.

Characteristic	Description
Longevity/lifespan	Individuals of 19+ years and 26 years have been found in the Murray-Darling. (Pusey et al. 2004).
Age at sexual maturity	Maturation of Murray River populations is reported at 2–3 years in males, and 4 years in females (Mallen-Cooper & Stuart 2003).
Sex ratio	To be determined. Previous studies have identified biases of 2.2:1 for males (Bice 2010), contrasting with Ferguson and Ye (2012), who found ratios of 1:2.7 and 1:3.6, for Southern Australian locations.
Peak spawning season	Pusey et al. (2004) (C = 3.3) suggest September to April. Lake (1967a) suggests that there isn't a spawning season as such but rather that spawning occurs whenever the temperature is suitable (>23.6 ^{o}C) and there is a corresponding rise in water level. Gilligan and Schiller (2003) reported evidence of spawning at temperatures as low as 17 ^{o}C. Spawning dates with isolated spawning at lower temperatures than 23.6 ^{o}C have also been backdated from otolith-aged fish for the Weir River (Queensland Government, unpublished data).
Spawning frequency	While gonad maturity is maintained over an extended period, approximately from September to March or April (Pusey et al. 2004), most juvenile age determinations link spawning to a single summer flood event (T. Khan, unpublished monitoring data). There is a statistically significant relationship between the height of the biggest summer flood and the annual recruitment rate in all examined Northern MDB basins, also suggesting mechanisms to focus reproductive efforts on the biggest annual flood (data by Hagedoorn & Smallwood 2007, and Balcombe unpublished data). If conditions are less than favourable females may release only a fraction of their eggs (Battaglene & Callanan 1991). Fish failing to spawn resorb their gonads by involution, usually in February/March in the Murray-Darling (Pusey et al. 2004).
Spawning cues	Multiple spawning cues have been suggested over the decades, including seemingly contradictory flood-spawning and low-flow spawning (Pusey et al. 2004). The Murray Darling Basin, to which these observations apply, is affected by a range of seasonal patterns, from predominant winter rain to predominant monsoonal/summer rain (CSIRO 2008). These partially conflicting reports on the behaviour of the species are consistent with these climatic regimes. Reports which link spawning cues to the subsequent food availability of larvae appear relevant. Mechanisms leading to the food availability are raised water temperatures, as well as mobilisation of nutrients through surface runoff and floodplain inundation, occasionally referred to as priming events (Bernie Cockayne, pers. comm.). Most relevant for understanding the Northern Murray Darling Basin populations are observations of spawning evidence from aging of eggs, larvae and juveniles from the Weir River over multiple years, which show that 94% of recruits were spawned in conjunction with first post winter flow spawning cues (DERM 2010a). The cues consist of temperatures above 23 degrees during some of the event, and water level rises of more than 0.65 m in a short period of time (Pusey et al. 2004).
Fecundity	300 000–500 000 eggs per female per spawning event (Allen et al. 2002).
Spawning migration	Spawning is believed to be preceded by substantial up- and downstream migration, where generally an initial upstream migration is expected (Pusey et al. 2004). In the southern MDB females migrate greater distances than males, with 3% of fish (mean length 41.7 cm) migrating over 1000 km as observed (Reynolds 1983). Migration distances in the Moonie River in the northern MDB over three years were found to be much smaller. While maximum migration distances up to the size of the monitoring network of 80 km were observed, the participation rate was less than 5%, and the median of migration distances was found to be 20 km (DERM 2010b, DSITIA unpublished data).
Critical physical/chemical attributes required at breeding site	Optimal water temperatures for spawning are above 23ºC, but spawning also appears to happen at lower temperatures (Pusey et al. 2004; DERM 2010a). The species is tolerant of highly turbid and saline waters (Pusey et al. 2004).

Characteristic	Description
Critical physical/chemical attributes for larval development	O'Connor et al. (2003) suggested that downstream spawning migrations at or close to the peak of high water would be likely to place larvae on the floodplain at times of maximum inundation, and maximum exposure to larval food supplies.
Egg characteristics	Mature oocytes are spherical, with a diameter of 1.1 mm. Fertilised and water-hardened eggs are on average 3.9 mm in diameter. The eggs of Yellowbelly are pelagic and drift with the current for their short incubation period of 24–46 hours (Lake 1967b).
Time to hatching	The time to hatching is generally about 18 hours but may be as short as 12 hours, dependent on water temperature (Pusey et al. 2004).
Larval development	Time to hatching is 33–34 hours at 24–25^{O}C. The length at hatching is on average 3.2 mm. Length at first feeding is 5.9 mm total length. Age at first feeding is at 6 days, and teeth are evident at 10 days, at 7 mm length. At 5–6 days the yolk is almost completely absorbed. Duration of larval development is 18–20 days, and the length at metamorphosis is 9.5–11.5 mm (Lake 1967b). During the first 25 days larvae show little ability of swimming against the flow (Gehrke 1990).

Ecological value supported by Yellowbelly

Yellowbelly is one of a guild of flow spawning fish species in the plan area (DSITIA 2013a). It has been selected as an indicator of this guild on the basis that scientific understanding of its flow requirements for spawning is superior to other guild members. Results from the risk assessment applied to Yellowbelly will generally represent risks to the other species in the guild, but these should be used with caution, as further research is required to identify how nuances of the specific eco-hydraulic requirements for flooding may vary from species to species. Flow spawning fish are linked to a number of ecological outcomes in the plan (9f (i, iii, vi, vii)).

Yellowbelly is an iconic species that is highly prized as an angling fish, even supporting a small commercial fishery in the Murray-Darling system outside the plan area (Allen et al. 2002). The current abundance, distribution and sustainability of populations are attributes valued by the community.

Yellowbelly is not currently listed as threatened; however their long-term persistence may be negatively impacted as a result of human activities, including water resource development (Pusey et al. 2004).

Spatial relevance

Yellowbelly is present in all catchments of the northern Murray-Darling Basin and the Bulloo. While juvenile fish are assumed to utilise more marginal, shallower habitat, adult fish are expected in the main channels of rivers.

Assessment endpoint

Viability of Yellowbelly populations within the plan area catchments.

Measurement endpoint

Annual and long term average abundance of Yellowbelly.

Eco-hydraulic rules

Because Yellowbelly are both highly valued and their eco-hydraulic requirements for population viability comparatively well understood, a more rigorous and comprehensive modelling approach was applied to this asset than to the others used in this assessment. Risk was assessed using spatially and temporally explicit meta-population models within each plan catchment, which generate annual time-series of Yellowbelly population abundance at the assessment node and catchment scales.

These models integrate eco-hydraulic habitat elements derived from pre-development and full entitlement IQQM daily flow scenarios and waterhole pumping licence locations and conditions under the full-entitlement scenario including:

- First post-winter flow: the size of annual flow spawning events calculated as a function of flow size between beginning of October and end of April.
- Waterhole persistence and size: time series of habitat, representing the persistence of waterholes based on spells between flow events, where each year is characterised according to the likely waterhole sizes as a function of natural flow patterns.
- Connectivity between waterholes and between channels: calculated implicitly based on the spatial availability of waterholes and the migration behavioural parameters of the species.
- Waterhole pumping (i.e. allocations with a nil passing flow condition): calculated as permitted under water licensing entitlements of the current plan and the effects of this on both waterhole persistence and connectivity.

Using meta-population models for population viability assessments

Risk to Yellowbelly populations was assessed using the RAMAS meta-population modelling software (Akçakaya et al. 1999; Akçakaya 2005). A meta-population is a population of a species which consists of multiple sub-populations—otherwise known as local populations—which are defined as a consequence of temporally dynamic physical separation. Individual sup-populations (e.g. at an assessment node) may be at greater risk from water resource development than the whole population. Migration between sup-populations may re-populate vacated habitat patches, dependent on connectivity providing migration pathways, or may stabilise local populations. Such spatial and temporal dynamics are best represented using a meta-population modelling approach.

RAMAS software represents measures of key characteristics of population dynamics using matrix modelling, and is widely accepted as a tool for a variety of population management purposes throughout the world (Akçakaya et al. 2004). Models are developed for a species by integrating data sources and expert knowledge concerning its life history traits and habitat requirements. A process of sensitivity testing is applied (Curtis & Naujokaitis-Lewis 2008) to evaluate the realism of the assessment models, and guide model development for assessments in the future. These models output quantitative population abundance simulations with an annual time-step for the overall population of a catchment as well as for sub-populations in areas of interest at assessment nodes, under the conditions provided by each flow scenario tested. At each yearly time step RAMAS was run with 200 stochastically generated starting configurations which generated an average annual Yellowbelly abundance estimate. The standard deviation of the annual simulation results are provided as a measure of confidence and variability about that average model results. Model parameter values were stochastically varied according to observed and literature values of uncertainty. This randomisation process produces more robust results and allows more confidence of model interpretation. Abundances represent the number of adults in the population at each time-step.

The conceptual model (Figure 2) illustrates the main elements of the meta-population simulation model, with emphasis on its sensitivity to water management scenarios. There are additional components required to model freshwater fish populations (e.g. Nicol & Todd 2004). In general terms, the approach used in this assessment uses the following components to represent annual Yellowbelly population abundance:

1. Hydrology

   Daily stream flow IQQM simulations, waterhole pumping locations and licence rules; and information on the spatial distribution and patterns of connectivity of waterholes throughout the system.

2. Yellowbelly biology

   Population traits (e.g. recruitment, survival, age class structure, etc.), and species traits (e.g. migration behaviours, habitat preferences, fecundity, etc.).

This information is combined to simulate time series of habitat availability, and reproductive and migration opportunities, which in turn are inputs to the metapopulation model. Outputs of this model are expressed in terms of an annual abundance of Yellowbelly at the node and catchment scales.

A summary of the population parameters and information sources used in the model is presented in Appendix 1.

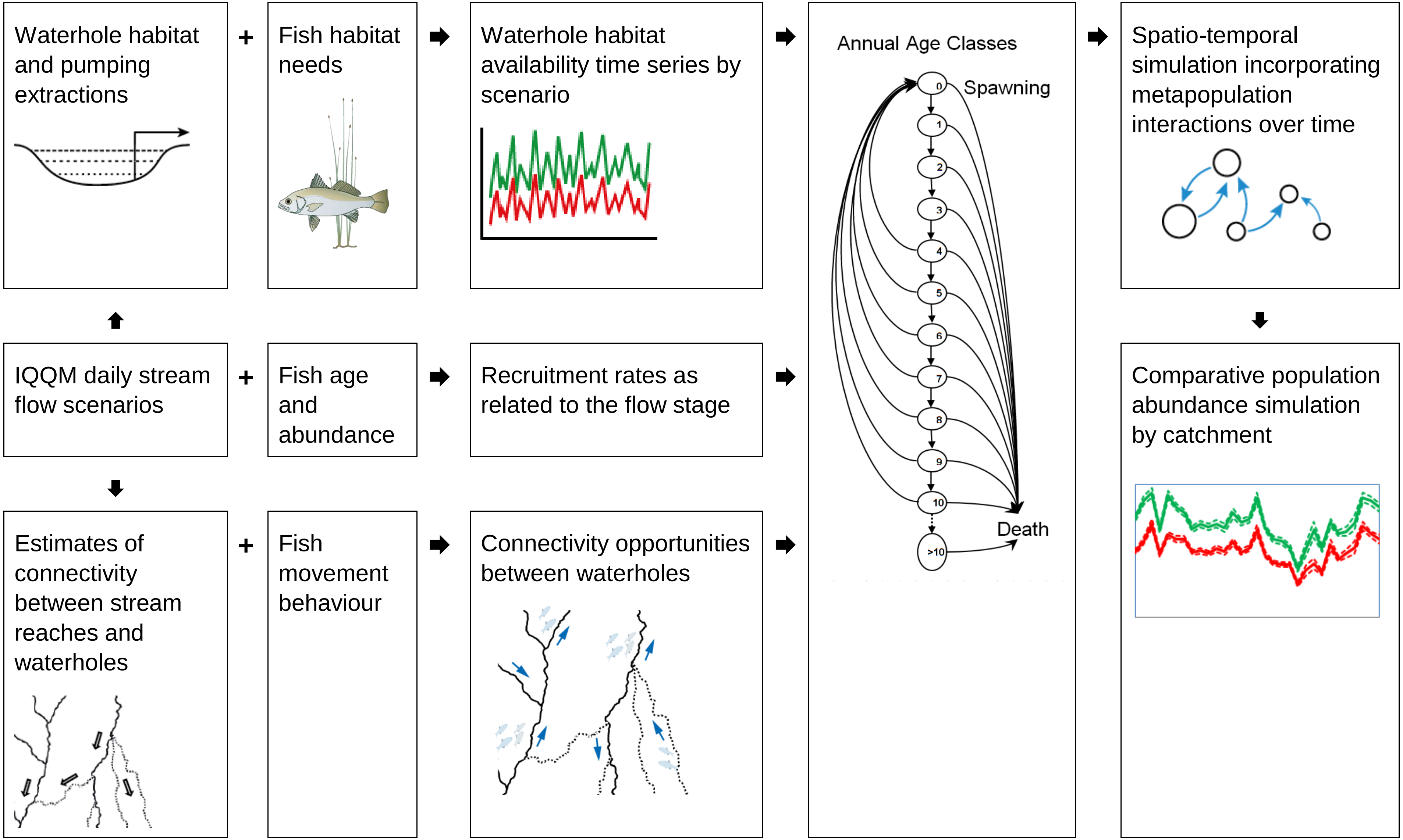

Figure 2: Conceptual model of the Yellowbelly meta-population model for the Warrego, Paroo, Bulloo and Nebine rivers.

Threshold of concern

A ToC of 5000 adult individuals was selected to represent the minimum abundance to maintain population viability at the catchment scale. This is also known as minimum viable population size (MVP), and is applied to the time-series of annual adult population abundances produced by the RAMAS models.

A systematic analysis of published estimates of population viability spread over 30 years of research identified a minimum viable population size of approximately 5000 adults for a wide range of taxonomic groups (Traill et al. 2007). When compared with MVP estimates for freshwater fish, this figure is consistent, and a threshold of 5000 adults is also supported by findings from Reed et al. (2003) for vertebrates, and genetic studies by Frankham (1995). Subsequently, Traill et al. (2010) have proposed 5000 adults as a suitable estimate for minimum viable population size in the absence of more specific information.

A value of 5000 adults is used as the minimum viable population size for the purposes of the risk assessment. This threshold of concern will be applied to each assessment catchment. In the case of the Bulloo, this applies to the endemic sub-species of the Bulloo Yellowbelly. Here the species extent and assessment area are identical. The Yellowbelly of the Paroo, Warrego and Nebine all belong to the Murray Darling sub-species. Therefore the Yellowbelly populations for the eastern catchments extend beyond the assessment basins, and consequently the assessments, applied to each sub-population separately, are more conservative than commonly used in the literature.

Migratory fish species

Background

Fish move over a variety of spatial scales and for a range of biological and ecological incentives (Pusey et al. 2004). Some fish display strong site fidelity (Crook 2004a) and may only move within their own home range, which can be as small as a single pool or riffle (Pusey et al. 2004) while others may undertake large movements throughout stream networks (Humphries et al. 1999) and between marine, estuarine and freshwater habitat (Myers 1949; McDowall 1988). There are two major types of migration in freshwater fishes of Australia; species that migrate between the sea and freshwater (diadromous) and species that migrate wholly within freshwater (potamodromous) (Mallen-Cooper 1999). Of the 26 native species within the Murray-Darling Basin (Cadwallander & Lawrence 1990, cited in Humphries et al. 1999) eight are considered to be potamodromous migratory species and none within the plan area are diadromous (Table 5).

Ecological significance

Potamodromous migrations occur for either spawning or dispersal reasons (Mallen-Cooper 1999). The spawning migrations undertaken by species within the WRP area can be described as 'habitat specific' (Mallen-Cooper 1999) meaning that the site of spawning may vary but the habitat conditions required of a spawning location remain the same, (e.g. peak of a flood or recently inundated ground). Upstream spawning migrations are also thought to be a strategy that compensates for the subsequent passive downstream drift of eggs and larvae. For example, the large distances travelled by certain species may compensate for passive downstream dispersal of their relatively small and buoyant eggs, embryos and larvae (Humphries et al.1999).

Dispersal is often used to explain migrations, and it is sometimes used where the reason for the migrations are not fully understood (Mallen-Cooper 1999). Upstream dispersal may be a mechanism for maximising distribution and gene flow, factors that are significant for colonisation after droughts (Mallen-Cooper 1999), or to reduce intra- and inter-species competition for limited resources (Pusey et al. 2004). In dryland rivers such as those in the plan area, migrations following a spell of isolation may also be particularly important as a mechanism of recolonising previously dry habitat patches (i.e. population resilience) from populations in refuge waterholes.

Migratory movements may be an important life history strategy of fish and can occur in the larval, juvenile or adult phase of development (Pusey et al. 2004). A number of species are known to undertake migrations in the Murray-Darling river system (Table 5).

The factors that trigger migration for most species in the Murray-Darling basin are not well known however the literature suggest that a seasonal rise in water levels (particularly the first post winter flows), increasing temperature and longer day length (Humphries et al. 1999; Mallen-Cooper 1999; Pusey et al. 2004) may trigger movement in some species. Migration traits vary widely among species and may not be consistent for one species across different areas (Humphries et al. 1999).

Fish migration behaviour has been studied for two species (Yellowbelly and tandanus catfish) over three years in the Moonie River, part of the northern Murray-Darling basin that exhibits similar hydrology to the plan area (DERM 2010b). Most individuals of both species moved between previously isolated waterholes during flow events that provided longitudinal connectivity. There was no clear upstream or downstream directional preference and most individuals utilised a reach of approximately 20 km, though some individuals ranged more than 70 km in only several days. Timing of flow was more important than magnitude, as most movement occurred in response to the first post-winter flow event independent of its magnitude and duration. Many of the fish that moved returned to their starting waterhole either by the end of an event, or on subsequent events during a season, suggesting ability to home and a preference for more permanent refuge pools. These findings highlight that fish in these systems utilise networks of waterholes and that management should aim to maintain movement opportunities at large spatial scales to preserve population resilience.

Table 5: Fish species recorded moving upstream in the Murray-Darling river system (after Mallen-Cooper 1999)

Species name	Common name	Migratory Stage		
		adults	sub-adults	juveniles
Macquaria ambigua	Yellowbelly	+	+	
Macquaria sp.	Bulloo Yellowbelly	?	?	
Bidyanus bidyanus	Silver perch	+	+	
Leiopotherapon unicolour	Spangled perch	+		+
Tandanus tandanus	Tandanus catfish	+	?	?
Neosiliurus hyrtlii	Hyrtl's tandan	+	?	?
Porochilus argenteus	Silver tandan	?	?	
Scortum barcoo	Barcoo grunter	?	?	
Bidyanus welchi	Welch's grunter	?	?	
Cyprinus carpio	European carp	+	+	+

? indicates unconfirmed

Distribution

Nine migratory fish species, including eight native species and one alien species are found within the plan area. All are considered to be potamodromous migratory species (Table 6).

Table 6: Migratory fish species and their distributions (DERM 2012) within the plan area (W = Warrego, P = Paroo, B = Bulloo, N = Nebine).

Species name	Common name	Distribution
Macquaria ambigua	Yellowbelly	W, P, N
Macquaria sp.	Bulloo Yellowbelly	B
Bidyanus bidyanus	Silver perch	W, P, N
Leiopotherapon unicolour	Spangled perch	W, P, B, N
Tandanus tandanus	Tandanus catfish	W, P, B, N
Neosiliurus hyrtlii	Hyrtl's tandan	W, P, B, N
Porochilus argenteus	Silver tandan	B
Scortum barcoo	Barcoo grunter	B
Bidyanus welchi	Welch's grunter	B
Cyprinus carpio	European carp	W, P, N

Ecological value it supports

Migratory fish species represent a significant component of the freshwater biodiversity of rivers and streams of the four catchments which comprise the plan area. None are currently listed under any state or commonwealth government legislation. Their persistence is linked to a number of ecological outcomes in the plan (9f (i, iii, vi, vii)).

Assessment end point

The maintenance of opportunities for movement by migratory fish species within the Warrego, Paroo, Bulloo, and Nebine catchments, to support population viability in these dryland rivers.

Measurement end point

Frequency and duration of flow events that provide longitudinal connectivity.

Eco-hydraulic rules

Modelled stream discharge of ≥ 2 ML/day was adopted to represent the threshold at which flow in the channel begins (Craig Johansen, pers. comm.). The minimum event duration that represents an opportunity for migration between suitable habitats was determined to be eight days based on movement studies (see Flow-spawning fish methods above; DERM 2010b).

The following eco-hydraulic rule was used to assess the risk to longitudinal connectivity.

- A minimum eight day event duration of discharge ≥ 2 ML/day

Threshold of concern

ToCs were defined to establish the consequence of altering longitudinal connectivity for migratory fish based on the results of otolith analysis of Yellowbelly populations in the Border Rivers (DERM 2010a). As with the flow spawning fish guild, Yellowbelly has been selected as an indicator of the migratory fish guild on the basis that scientific understanding of its flow requirements and life history in the plan area is superior to other guild members.

Analysis of age structure of Yellowbelly in northern Murray-Darling Basin rivers in which Yellowbelly have frequent opportunities for connectivity (DERM unpublished data), showed that the maximum age of Yellowbelly for these populations was ten years. Therefore it is critical for these species to have longitudinal connectivity opportunities within this time-span to facilitate spawning and recruitment. Furthermore, 75% of individuals in the population were less than four years old. Therefore, after a spell of four years without opportunity to migrate and spawn, only 25% of the potential reproductive population would persist to repopulate and recolonise the broader river system during a subsequent event. Collectively this information suggests that the longer duration between longitudinal connectivity opportunities, the greater the risk to the resilience and hence viability of the population. Based on this information, two ToCs for waterhole connectivity events were set in order to maintain population viability of Yellowbelly (representing the migratory fish guild) in the system–4 years and 10 years (Table 7).

Table 7: Risk categories for migratory fish*

Low risk	Moderate risk	High risk
< 4 years	4–10 years	> 10 years

* number of consecutive years without longitudinal connectivity opportunities

Aspects of hydrology

Opportunities for fish migration have direct links to both the frequency and duration of no-flow spells and medium flow events.

Spatial relevance

Migratory fish species are present in all four catchments of the plan area.

Eastern snake-necked turtle (*Chelodina longicollis*)

Background

Chelodina longicollis is a moderate-sized fresh water turtle; individuals can grow to over 250 mm in length and weigh over 1.5 kg. *C. longicollis* is a member of the subgenus *Chelodina*, and as such, the length of its head and neck is not as prominent as many other long-necked turtles, being equal to, or slightly less than, the length of the carapace (Cann 1998; Georges & Thomson 2010). The plastron is broad, covering the anterior orifice of the shell in ventral view so that the limbs, head and neck are not visible when withdrawn which is a diagnostic feature of this species. The carapace is brown, dark brown or black above. The sutures of the cream or yellow plastron are broadly edged with black; posterior marginal scutes elevated medially to accommodate the tail; which is another diagnostic feature of this species.

Distribution

The distribution of *C. longicollis* includes the Murray-Darling basin, coastal rivers and larger offshore islands from Eyre Peninsula west of Adelaide in South Australia to the Burdekin drainage of north Queensland, and the headwaters of the Cooper Creek drainage. Populations in northwest Tasmania are presumed to be introduced (Georges & Thomson 2010). Within the plan area *C. longicollis* is present in the Warrego, Paroo, Bulloo and Nebine catchments (Cann 1998; DERM 2012; DSITIA unpublished data).

Habitat

C. longicollis occurs in both permanent and ephemeral water bodies (Chessman 1988; Kennett & Georges 1990), and in both clear and highly turbid water (Chessman 1983, 1988). In a study in the Murray valley, this species occurred in all habitats surveyed including rivers, backwaters, oxbows, anabranches, ponds, rain pools and swamps (Chessman 1988). It has also been recorded in farm dams and drainage channels with dense macrophyte growth (Cann 1998).

C. longicollis is vagile with great propensity for overland migration and is capable of travelling overland up to 2.5 km when off-stream wetlands start filling (Kennett & Georges 1990). It has several adaptations to minimise water-loss during terrestrial movements in areas with unpredictable flow regimes including the capacity to: (i) store and reabsorb water from the cloacal bladder, (ii) adjust uric acid excretions, (iii) limit cutaneous water loss, and (iv) conserve water by burying in soil and debris (Roe et al. 2008).

When ephemeral off-stream water bodies dry or food resources in those water bodies become limited, *C. longicollis* is capable of terrestrial aestivation (Chessman 1983). Aestivation is a state of torpor similar to hibernation in which metabolic activity is physiologically minimised to conserve energy (fat) and water reserves (Cann 1998; Roe & Georges 2008). Maximum aestivation time is likely to vary between individuals depending upon initial body condition, but is generally thought to be approximately seven months (Roe et al. 2009). After seven months of aestivation, if conditions in off-stream wetlands remain unfavourable, individuals must migrate overland seeking water in permanent wetlands or waterholes in the main river channels (Kennett & Georges 1995; Roe et al. 2009). This is a time of high stress and high mortality for a turtle population (Arthur Georges pers. comm.).

Feeding

C. longicollis is an obligate but opportunistic carnivore, eating virtually any prey it can catch. It can feed only in the water and hence diet consists mostly of aquatic species. However they will also take terrestrial prey that has fallen into the water (Chessman 1983). Its broad diet includes zooplankton, benthic macro-invertebrates, frog eggs (Georges et al. 1986; Kennett & Georges 1990) and carrion (Chessman 1983; Georges et al. 1986).

When such habitats are available, *C. longicollis* prefer to exploit the high productivity of ephemeral water bodies such as floodplain billabongs and wetlands. Another advantage of feeding in ephemeral water bodies is that those habitats generally lack large predatory fish, which are significant competitors of the turtles for invertebrate prey (Chessman 1988; Kennett & Georges 1990). When ephemeral wetlands dry and turtles must return to permanent water bodies, *C. longicollis* compete with each other in high densities and with fish and other aquatic species for limited food resources. Consequently they lose substantial body condition during this time (Kennett & Georges 1990).

Poor body condition resulting from food limitation is known to reduce turtle reproduction rates. For example, *C. longicollis* populations in the Jervis Bay area (NSW) were subjected to food limitations during drought conditions between 1979 and 1983 (Kennett & Georges 1990). Due to the loss of condition (i.e. loss of body fat reserves) experienced by these individuals during this time, after drought conditions ended the populations failed to reproduce for 10 years (Arthur Georges pers. comm.).

Reproduction

Males mature at a smaller size than females and attain a smaller maximum size (Parmenter 1985). Male *C. longicollis* mature between the ages of 7 to 12 years whereas sexual maturity for females is attained between 11 and 15 years (Burgin et al. 1999; Arthur Georges pers. comm.). *C. longicollis* mating has been observed during April and May (Cann 1998), but varies over the range of the species (Parmenter 1985; Cann 1998). Parmenter (1985) observed mating in September in the vicinity of Armidale, New South Wales.

Clutch sizes range from 8–12 hard-shelled eggs (Parmenter 1985; Cann 1998). Multiple clutch production within a single reproductive season is possible and varies between locations from one per season up to a maximum three per season (Chessman 1978; Parmenter 1985). In a comparison of studies Parmenter (1985) found that populations of *C. longicollis* from higher latitudes produce fewer, larger clutches. Clutch size has also been shown to vary between habitat types. Clutches from dams and lagoons in the Armidale region were significantly larger than those found adjacent to running streams (Parmenter 1976). Larger females are also known to produce larger eggs (Parmenter 1985).

Incubation periods for *C. longicollis* eggs range between 93 and 168 days in the wild (Parmenter 1985; Cann 1998) although an incubation period of 185 days has been recorded. Incubation periods are temperature dependant with longer incubation periods experienced at lower temperatures (Parmenter 1985). *C. longicollis* life history attributes are summarised in Table 8. A study of nests along the Murray River, found that 96% of all turtle eggs were taken by predators, most of these (97%) by the European red fox (*Vulpes vulpes*) with only 3% eaten by native predators, including; the golden water rat (*Hydromys chrysogaster*), two species of goanna (*Varanus varius* and *V. gouldii*) and ravens (*Corvus coronoides*) (Thompson 1983). In the absence of predation, choice of nesting site is the single most important factor determining egg survival

(Parmenter 1985). For example, eggs in nests with underlying rock are susceptible to mortality from inundation.

Table 8: Life history attributes of *Chelodina longicollis*

Characteristic	Description
Thermal tolerance	Inactive below 12 ^{o}C (Arthur Georges pers. comm.).
Longevity/lifespan	Up to 70 years (Mussared 1997; Arthur Georges pers. comm.).
Age at sexual maturity	Approximately 7 years for males and 10–11 years for females (Burgin et al. 1999). Approximately 12 years for males and 15 years for females (Arthur Georges pers. com.).
Peak nesting season	In central New South Wales nesting occurs in April and May but probably varies within the range of the species (Cann 1998)
Nesting frequency	One to three clutches of eggs annually depending upon population location (Chessman, 1978; Parmenter 1985). Parmenter (1985) found individuals in the vicinity of Armidale (New South Wales) produced one clutch of eggs per year. Reproduction ceases if female body condition is poor such as when food is limiting (Arthur Georges pers. comm.).
Fecundity	8–12 eggs per clutch (Parmenter 1985; Cann 1998) with direct relationship between female size and egg size (Parmenter 1985)
Nesting migration	*C. longicollis* usually nest close to the water's edge although in some areas they travel to sites 500 m from the water and about 100 m above the water, probably to protect the eggs from flooding (Cann 1998).
Critical physical/chemical attributes required at breeding site	Areas selected for nesting are usually open and eggs are deposited in a firm substrate, in straight-sided or flask shaped nest to a depth of about 12 cm (Cann 1998).
Critical physical/chemical attributes for egg development	Egg failures are known to occur if nests become inundated (Parmenter 1985)
Parental care	None known
Egg characteristics	Hard-shelled
Time to hatching	Incubation periods range from 93 to 185 days in the wild (Parmenter 1985; Cann 1998) and are known to be temperature dependant, with longer periods resulting from cooler temperatures (Parmenter 1985).
Time from hatching to independence	Juveniles are independent upon hatching (Cann 1998).

Ecological value it supports

C. longicollis is an iconic turtle species native to the Warrego, Paroo, Bulloo and Nebine catchments (Cann 1998; DSITIA 2013a; DSITIA unpublished data). It is not currently listed under any state or commonwealth government legislation. Its persistence is linked to a number of ecological outcomes in the plan (9f (i, iii, vi, vii), 9j).

Assessment end point

Viability of *C. longicollis* populations within the Warrego, Paroo, Bulloo and Nebine catchments.

Measurement end point

Frequency of high stress events for *C. longicollis* populations. High stress events are associated with the persistence of off-stream water bodies, and the persistence of permanent waterholes (i.e. the duration of no-flow spells interacting with wetland and waterhole persistence characteristics).

Eco-hydraulic rules

In the absence of suitable inundated ephemeral off-stream water bodies *C. longicollis* aestivates on floodplains for up to seven months after which time, if off-stream conditions remain unfavourable, individuals must return to the refuge of permanent in-channel waterholes. *C. longicollis* individuals are known to lose body condition when restricted to these waterholes. Subsequently this period represents high stress and therefore high risk events for *C. longicollis* populations (Figure 3). The level of risk increases with an increase in the time between the availability of off-stream habitat.

- Low stress periods = feeding activity following the inundation of floodplain wetlands + 7 month aestivation period.
- High stress periods = time following low stress periods, prior to a subsequent floodplain wetland inundation when turtles are restricted to in-channel waterholes.

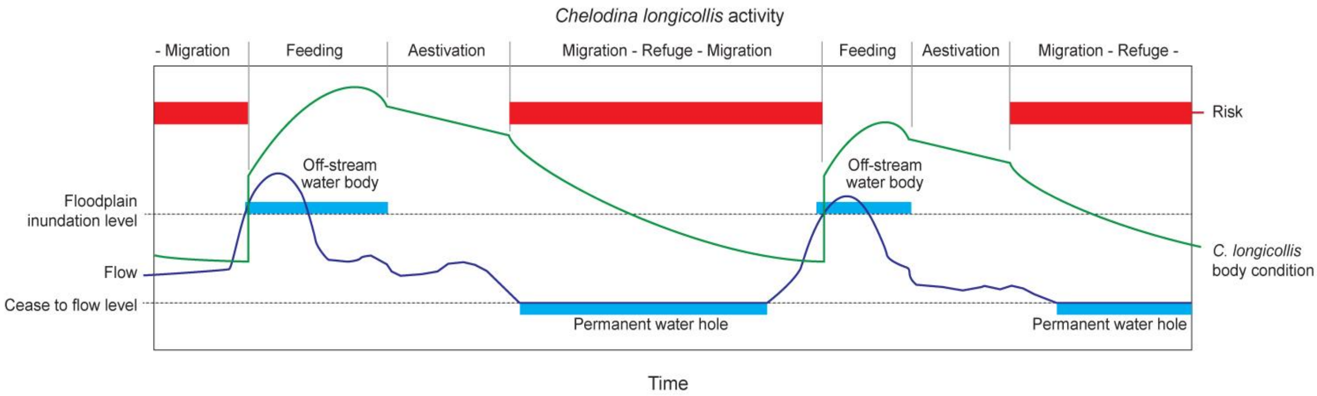

Figure 3: Conceptual relationships between flow magnitude, habitat availability, and risk to *C. longicollis* over time. The red horizontal bars indicate the periods of risk that will be modelled. Those periods include refuge in permanent water holes and well as migration to and from those water holes.

Threshold of concern

A threshold of concern was defined which represents the maximum length of high stress events *C. longicollis* populations can tolerate. A population of *C. longicollis* in the Jervis Bay area (NSW) took ten years to recover body condition sufficient for them to reproduce following a four year drought (A. Georges, pers. comm.). Based on this information a ToC of 4 years was used in the assessment (Table 9). Spells between floodplain inundation events at an environmental assessment node ≥ 4 years represents a threat to the long term persistence of the local population (Table 9). For each assessment node, the flow threshold representing the floodplain inundation event for the ToC was determined based on the inundation evaluations explained below in the methods section on Floodplain Wetlands.

Table 9: ToC and node failure threshold for *Chelodina longicollis*

Threshold of concern	Environmental assessment node failure
4 year flood inundation return frequency	4 years without a floodplain inundation event

Aspects of hydrology

C. longicollis has links to high flows that inundate floodplain wetlands and no-flow spells which determine wetland and waterhole persistence.

Spatial relevance

C. longicollis occurs in the Warrego, Paroo, Bulloo and Nebine catchments (Cann 1998; DSITIA 2013a; DSITIA unpublished data).

Absence of exotic fish species

Background

Many exotic fish have been introduced into Australian waterways since European settlement. Some of these introductions were accidental and many deliberate. The deliberate introductions include the well known salmonids including trout and salmon. Other species introduced primarily for angling include redfin, carp, roach and tench. Introduced fish represent a significant environmental stressor and threat to aquatic ecosystems (DSITIA 2013b). These invasive species impact on native species through predation, competition for food or habitat, uprooting aquatic vegetation and disturbing sediments, and by spreading diseases or parasites. Consequently the absence of exotic species represents a key ecological value. For the purposes of this assessment, the threat posed by exotic fish species in the plan area has been represented by the European carp (*Cyprinus carpio*) as it poses the greatest threat in the area (DSITIA 2013b).

C. carpio is considered to be the most abundant freshwater fish species in Australia and is the most abundant large freshwater fish in the Murray-Darling Basin (Gehrke et al. 1995; Lapidge 2003; Gilligan & Rayner 2007). It constitutes more than 90% of the fish biomass in many south-eastern Australian waterways. *C. carpio* was the most abundant and dominant species in the lower Balonne and Border Rivers of Queensland in the northern Murray-Darling Basin, representing 53% of the total fish abundance and 78% of the total fish biomass (Woods et al. 2012).

Appearance

C. carpio is a variably coloured fish, olive green or yellow green to golden overall with a silvery yellow belly. The mouth is surrounded by four barbels (Allen et al. 2002) with one pair in each corner. It has small eyes, thick lips, a forked tail and a single dorsal fin with strongly serrated spines. *C. carpio* scales are large and thick. *C. carpio* is Australia's largest alien freshwater fish (Koehn 2004). The species can grow up to 1200 mm in length (Allen et al. 2002) and there are overseas reports of fish weighing 60 kg. Fish of up to 10 kg have been caught in Australia, but weights of around 4–5 kg are more common. *C. carpio* tend to form small, loose shoals that swim actively in shallow water (Moffatt & Voller 2002).

Distribution

C. carpio are native to Eurasia (Haynes et al. 2009) and since introduction, have become established in all Australia states except the Northern Territory (Gillgan & Rayner 2007). The first carp were introduced into irrigation canals in southern-central NSW sometime during the 1940s or 1950s. Those fish were a strain of Asian koi carp and, although establishing a self-sustaining population, failed to disperse widely (Gilligan & Rayner 2007). It was not until the introduction of a second strain, imported from Europe and raised on a fish farm in Victoria, that the species became a 'pest' (Gilligan & Rayner 2007). Large floods in 1974–75 accelerated the spread of carp in the Murray-Darling Basin (Lapidge 2003) and have since become well established (Gilligan & Rayner 2007). *C. carpio* are present in the Warrego, Paroo and Nebine catchments of the WRP area, but are believed to be absent in the Bulloo River.

C. carpio is a highly vagile species that migrates up and down rivers throughout the year (Stuart & Jones 2002; Moffatt & Voller 2002) and has therefore a great ability to disperse (Gilligan & Rayner 2007). Although *C. carpio* is now distributed over a range of more than 1 million km^2 of south-east

Australia (Koehn 2004), the upper reaches of the Murray-Darling Basin have been identified as an area of most concern for the potential future expansion of *C. carpio* (Koehn 2004).

Migration in *C. carpio* is an important life history strategy. Juveniles are known to move from ‘source’ habitats, used for spawning and larval development, upstream to form populations in ‘sink’ habitats, where conditions are not necessarily suitable for spawning but can sustain juvenile and adult carp (Driver et al. 2005; Gilligan & Rayner 2007). Migrations of *C. carpio* in the Queensland’s Murray-Darling basin have been found to follow this model. For example, Bennett (2008) using otolith micro-chemistry, found that *C. carpio* in the Moonie River of south western Queensland represent a sink population that was sourced from a nursery site in the vicinity of the Weir River. Bennett’s results were supported by Woods et al. (2012) who found a higher proportion of young of the year carp in downstream sites compared to upstream sites in the Balonne and Border Rivers of south western Queensland.

There is little empirical information that examines the processes and mechanisms that have facilitated the rapid distribution of this species (Nicol et al. 2004). The population dynamics of wild *C. carpio* in Australia are poorly understood and the current paucity of basic information on age structure and growth rates makes it difficult to relate carp abundance to environmental factors (Roberts & Tilzey 1997; Brown et al. 2005). *C. carpio* are known to have very broad environmental tolerances and thrive in habitats that have been disturbed by humans (Lapidge 2003), particularly habitat altered by the regulation of rivers (Nicol et al. 2004). For example, the construction of dams, weirs, reservoirs and irrigation canals in the Murray-Darling Basin have provided still water habitats that are ideally suited to *C. carpio* (Haynes 2009).

Altered flow regimes also influence the distribution of *C. carpio.* Greater stability of the human-controlled environment favours only a small number of native fish species (Haynes 2009) and increases the dispersal opportunities for *C. carpio* by increasing longitudinal connectivity. Transport of their sticky eggs on waterbirds has been proposed as a mechanism for dispersal of *C. carpio* (Gilligan & Rayner 2007). If such opportunities exist they pose a risk for the non-anthropogenic spread of *C. carpio* beyond its current range and potentially across geographical barriers (e.g. into the Bulloo catchment).

Habitat

C. carpio occur in large water storages, irrigation canals, large rivers, off-stream wetlands, and billabongs (Gilligan & Rayner 2007). This species prefers slow flowing or still water (Koehn & Nicol 1998), with a silty substrate as well as access to shallow vegetated areas for spawning (Gilligan & Rayner 2007). *C. carpio* have been recorded from altitudes as high as 760 m in the Murrumbidgee catchment (Gilligan & Rayner 2007) but generally show a preference for altitudes less than 500 m (Koehn 2004) and are most abundant at altitudes less than 310 m (Gilligan & Rayner 2007).

C. carpio are ecological generalists (Haynes 2009) and exhibit most of the traits predicted for a successful invasive fish species (Koehn 2004) including broad physiological tolerances such as withstanding water temperatures of 2–40^{o}C, pH range of 5–10.5, high turbidity, moderate salinities, high toxicant loads and dissolved oxygen levels as low as 7% (Lapidge 2003; Koehn 2004, DEEDI 2011). This enables carp to survive periods of poor water quality and gives it a competitive advantage over many native fish species in Australia (Lapidge 2003). For example, some cyprinids use lactic acid fermentation during anoxia and some of the lactate (up to 70%) is oxidised (Hockachka 1980, cited in Magoulick & Kobza 2003), enhancing their ability to use stagnant pools as refugia during drought conditions (Magoulick & Kobza 2003). Habitat degradation may also preferentially favour *C. carpio* and as the suitability of habitat for native species has declined because of river regulation, it has improved for *C. carpio* (Nicol et al. 2004).

Reproduction

C. carpio reach sexual maturity early, with males becoming sexually mature at one year and females reaching sexual maturity at two years (Kohen 2004; Lapidge 2003). Mature male *C. carpio* are smaller than mature females. The minimum total length of sexually mature fish has been recorded at 250 mm for males and 390 mm for females (Pinto et al. 2005). *C. carpio* life history attributes are summarised in Table 10.

Spawning usually occurs in late spring or early summer (Lapidge 2003) and can continue to autumn with temperatures between 17–29^{o}C. *C. carpio* has been observed to spawn year-round in the Botany Wetlands in Sydney (Pinto et al. 2005). *C. carpio* is known to preferentially spawn in areas with abundant aquatic vegetation (Haynes 2009) or inundated terrestrial vegetation (Taylor et al. 2012) and during flood conditions are most often found on floodplains rather than the adjacent main river channels (Gilligan & Rayner 2007). *C. carpio* are serial spawners and may spawn several times in one season (Lapidge 2003). Females are highly fecund (Kohen 2004) and are capable of producing more than 1 million eggs per year. The adhesive eggs of *C. carpio* are attached to submerged vegetation (Haynes 2009) or scattered by spawning adults in aquatic vegetation (Moffatt & Voller 2002; Kohen 2004).

Hatching of *C. carpio* eggs is rapid, taking only two days at 25^{o}C (Koehn 2004) and six days at 18^{o}C (Lapidge 2003). Newly hatched larvae are not protected by adults (Moffatt & Voller 2002) but grow very rapidly (Koehn 2004) with growth rates varying between regions depending upon temperature, food availability and population density (Lapidge 2003). Mortality rates can be high and may exceed 98% in the first year (Lapidge 2003). Recruitment success rates have a direct relationship with access to suitable inundated habitat and are often much greater in years with large floods (Lapidge 2003). Flood conditions are especially favourable to *C. carpio* spawning and recruitment, as they provide abundant food resources for adults and abundant vegetation for the attachment of eggs and result in plankton blooms that provide food resources for growing larvae and juveniles (Haynes 2009; Woods et al. 2012).

Table 10: Life history attributes of *Cyprinus carpio*.

Characteristic	Description
Temperature range	4–35°C (Gilligan & Rayner 2007) 2–40°C (Koehn 2004)
pH range	5–10.5 (Koehn 2004)
Dissolved oxygen	7% saturation (Koehn 2004)
Longevity/lifespan	Typically 15 years (Moffatt & Voller 2002) up to 17 years although it is thought that they can live much longer (Lapidge 2003). More than 20 years (Stuart & Jones 2006).
Age at sexual maturity	2 years (Moffatt & Voller 2002). 1 year for males and 2 years for females (Koehn 2004).
Minimum total length at sexual maturity	250 mm (TL) for males and 390 mm (TL) for females (Pinto et al. 2005).
Spawning frequency	May spawn several times in a single season (Lapidge 2003).
Fecundity	Highly fecund (Koehn 2004) Females may produce more than 1 million eggs per year (Lapidge 2003).
Egg characteristics	Adhesive eggs (Moffatt & Voller 2002) Broadcast spawners (Koehn 2004)
Nesting site	Adhesive eggs are attached to, or scattered over, aquatic vegetation and submerged terrestrial vegetation (Moffatt & Voller 2002; Haynes 2009).
Time to hatching	2 days at 25°C (Koehn 2004) and 6 days at 18°C (Lapidge 2003).
Critical physical/chemical attributes required at breeding site	Spawning temperature between 15–28.2°C (Koehn 2004)
Parental care	None (Moffatt & Voller 2002).

Ecological value it supports

The absence (and/or low abundance) of exotic species represents a key ecological value. For the purposes of this assessment, the threat posed by exotic fish species in the plan area has been represented by the European carp (*Cyprinus carpio*). There are currently no known *C. carpio* populations in the Bulloo catchment. Carp abundance is linked to a number of ecological outcomes in the plan (9f (iii, vii)).

Assessment end point

Minimised abundance and distribution of *C. carpio* populations in Warrego, Paroo, and Nebine catchments of the WRP area.

Measurement end point

The frequency of strong recruitment opportunities for *C. carpio.*

Eco-hydraulic rules

Access to inundated floodplain wetlands has been shown to positively influence *C. carpio* spawning rates and the habitat provided by inundated floodplains is also known to increase recruitment rates. There were two components to the eco-hydraulic rule for *C. carpio* strong recruitment opportunities:

1. Spawning and recruitment was associated with overbank flows (i.e. lowest flow threshold that inundated floodplain wetlands–see Floodplain wetlands methods below).
2. Opportunities for dispersal of recruits were associated with any flows ≥ 2 ML/day within 12 months of an overbank flow (Craig Johansen, pers. comm.).

Threshold of concern

A ToC could not be derived as the relationship between the frequency of strong recruitment opportunities for *C. carpio* and their abundance and distribution across the plan area has not been adequately characterised. The potential for water resource development to affect recruitment opportunities for *C. carpio* was assessed based on a comparison of the proportional change between the pre-development and full entitlement scenarios.

Aspects of hydrology

C. carpio has direct links to two aspects of hydrology; high flows that inundate floodplain wetlands and medium flows for longitudinal connectivity.

Spatial relevance

C. carpio occurs in the Warrego, Paroo, and Nebine catchments of the plan area.

Floodplain vegetation

Background

Floodplains are complex riverine landscapes that are highly spatially and temporally variable. The combination of topography, soil type and land use leads to variability in both the frequency with which different parts of the floodplain receive water and the duration that surface water persists. Because of this, floodplains are a mosaic of ecosystems that utilise the hydrologic regime at relatively small scales. Variability in groundwater depth across floodplains and thus its availability to floodplain vegetation further magnifies this complexity.

Flood harvesting may reduce the magnitude of individual floods and thus increase the duration of spells between inundation by flooding for some habitats and, over longer periods, influence the distribution of species (Bren 1992; Kingsford & Thomas 1995; Bowen et al. 2003; Thoms 2003).

Several of the ecological assets identified for the plan area are critically linked to floodplain inundation (DSITIA 2013a). Iconic floodplain plant species, (including river red gum, black box, coolabah, tangled lignum, river cooba and yapunyah gum), require riverine flooding for both successful reproduction and recruitment, and to maintain the vigour and condition of adult plants (Roberts & Marston 2000; Rogers 2011).

Ecological significance

Floodplain vegetation communities are diverse assemblages of terrestrial plant species with specific requirements related to periodic flooding or water ponding (Roberts & Marston 2011). Flow regimes are relevant to the maintenance and persistence of floodplain vegetation in terms of the length of spells without inundation, the duration of inundation events and in some cases the timing of these events. Flow magnitude interacting with landscape morphology determines the size of events necessary to meet species-specific inundation requirements in a particular floodplain setting (Overton et al. 2009). Many species have flexible growth strategies which allow them to grow in arid and semi-arid areas. For example river red gum (*E. camaldulensis*) has a dual root system, with lateral roots that are close to the surface and extend sideways, and a taproot or ‘sinker’ that penetrates deep into the soil (Horner et al. 2009, cited in Roberts & Marston 2011). Collectively floodplain vegetation contributes to floodplain productivity, and provides critical habitat for many species including vulnerable Bulloo Grey Grasswren *(Amytornis barbatus barbatus*) (DSEWPC 2012a).

Distribution

Floodplain vegetation is found throughout all four catchments of the plan area.

Ecological value it supports

Floodplain vegetation is valued in its own right as a conspicuous element of rivers in the plan area. It also directly contributes to floodplain productivity and provides critical habitat to many species. Its persistence is linked to a number of ecological outcomes in the plan (9f (iii, vi, vii), 9j)

Assessment endpoint

Viability of floodplain vegetation communities in the plan area.

Measurement endpoint

Pattern of inundation events across the floodplain.

Eco-hydraulic rules

Flow thresholds representing wetting frequencies for floodplain vegetation communities (represented by REs) were derived from evaluation of time-series of satellite imagery using the following four step process:

1. Definition of floodplain assessment reaches

The relatively sparse distribution of gauging stations coupled with the extensive areas of floodplain in the area and the complex hydrology of floodplains, mean that it is unlikely that available gauge data is representative of entire floodplains. The plan area was mapped by catchment including the location of environmental assessment nodes and the areas identified as floodplains by Queensland Reconstruction Authority Interim Floodplain Assessment Layer (IFAO). Queensland government hydrologists and hydrographers used their local expert knowledge and an assessment of the locations of tributaries and constrictions in the stream network to identify the relevant areas of floodplain associated with each environmental assessment node. These are termed *floodplain assessment reaches*. The upstream, downstream and lateral limits of each floodplain assessment reach were digitised and adopted for the purposes of the environmental assessment (Figures 4–7). All floodplain asset assessments were confined to these reaches, as they are the parts of the floodplain that are represented by hydrology at the environmental assessment nodes.

2. Identification of floodplain vegetation patches

Queensland Regional Ecosystem (RE) types (DERM 2011a) that included floodplain plant asset species as dominant community components were identified based on RE descriptions. Patches of these RE types were then identified by overlaying RE mapping data (DERM 2011a) with the floodplain assessment reaches. These RE patches were used to represent the spatial distribution of floodplain vegetation asset species for the environmental assessment.

3. Selection and analysis of satellite image scenes

Landsat satellite imagery was used to distinguish the location and extent of surface water in RE patches. For each assessment reach, Landsat satellite image scenes covering the area were identified. Images were available at approximately fortnightly intervals for the period 2000–2012. Landsat images were analysed to determine if RE patches within the assessment reaches were wet or dry on each scene date. Wet Landsat pixels were identified using the spectral analysis method described in EPA (2005a, b). RE patches were classified as wet if any pixel within the patch area was identified as wet, which is a modification of the published method (designed for wetlands rather than RE patches), because the minimum adjacent pixel rule has been relaxed (EPA 2005a, b). This modification was made based on the assumption that inundation of RE patches may be much more transient than inundation of wetlands. These analyses generated a matrix in which each RE patch was classified as wet, dry or cloud obscured on each scene date.

4. Interpretation of wetting patterns

Patterns of wetting and drying for each RE patch within a floodplain assessment reach were interpreted in the context of gauged stream flow, to determine which patches experienced hydrologic connectivity to the river network, and the magnitude of floods required for their inundation. For each environmental assessment node, daily flows recorded at stream gauges for the period concurrent with satellite image availability were plotted as hydrographs. The timing and magnitude of flow events was then compared to the time series of satellite imaging results, considering events in order of size, starting with the largest, until the threshold for inundation of each feature could be determined. This logical and progressive approach was necessary as many RE patches were inundated by sources other than river flooding (presumably by local rainfall) as they filled when the river was not flowing. The influence of local rainfall and river flooding was often difficult to separate, so a precautionary approach was applied whereby the magnitude of the smallest size flood event that may have inundated each RE patch was adopted, and for this to be the case, every event at or above the particular threshold had to result in inundation. In many cases long periods of cloud cover further obfuscated the process and in line with the precautionary approach used, these occasions were not used to exclude smaller magnitude thresholds and adopt larger ones. Information on floodplain inundation height from available cross sections and ‘points of inflection’ in gauge rating curves were used (*sensu* Woods et al. 2012) as additional information sources to confirm that the flood magnitudes adopted were reasonable estimates of floodplain inundation.

These magnitudes were used as the eco-hydraulic rules to define opportunities for vegetation inundation (Table 11) for each RE patch. Because, in many cases, asset species were represented by both multiple RE types and multiple patches of each RE type, there were often multiple inundation thresholds identified for each within floodplain assessment reaches.

Table 11: Floodplain vegetation flow inundation thresholds (as ML/day)*

Species	Environmental assessment node						
	423004	423005	423202C	423206A	423203A	423204A	422502A
Acacia stenophylla	5000 5431 7970 8470 19140	6691	20000 47000 50498 68209 126466	3044 20487 208015	3431 64244 244770		7626
Eucalyptus camaldulensis	5000 5431 8470 19140	6691	20000 47000 68209 126466	3044	3431 7935 64244 244770	6827 11400 30557	
Eucalyptus ochrophloia	5000 5431 8470 19140	6691	20000 47000 68209 126466	3044 20487 69317 208015	3431 7935 244770		
Eucalyptus coolabah	5000 5431 7970 8470 19140	1322 6691 16115 46215	20000 47000 50498 126466	3044 20487 39152 69317 208015	3431 7935 20572 64244 244770	6827 11400 30557	7626
Eucalyptus largiflorens	19140		126466				
Muehlenbeckia florulenta	5431 19140		47000 126466		64244		

*REs dominated by each of the asset vegetation species in the plan area are shown in Appendix 2.

Threshold of concern

A ToC could not be derived for a water regime which maintains the viability of floodplain vegetation communities in the plan area (including recruitment processes and maintenance of mature plants). Although watering requirements have been identified for a number of floodplain species, most ignore the obvious surface water–groundwater interactions which occur on the floodplain and how this influences vegetation community distribution and resilience to desiccation. This is reinforced here by the finding that different patches of the same species within the plan area, and even within individual floodplain assessment reaches, were inundated with very different frequencies even under the pre-development scenario (note the very different inundation thresholds per species in Table 11). Consequently the potential for water resource development to affect the floodplain vegetation inundation regime was assessed based on a comparison of the proportional change between the pre-development and full entitlement scenarios.

Aspects of hydrology

Floodplain wetlands have a direct link to the frequency of high flows.

Spatial relevance

Floodplain vegetation is relevant to the entire plan area.

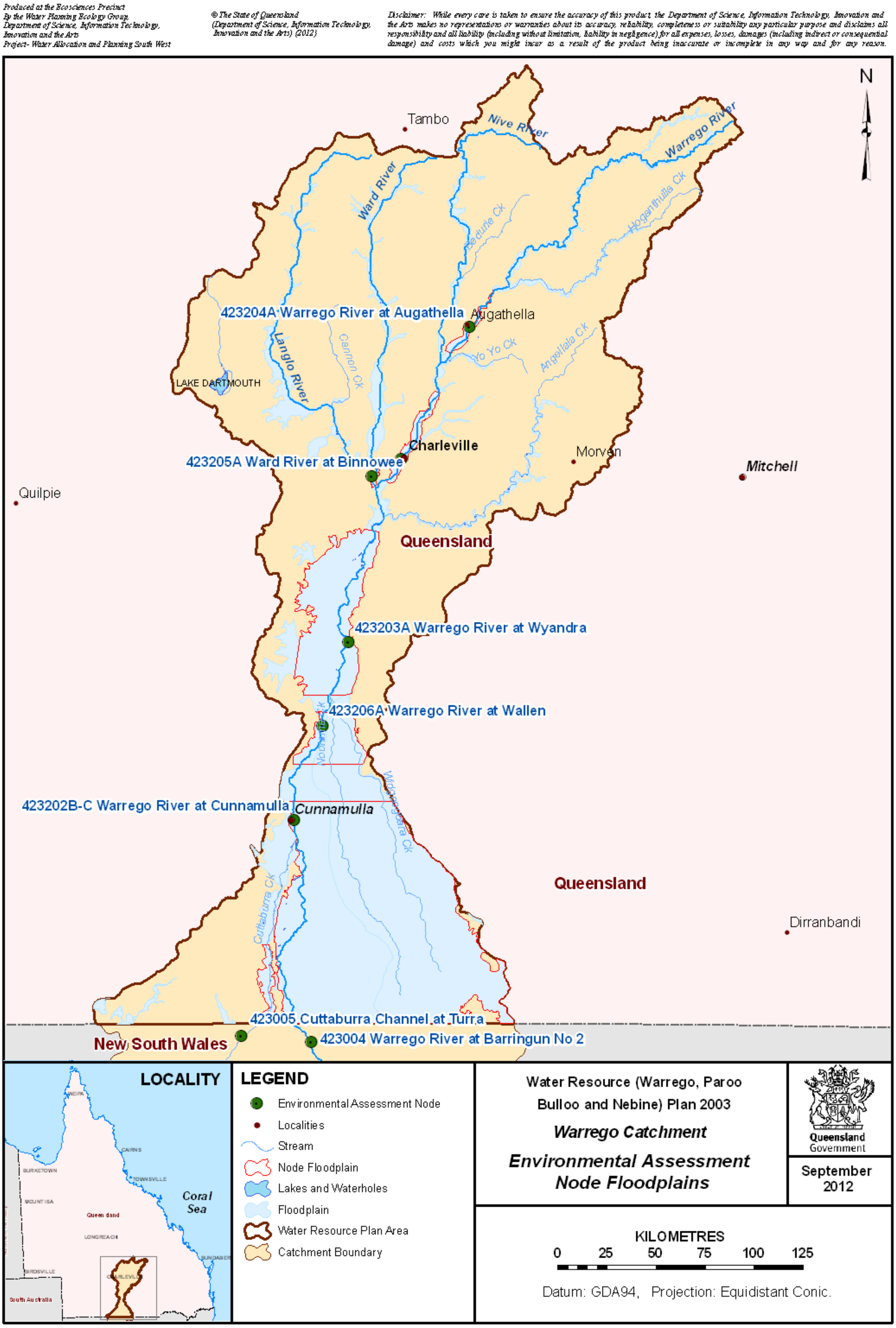

Figure 4: Floodplain areas associated with each environmental assessment node in the Warrego catchment

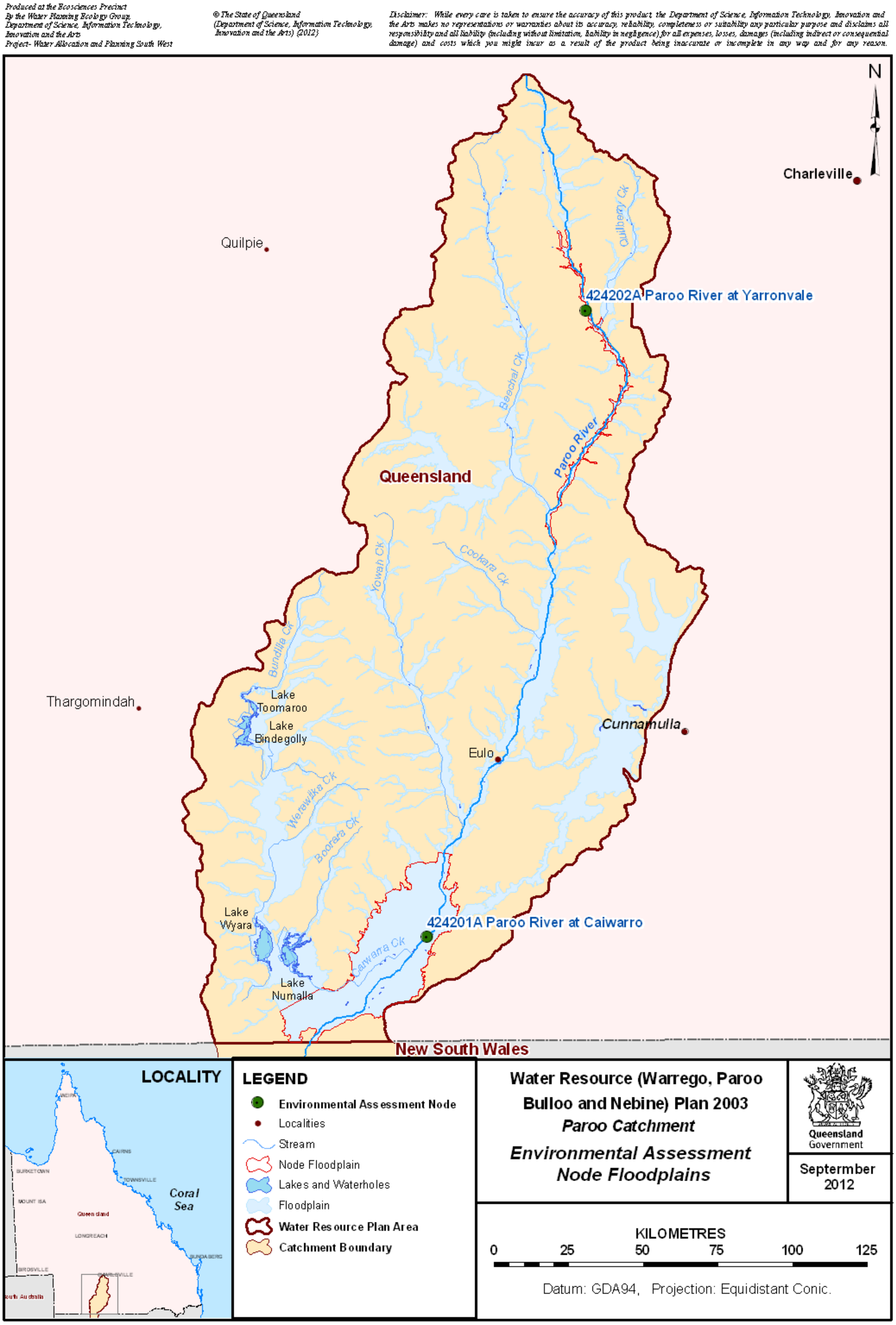

Figure 5: Floodplain areas associated with each environmental assessment node in the Paroo catchment

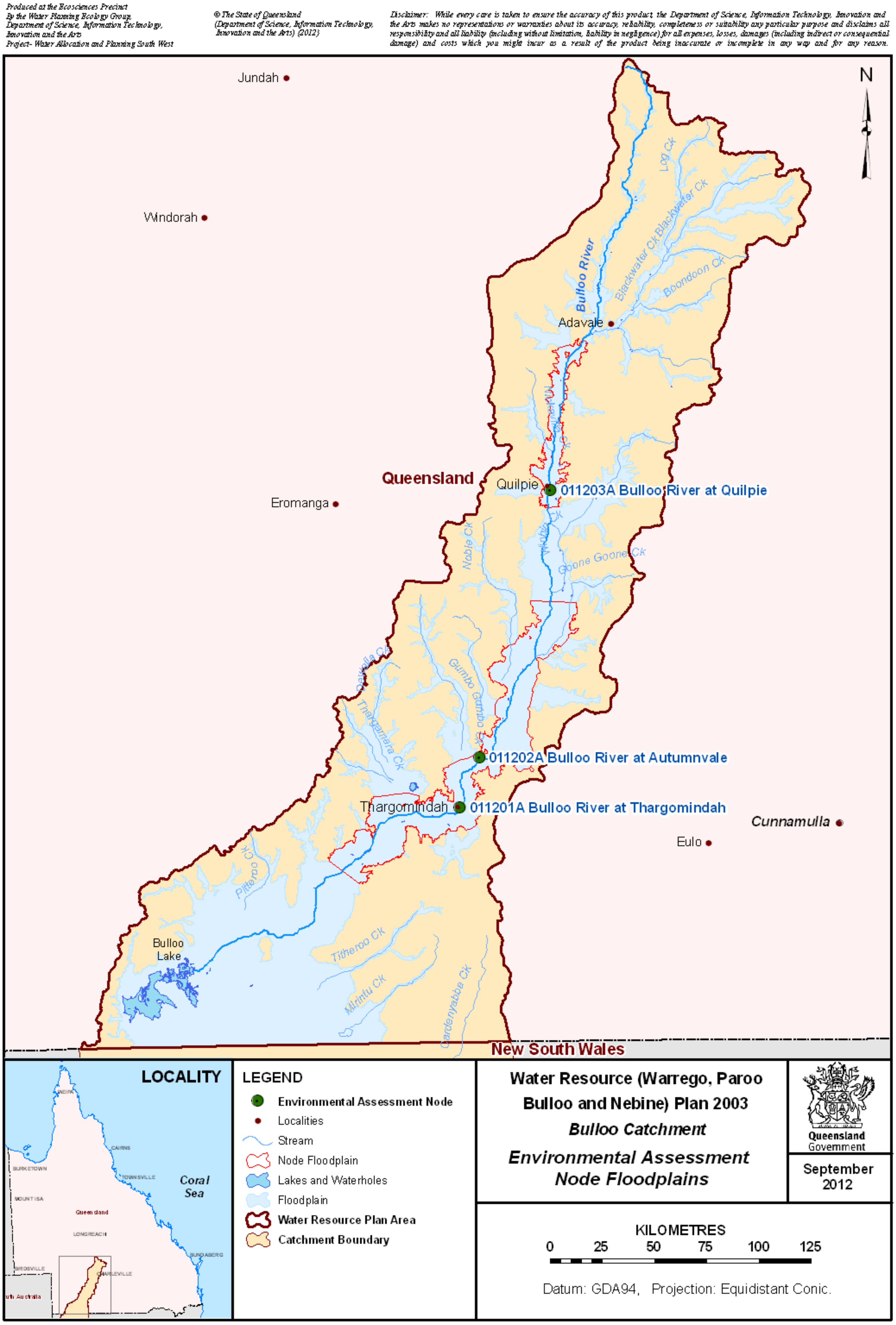

Figure 6: Floodplain areas associated with each environmental assessment node in the Bulloo catchment

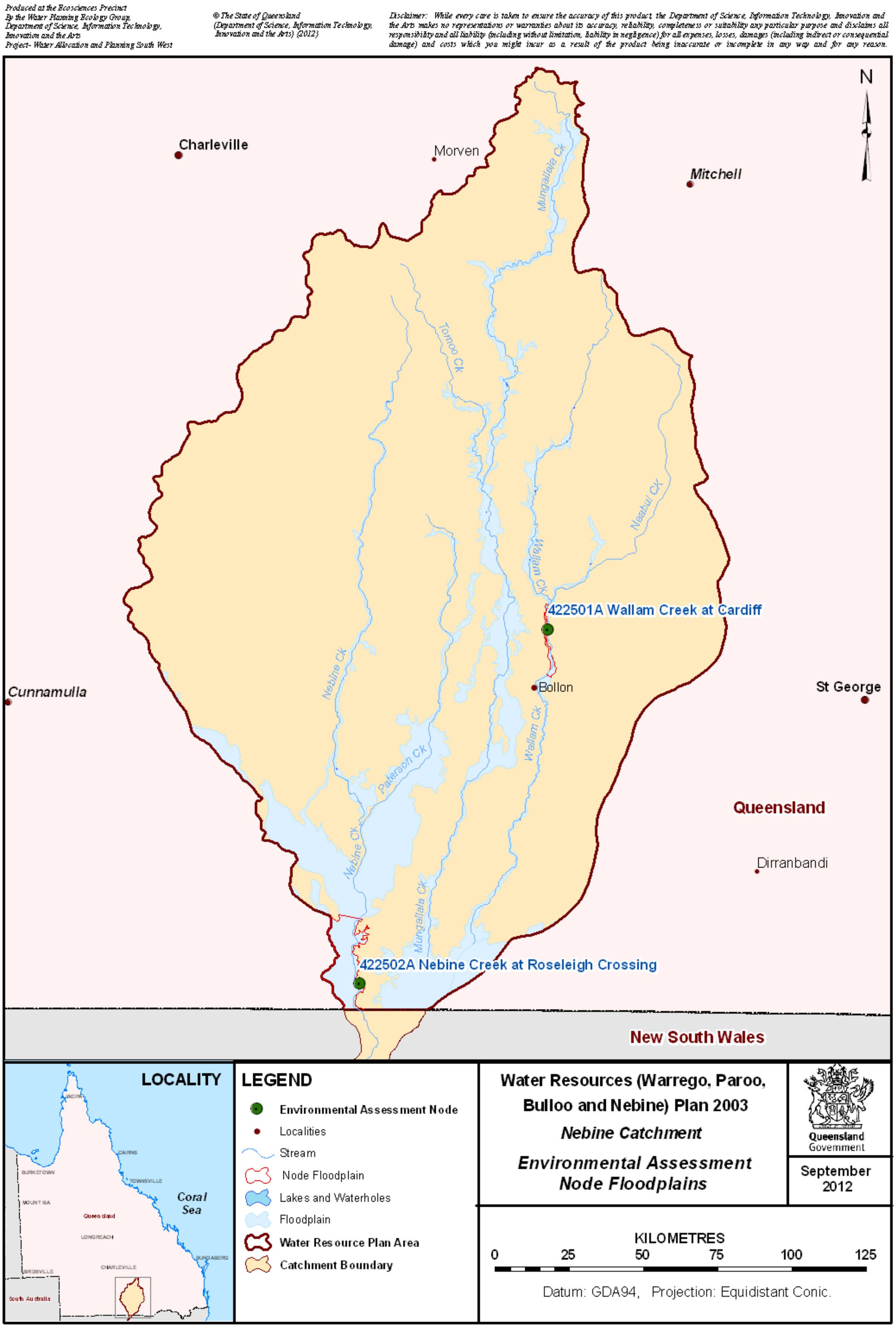

Figure 7: Floodplain areas associated with each environmental assessment node in the Nebine catchment

Floodplain wetlands

Background

The term wetland describes a range of landscape features defined as areas where water covers the ground either permanently or seasonally. They support diverse ecological communities that are adapted to take advantage of the resources that are periodically available (Westlake & Pratt 2012). For the purpose of this assessment, floodplain wetlands are defined as those within the floodplain assessment reaches that receive water from riverine flooding, excluding the river channel itself.

Floods are required to inundate floodplain wetlands, which act as critical habitat for many species of plants and animals (DERM 2012). Inundated wetlands also provide the optimum habitat for another plan asset, the eastern snake-necked turtle (Chessman 2011) and, as a negative outcome of flooding, provide nurseries for recruitment of the invasive European Carp (Gilligan & Rayner 2007).

Ecological significance

In dryland river landscapes, floodplain wetlands play an important role in the exchange of carbon and nutrients, provide ecosystem services such as water quality buffering, act as refuges during dry spells and provide habitat to a diverse community of plants and animals (Thoms 2003; DSEWPC 2012b).

The length of time between inundation events and the persistence of water in a wetland are governed by a number of factors including position in the landscape, water source, climate and substrate (Jaensch & Young 2010). The sequence of drying and rewetting in temporary floodplain wetlands makes them highly productive. As the wetlands dry, decaying aquatic organisms create a rich substrate for the growth of dryland grasses and herbs and upon rewetting, breakdown of these provide substantial resources for aquatic invertebrates, algae and plants (Scott 1997). The rapid development of these food sources makes inundated wetlands excellent breeding habitats for consumers such as waterbirds, fish and turtles.

In the plan area, floodplain wetlands support several ecological values including waterbird breeding opportunities and provision of preferred habitat for the eastern snake-necked turtle, and in some cases, are recognised by state, national and international conservation agreements such as RAMSAR or protected in National Parks (DSITIA 2013a).

Distribution

Floodplain wetlands are present in all four plan catchments, particularly in the lower reaches that have wide, well developed floodplains and complex distributary networks. Queensland Wetland Mapping (DERM 2011b) was used to determine wetland location and type. Those that are filled by overbank flooding, and therefore potentially influenced by water resource development were identified by satellite image analysis as described below.

Ecological value it supports

Within the WRP area floodplain wetlands support a number of ecological outcomes of the current plan–9f (ii, iii, v, vii), 9g, 9j.

Assessment endpoint

Maintenance of the wetting regime to support floodplain wetlands

Measurement endpoint

Frequency of spells between wetland inundation thresholds as defined in this method

Eco-hydraulic rules

Flow thresholds representing wetting frequencies for floodplain wetlands were derived from evaluation of time-series of satellite imagery using the following four step process:

1. Definition of floodplain assessment reaches

Floodplain assessment reaches for wetlands are the same as those for the floodplain vegetation assessment described above (Figures 4–7).

2. Identification of floodplain wetlands

Floodplain wetlands within the assessment reaches were identified using the Queensland Wetland Mapping Database (DERM 2011b) filtered to exclude riverine wetlands (i.e. river channels, R code in the 'wetland system' attribute) and artificial wetlands (H3 code in the 'local hydrological modification' attribute).

3. Selection and analysis of satellite image scenes

Landsat satellite imagery was used to distinguish the location and extent of surface water in each of the floodplain wetlands within assessment reaches. For each assessment reach, Landsat satellite image scenes covering the area were identified. Images were available at approximately fortnightly intervals for the period 2000–2012. Landsat images were analysed to determine if floodplain wetlands were wet or dry on each scene date. Wet Landsat pixels were identified using the spectral analysis method described in EPA (2005a, b). A wetland was classified as wet when a minimum of eight adjacent pixels within its extent (as defined by the Wetland Mapping data) had a spectral wet signature, and the area was greater than 0.25 ha (EPA 2005a, b). Use of this image analysis method ensured results were comparable with the Queensland Wetland Mapping database (EPA 2005a, b). These analyses generated a matrix in which each floodplain wetland was classified as wet, dry or cloud obscured on each scene date.

4. Interpretation of wetting patterns

Patterns of wetting and drying for each floodplain wetland were interpreted in the context of gauged stream flow, to determine which wetlands experienced hydrologic connectivity to the river network, and the magnitude of floods required for their inundation. For each environmental assessment node, daily flows recorded at stream gauges for the period concurrent with satellite image availability were plotted as hydrographs. The timing and magnitude of flow events was then compared to the time series of satellite imaging results, considering events in order of size, starting with the largest, until the threshold for inundation of each feature could be determined. This logical and progressive approach was necessary as many floodplain wetlands were inundated by sources other than river flooding (presumably by local rainfall) as they filled when the river was not flowing. The influence of local rainfall and river flooding was often difficult to separate, so a precautionary approach was applied whereby the magnitude of the smallest size flood event that may have inundated each RE patch was adopted, and for this to be the case, every event at or above the particular threshold had to result in inundation. In many cases long periods of cloud cover further obfuscated the process and in line with the precautionary approach used, these occasions were not used to exclude smaller magnitude thresholds and adopt larger ones. Information on floodplain inundation height from available cross sections and 'points of inflection' in gauge rating curves were used (*sensu* Woods et al. 2012) as additional information sources to confirm that the flood magnitudes adopted were reasonable estimates of floodplain inundation.

Wetland inundation occurs when flow volume exceeds the thresholds listed in Table 12. In some cases multiple thresholds were identified, indicating that different size floods are needed to fill different wetlands within a floodplain assessment reach.

The duration that each floodplain wetland held water following filling was also approximated based on satellite image analysis and this contributed to assessments of other assets such as turtles and carp.

Table 12: Flow thresholds (ML/day) for inundation of floodplain wetlands

Assessment node	Flow threshold (ML/day)
Warrego River at Barringun 423004	5431
Warrego River at Turra 423005	46125
Warrego River at Cunnamulla Weir 423202C	47000 126466 136138
Warrego River at Wallen 423206A	n/a
Warrego River at Wyandra 423203A	64244 244770
Warrego River at Augathella 423204A	30557
Nebine Creek at Roseleigh Crossing 422502A	2540

Threshold of concern

There is insufficient information to set ToC relating to floodplain wetlands however an assessment of hazard was made by comparing hydrological deviation from the benchmark of the pre-development hydrological scenario.

Aspects of hydrology

Floodplain wetlands have a direct link to the frequency of high flows.

Spatial relevance

Floodplain wetlands occur throughout the plan area.

Waterholes as refugia

Background

For the purposes of this assessment, permanent refugial waterholes are defined as main-channel river-pools that retain an ecologically significant depth of water between no-flow spells. Main channel waterholes are those that can be maintained by in-channel surface water flows. In some cases waterholes may also be supplemented by groundwater. An evaluation of the groundwater contribution to waterhole persistence is outside the scope of this assessment.

Ecological significance

Permanent refugial waterholes have been selected as an asset to represent the value of maintaining habitat for biota of the Warrego, Paroo, Bulloo, and Nebine catchments. A key feature of ephemeral systems such as these is that much of the riverine fauna is dependent upon the persistence of a network of refugial waterholes during frequent and often prolonged no-flow periods (Balcome et al. 2006, DERM 2010b). These waterholes enable resistance and resilience of aquatic populations, processes essential for vigour and long term viability, by providing habitat to 'ride out' dry spells (Davis & Thoms 2002; Humphries & Baldwin 2003; Magoulick & Kobza 2003) and allow dispersal during subsequent flow events (Puckridge et al.1998; Balcome et al. 2007; DERM 2010b).

Refugial waterholes provide one of the few sources of habitat for aquatic biota such as fish, turtles and aquatic invertebrates during dry spells. The persistence of a refugial waterhole (i.e. the length of time it contains water in the absence of flow) defines the maximum survival time of obligate aquatic biota that reside in it (Balcombe et al. 2007). The persistence of waterholes during no-flow spells is, in part, dependent upon the depth of those waterholes at the onset of no-flow spells (DERM 2010b). Depth loss in waterholes can be due to evaporation and seepage as well as extraction. All else being equal, the deepest waterholes are the most persistent and therefore the most ecologically important because of their ability to provide drought refugia for biota (DERM 2010b).

The longest no-flow spell experienced in a river determines which waterholes are permanent drought refuges in terms of supporting long-term population viability at this, the harshest of times. Water resource development which lengthens the duration of the longest no-flow spell in a river poses a hazard to the long term viability of multiple species that utilise refugia, as it may threaten the persistence of these naturally permanent waterholes. By understanding the rates of water loss from waterholes in a region and their depth, it is possible to model the maximum persistence time of each and this can then be compared with the duration of the longest spells expected to quantify risk (see DERM 2010b). However, this information is not available for the plan area.

Habitat conditions within a waterhole may also decline as water level recedes during a no-flow spell, meaning that it could become poor habitat for some species before it dries completely. As water level drops, changes can occur in water quality (e.g. reduction of dissolved oxygen, concentration of ions), water temperature, availability of food and complex habitat features, and the effect of disturbance, e.g. trampling by livestock, becomes more pronounced (Seehausen & Bouton 1997; Bouvy et al. 2003; Lake 2003; Magoulick & Kobza 2003; Bond et al. 2008; Beesley & Prince 2010). Bunn & Arthington (2002) suggest that mortality of fish trapped in dry season refuges may be very high due to deteriorating physico-chemical conditions, reductions in food availability and lack of refuge from predators. This means that the length of spells without flow is a key factor determining the quality of waterhole refugia to sustain aquatic biota. However, as long as some individuals survive these hardships, waterholes allow population recovery once no-flow spells end (DERM 2010b).

Along with the number and condition of surviving individuals, the capacity to recolonise after drought and maintain a wide biogeographic range is determined by the spatial arrangement of waterhole habitats and the extent of hydrologic connectivity, along with the dispersal traits of organisms (Bunn & Arthington 2002; Magoulick & Kobza 2003; Hughes 2007). Dryland fish species, such as Yellowbelly, use periods of hydrologic connectivity to move and explore the river channel to select optimum local habitats (Crook 2004a, b; Crook et al. 2010). Over multiple seasons, by using refugia as stepping stones in this way, individuals can move widely throughout catchments, maintaining their range despite local extinctions (Harrison 1991). By enabling fish to occupy the best feeding and breeding locations, such behaviour improves the long-term survival and fecundity of individuals which in turn maintains population viability (Crook 2004b). Opportunities for intermittent connectivity between sub-populations also allow gene flow, improving the vigour of organisms over generations (Dunham & Minckley 1998; Whitlock et al. 2000). However because flow events tend to be short, long distances between waterhole refugia may exceed the movement capacity of biota and act as a barrier to dispersal.

In northern Murray-Darling basin rivers, individual fish move between waterholes over distances of approximately 20 km when the river is flowing (DERM 2010b; see also Migratory fish methods above); therefore this represents an appropriate spatial scale to assess waterhole refuge availability to support fish population viability.

Distribution

Permanent waterholes occur across the plan area catchments and were identified using mapping data from Silcock (2009).

Ecological value it supports

Waterholes represent the value of maintaining refugial habitat for biota of the Warrego, Paroo, Bulloo, and Nebine catchments and support a number of plan ecological outcomes (9f (i, ii, iii, iv, vii), 9g).

Assessment end point

The spatial distribution and pattern of connectivity of permanent waterholes, including those with and without pumping licences.

Measurement end point

There are two measurement end points for the spatial distribution and pattern of connectivity of permanent waterholes:

1. Frequency and maximum duration of waterhole isolation spells (no-flow spells).
2. Change in the distance between permanent waterholes due to licensed waterhole pumping activities.

Eco-hydraulic rules

1. Isolation spells correspond to those periods when flow ceases and waterholes are disconnected from each other. No-flow spells were identified as days when the simulated flow was ≤ 2 ML/day (Craig Johansen pers. comm.). The maximum spell duration was calculated and the cumulative frequency of spell durations over the simulation period plotted under each flow scenario.
2. Distance between permanent waterholes with licensed pumping activities and the next adjacent upstream or downstream permanent waterhole < 20 km distance. The eco-hydraulic analysis assumed that under the full entitlement scenario waterholes with licensed pumping are lost from the network of waterholes during no-flow spells, in accordance with conditions associated with individual licenses. Full details of this analysis are presented in Appendix 3.

Threshold of concern

A ToC could not be derived for either measurement endpoint as waterhole persistence in the plan area has not been adequately characterised. The potential for water resource development to affect the pattern of waterhole persistence and distribution was therefore assessed based on a comparison of the proportional change between the pre-development and full entitlement scenarios.

Spatial relevance

Permanent waterholes are present in all four catchments of the Water Resource Plan area. Changes to the hydrology of no-flow spells were assessed at individual nodes and assessments of changes to waterhole distributions from pumping were made at the catchment scale as this is the scale at which networks of waterholes need to function to provide refuges for biota.

Fluvial geomorphology and river forming processes

Background

The primary drivers of channel morphology are hydrology, the underlying geology of the river channel and sediment availability (Clifford & Richards 1992). The underlying geology determines the extent to which flows can alter channel characteristics such as, stream bed, bed slope and meander, whereas sediment availability and entrainment processes are two factors that determine the creation and maintenance of pools and bars. As the water level falls during no-flow periods pools become a series of isolated waterholes separated by bars and those waterholes become important refugial habitats for biota.

The persistence of waterholes during no-flow spells is, in part, dependent upon the depth of those waterholes at the onset of no-flow spells (DERM 2010b). Depth loss in waterholes can be due to evaporation and seepage as well as extraction. All else being equal, the deepest waterholes are the most persistent and therefore the most ecologically important because of their ability to provide drought refugia for biota (DERM 2010b).

The aspects of hydrology considered in this assessment are those with the greatest potential to influence pool depth and, by implication, waterhole persistence. Stream channels are generally comprised of an alternating series of shallow bars (sometimes known as riffles) and deeper pools (Newbury & Gaboury 1994). The creation and maintenance of channel bars and deeper pools is known to be dependent on flow driven sediment entrainment and deposition processes (Haschenburger & Wilcock 2003; Wilkinson et al. 2003). Sediment transport occurs when shear stress, the force acting upon the substrate resulting from flow, is sufficient to entrain substrate sediments. The location of maximum shear stress moves up and down stream with changes in the magnitude of flow (Sear 1995; Wilkinson et al. 2003).

High flows have the greatest shear stress and the point of maximum shear stress occurs within the pool; low flows have reduced shear stress and the point of maximum shear stress moves to become located over shallow bars (Wilkinson et al. 2003). Scouring of pools occurs during high flows and the entrained pool substrates are deposited at the downstream end of pools and on channel bars. During low flows entrainment of bar substrates occurs and the entrained sediments are deposited in the upstream end of pools.

In the Moonie River (in the northern Murray Darling basin adjacent to the plan area), investigations of the age and composition of deposited sediments in waterholes has identified increased sedimentation since the 1950's, possibly associated with catchment vegetation clearing, as a threat to pool depth and thus persistence during dry spells (DERM 2010b). A survey of property owners confirms that sediment deposition is an important threat to waterhole persistence in the plan area too (Silcock 2009; Appendix 4). Individual flow events have been shown to scour up to 25% of this deposited sediment and thus be important to maintaining the function of waterholes as drought refuges for aquatic biota (DERM 2010b; DERM unpublished data). It follows that reductions to the magnitude, frequency and durations of high flow events, (sometimes termed 'maintenance flows'), have the potential to reduce pool depth over time leading to a reduction in waterhole persistence.

Bankfull discharge was chosen as the threshold at which the maximum shear stress and peak scouring within pools was reached. Bankfull discharge is typically the discharge at which the product of average cross-sectional flow velocity and water surface slope is at a maximum (Newbury & Gaboury 1994).

Ecological significance

The ecological significance of refugial habitat provided by persistent waterholes has been discussed in the previous section of this report. By implication fluvial geomorphology and river forming processes that maintain the depth of pools over time have an indirect influence on the persistence of waterholes as well as their suitability as refugia for biota.

Distribution

River forming processes, including sediment transport, occur throughout the plan.

Ecological value it supports

Within the plan area active river forming processes and the habitat they create support plan ecological outcomes 9f (iv).

Assessment end point

Maintenance of river forming processes within the Warrego, Paroo, Bulloo, and Nebine catchments of the WRP area (including the frequency, magnitude and duration of high flow events).

Measurement end point

Frequency of bankfull flow events over the simulation period. For the purposes of the assessment, bankfull is defined as flows at the top of the river bank and was derived for each environmental assessment node based on expert opinion (Craig Johansen pers comm.) The flow threshold representing bankfull was either the 1 or 2 year Annual Return Interval (ARI) flow event, depending on the location of the node in the stream network (Table 13).

Table 13: Flow thresholds used to represent bankfull flow events

Environmental assessment node	Annual return interval (ARI)	Flow event size (ML/day)
Warrego River at Augathella (423204A)	2	8202
Warrego River at Charleville (423201A)	2	23161
Warrego River at Wyandra (423203A)	2	60517
Warrego River at Wallen (423206A)	1	14820
Warrego River at Cunnamulla Weir (423202C)	1	14362
Warrego River at Barringun (423004)	1	5394
Warrego River at Turra (423005)	1	5225

Eco-hydraulic rules

Active river forming processes are represented by the frequency of bankfull events. Spells between these flows were calculated for the pre-development and full entitlement scenarios to calculate the number and the frequency distribution of spells.

Threshold of concern

A ToC could not be derived for the measurement endpoint as the relationships between the frequency of bankfull flows and their role in driving river forming processes has not been adequately characterised. The potential for water resource development to affect the frequency of bankfull flows was assessed based on a comparison of the proportional change between the pre-development and full entitlement scenarios.

Spatial relevance

River forming processes are important throughout the plan area.

Genetic diversity of aquatic biota in the Bulloo catchment

Background

Ecological outcome 9f (vi) of the plan states that: water is to be allocated and managed in a way that seeks to achieve a balance— to achieve ecological outcomes consistent with maintaining a healthy riverine environment, floodplains and wetlands, including, for example, maintaining the unique genetic diversity of aquatic plants and animals within the Bulloo basin.

Ecological significance

Biodiversity has long been recognised as an important ecological value. At a fine scale, the diversity within species—genetic diversity—is critical for maintaining ecosystem function, population resilience and evolutionary processes (Latta IV et al. 2011, Koh et al. 2012). Genetic diversity can be threatened by the introduction of individuals of the same species from populations that have been isolated for long evolutionary times and consequently have large genetic differences (Page et al. 2010). Translocation of new species may also pose a hazard to biodiversity in the catchment by altering species interactions such as predation pressure and food webs (Kolar & Lodge 2000; Dunham et al. 2002; Njiru et al. 2005). Such an event in the Bulloo could lead to the loss of local endemic genotypes and thus the unique genetic diversity of aquatic biota. In particular, the Bulloo is known to support a unique endemic subspecies of Yellowbelly (Faulks et al. 2010) which would be threatened by the introduction of genetically distinct individuals from other catchments.

Movement of water between river catchments is one mechanism for introducing alien aquatic species and genotypes, both of which can readily be delivered as live biota within transferred water (Page et al. 2010). In some Queensland catchments, inter-basin transfer schemes are used to meet urban, industrial and agricultural water needs (e.g. the Water Grid in south-east Queensland, the Mareeba-Dimbulah Irrigation Area). Movement of water in this way can transport aquatic biota, which can affect ecosystems in the receiving rivers (Page et. al 2010).

No such schemes exist in the WPBN plan area; consequently the only potential mode of water and aquatic biota transfer across catchment boundaries is via water storages near catchment boundaries that could receive water from one catchment and release or use it in another. The location and conditions of water entitlements in the Bulloo and Paroo catchments were assessed to determine the hazard they pose to the genetic diversity of the Bulloo.

Ecological value it supports

Ecological outcome 9f (vi), maintaining the unique genetic diversity of aquatic plants and animals within the Bulloo basin

Assessment end point

Absence of translocated genotypes of Yellowbelly in the Bulloo catchment (Faulks et al. 2010).

Measurement end point

The number of existing water entitlements located near the boundary between the Bulloo catchment and the neighbouring Paroo catchment that may allow the transfer of water and Yellowbelly genes into the Bulloo as a result of the Water Resource Plan.

Eco-hydraulic rules

To evaluate the hazard posed to genetic diversity of the Bulloo by water transfers, an assessment was made of the location and conditions of existing water entitlements under the WRP to identify those located near the boundary between the Bulloo catchment and the neighbouring Paroo catchment that may allow the transfer of water. The likelihood of naturally occurring cross-catchment water transfer events between the Paroo and the Bulloo via the Bindegolly-Wyara (Dynevor) valley was also investigated.

Threshold of concern

Any possibility of interbasin transfer of water and Yellowbelly into the Bulloo catchment would represent a risk to the unique genetic diversity of the system, so one entitlement was set as the ToC.

Spatial relevance

The assessment considered entitlements in the Paroo catchment close to the Bulloo.

Warrego catchment ecological risk assessment results

General

The risk to eight ecological asset indicators including ecosystem components and processes (Table 14) was modelled at seven environmental assessment nodes in the Warrego catchment (Figure 8).

Table 14: Surface water ecological assets assessed in the Warrego catchment and their link to hydrology

Ecological asset	Measurement endpoint	Link to hydrology				Number of assessment nodes
		No-flow	Low flows	Medium flows	High flows	
Flow spawning fish	Annual and long-term abundance of Yellowbelly (*Macquaria ambigua*)	✓		✓	✓	7
Migratory fish species	Frequency of longitudinal dispersal opportunities	✓		✓		7
Eastern snake-necked turtle (*Chelodina longicollis*)	Frequency of high stress events	✓			✓	5
Absence of exotic fish species	Frequency of strong recruitment opportunities for the European carp (*Cyprinus carpio*)			✓		5
Floodplain vegetation	Length of spells between floodplain vegetation inundation events				✓	6
Floodplain wetlands	Length of spells between floodplain wetland inundation events				✓	6
Waterholes as refugia	Number and duration of no-flow spells Distance between waterholes	✓	✓			7
Fluvial geomorphology and river forming processes	Frequency of bankfull flow events			✓	✓	7

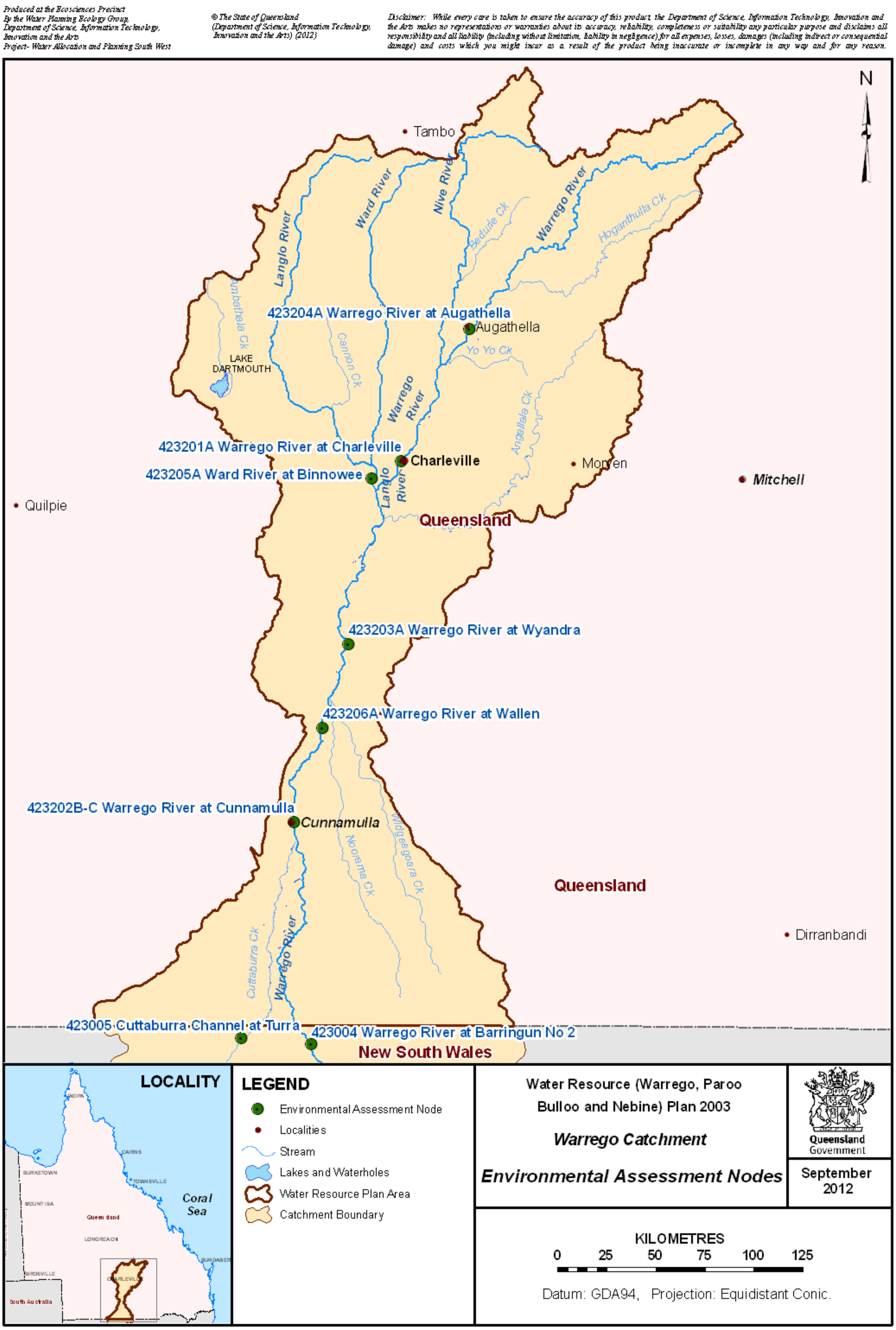

Figure 8: Location of environmental assessment nodes in the Warrego catchment

Flow spawning fish

There were no instances in the simulation where the modelled abundance of the Warrego catchment Yellowbelly population fell below the ToC of 5000 adults (Figure 9), indicating there is a low risk to population viability under both pre-development and full entitlement scenarios. The population is also well connected with the Paroo population, and with other rivers in the Northern Murray Darling Basin.

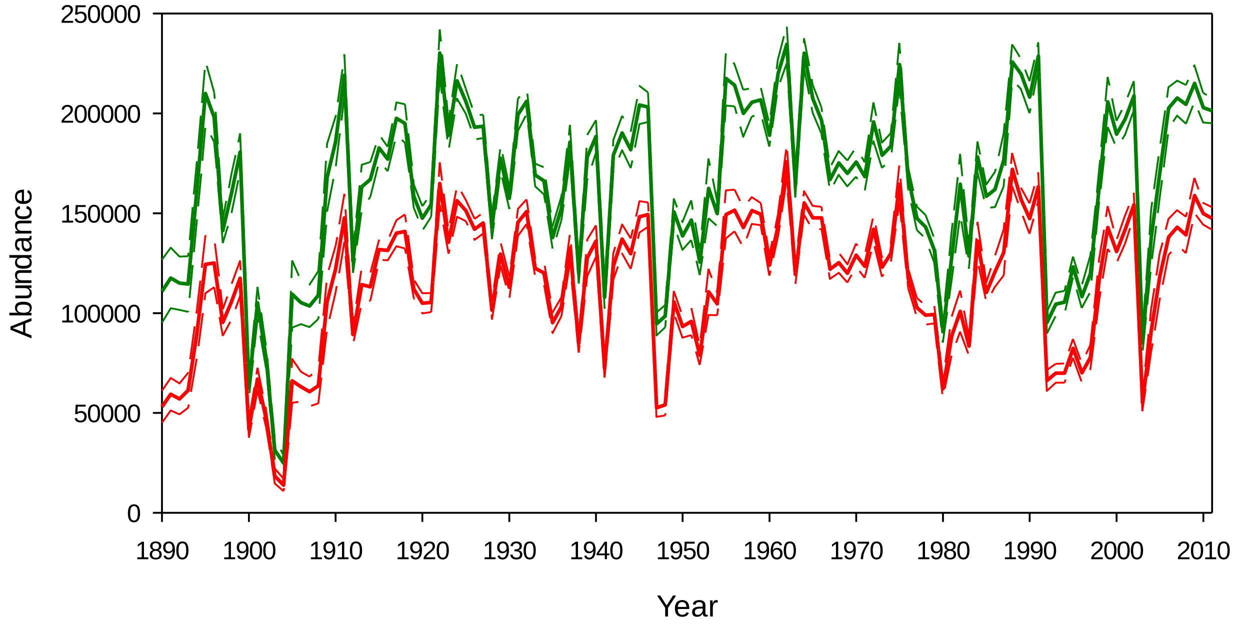

Figure 9**: Yellowbelly population abundance (adults) in the Warrego catchment for the simulation period 1890–2011 [pre-development (green), full entitlement (red) ± std dev (dashed lines)].**

The full entitlement scenario reduced average Yellowbelly abundance by 31% in comparison to the pre-development scenario at the catchment-scale (Figure 9, Table 15). The reduction in population abundance varied across individual environmental assessment nodes, and greater reductions were observed in the upper parts of the catchment. There is > 30 % reduction in modelled long-term average population abundance from pre-development identified at Warrego River at Augathella (423204A) and Warrego River at Wyandra (423203A) Warrego River at Charleville (423201A), and values of 10–30 % reduction in long-term average population abundance from pre-development at Warrego River at Wallen (423206A), Warrego River at Cunnamulla Weir (423202C) and Warrego River at Turra (423005). The only assessment node, where the reduction in population abundance was less than 10% is Warrego River at Barringun (423004).

Table 15: *Macquaria ambigua*–average population abundance and percentage change between development scenarios

Environmental assessment node	Population abundance[1]	
	Pre-development	Full entitlement[2]
Warrego catchment combined[3]	**164 406**	**113 877 (–30.7%)**
Warrego River at Augathella (423204A)	867	592 (–29%)
Warrego River at Charleville (423201A)	6616	2959 (–51.0%)
Warrego River at Wyandra (423203A)	61 556	33 505 (–44.5%)
Warrego River at Wallen (423206A)	26 713	21 691 (–18.6%)
Warrego River at Cunnamulla Weir (423202C)	30 013	25 969 (–13.3%)
Warrego River at Barringun (423004)	4097	3870 (–5.2%)
Warrego River at Turra (423005)	12 363	10 490 (–14.5%)

[1] average abundance of adults in the population from 1890 to 2011

[2] number in brackets indicate % change from pre-development

[3] for all nodes in the Queensland section of the Warrego catchment

There is a gradient of increasing no-flow spells towards the North of the Warrego, caused by rainfall variability and stream network structure. The 95th percentile of lengths of all dry spells for Augathella and Charleville is above 400 days, while the average length for all other reporting nodes in the Bulloo, Paroo, Warrego and Nebine area is half of that at 251 days. This increases the vulnerability of Yellowbelly to temporary loss of local waterholes in the upper Warrego and is compounded by the effected of licensed extraction from waterholes.

The annual volumetric limits for licensed water extraction are generally larger than the estimated volumes of waterholes. Multiple permanent waterholes may be drained under current licence conditions in most simulation years (Figure 10). In the upper Warrego, the location, capacity, and licenced extractions from waterholes, coupled with the longer than average no-flow spells represent a greater threat to local Yellowbelly populations than elsewhere in the catchment (Figure 10).

In the middle of the catchment, around Wyandra and Wallen, waterholes tend to be large and permanent, with only 40% and 20% of the waterholes around each node respectively subject to pumping. Further upstream, at Charleville, there is a mix of permanent and shallower temporary waterholes. Most of the permanent waterholes are subject to pumping, which under full entitlement has a significant effect on the local abundance of Yellowbelly. The waterholes in the farthest upstream reach of the reporting node network, around Augathella, tend to be shallow. While they are generally not subject to pumping licences, the lack of permanent waterholes in the reach, the reductions in source populations downstream and the large distance from those populations means that Yellowbelly abundance is reduced (Figure 10)

To maintain the spatial extent of the Yellowbelly population, upstream waterholes that have dried need to be recolonised from remaining waterholes downstream when flows return. This depends on the behaviour and physical ability of Yellowbelly to move, along with the number of opportunities afforded by the flow conditions. The relatively large distances and less frequent connectivity opportunities in the upper Warrego mean that there may be a significant time lag between when previously dry waterholes refill, and when they are recolonised by Yellowbelly. Another hazard to the population not currently implemented in the model is the drift of larvae downstream, meaning that frequent upstream migration of adults is required to maintain populations in the upper catchment. The likely changes in Yellowbelly abundance between the pre-development and full entitlement flow (including licensed waterhole pumping) scenarios are listed in Table 15.

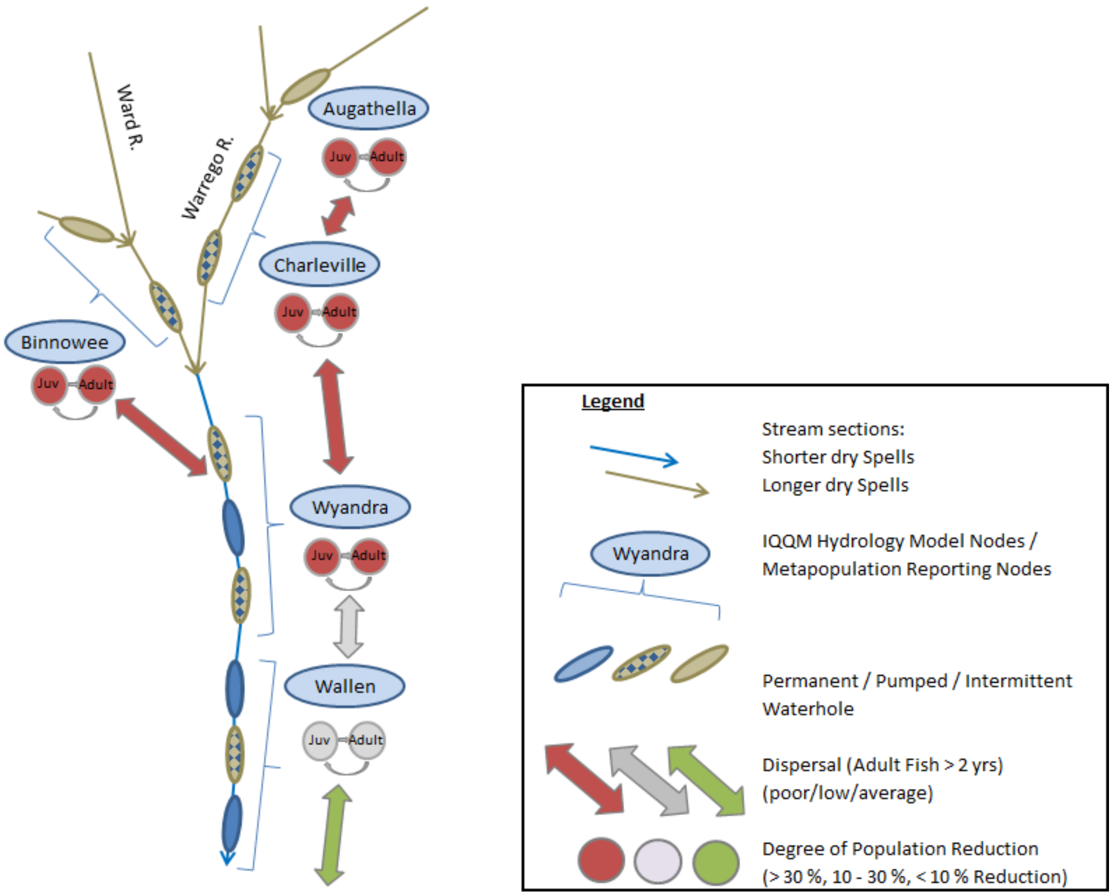

Figure 10: Representation of the threats to Yellowbelly populations in the upper Warrego catchment included in the metapopulation model. The potential loss of individual sub-populations in waterholes affects local fish abundance and recolonisation of waterholes after refilling once stream flow recommences.

Migratory fish species

The opportunities for migratory fish movement within the plan area were modelled at seven environmental assessment nodes in the Warrego catchment (Table 16).

Table 16: Environmental assessment nodes were opportunities for migratory fish movement were modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423001, 423004, 423005, 423202C, 423203A, 423104A, 423206A

The number and duration of connection events were both altered from pre-development conditions at six of the seven assessment nodes under the full entitlement scenario (Table 17). The least affected sites were in the upper part of the catchment (Warrego River at Augathella (423204A) and Warrego River at Charleville (423201A) which showed a 0% and 1.35% reduction respectively in the total duration of connection over the simulation period. The effect was more pronounced for assessment nodes in the mid and lower reaches of the catchment with reductions in the total duration of connection varying between 6.4% and 11.7%.

Table 17: Number and duration of longitudinal connectivity events for migratory fish species over the simulation period

Environmental assessment node	Number and duration of connection events (days)	
	Pre-development	Full entitlement*
Warrego River at Augathella (423204A)	578 events, 10935 days	578 events, 10935 days (0%)
Warrego River at Charleville (423201A)	704 events, 9078 days	668 events, 8955 days (–1.35%)
Warrego River at Wyandra (423203A)	532 events, 21900 days	523 events, 19336 days (–11.7%)
Warrego River at Wallen (423206A)	477 events, 24405 days	502 events, 22827 days (–6.4%)
Warrego River at Cunnamulla Weir (423202C)	538 events, 21334 days	551 events, 19558 days (–8.3%)
Warrego River at Barringun (423004)	520 events, 22102 days	901 events, 19653 days (–11%)
Warrego River at Turra (423005)	615 events, 16292 days	738 events, 15203 days (–6.7%)

* number in brackets indicate % change of connection duration (days) from pre-development

At four assessment nodes this change in longitudinal connectivity opportunities under the full entitlement scenario resulted in an increase in the proportion of moderate risk years (Figures 11 and 12), but these increases were small (less than 5% change). There was no increase in the number of high risk years at any of the assessment nodes as a consequence of the full entitlement scenario.

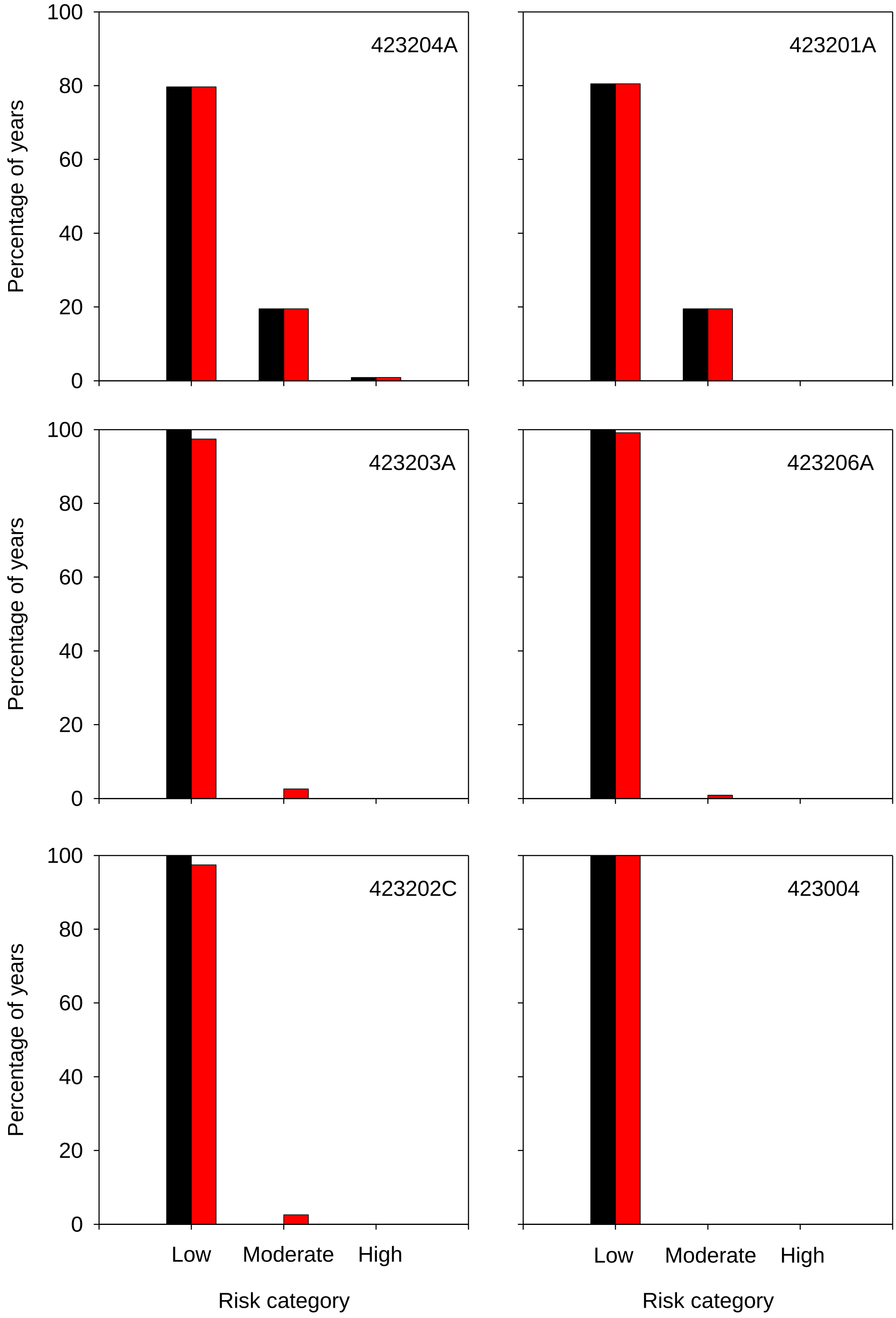

Figure 11: Risk profiles for migratory fish opportunities, as the percentage of years in the simulation period in each risk category (pre-development = black, full entitlement = red).

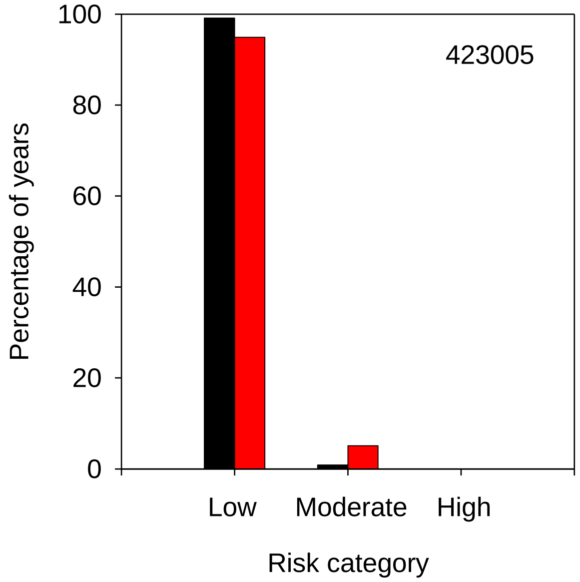

Figure 12: Risk profile for migratory fish opportunities, as the percentage of years in the simulation period in each risk category (pre-development = black, full entitlement = red).

Eastern snake-necked turtle (*Chelodina longicollis*)

The viability of *C. longicollis* populations was modelled at five environmental assessment nodes in the Warrego catchment (Table 18).

Table 18: Environmental assessment nodes were the viability of *C. longicollis* populations were modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423004, 423005, 423202C, 423203A, 423204A

There was no change in either the number or the duration of high stress periods due to the full entitlement scenario at any of the five assessment nodes (Table 19). Although there were long periods of the simulation where the four year flood inundation return frequency ToC was exceeded at all assessment nodes under both flow scenarios, there was no increase in risk due to the full entitlement scenario compared with pre-development (Figure 13). These results suggest that some of the assessment nodes are marginal habitat for this species with high stress periods exceeding 30 years in duration under pre-development hydrology (notably Warrego River at Augathella (423204A), Warrego River at Barringun (423004), and Warrego River at Turra (423005)).

Table 19: High stress periods for the eastern snake-necked turtle

Environmental assessment node	Number of high stress periods	Total duration (days)	Longest spell (years) exceeding the ToC
Warrego River at Augathella (423204A)	10	40 917	39.6
Warrego River at Wyandra (423203A)	24	30 158	4.9
Warrego River at Cunnamulla Weir (423202C)	21	28 229	4.7
Warrego River at Barringun (423004)	7	40 935	33.9
Warrego River at Turra (423005)	7	42 238	30.5

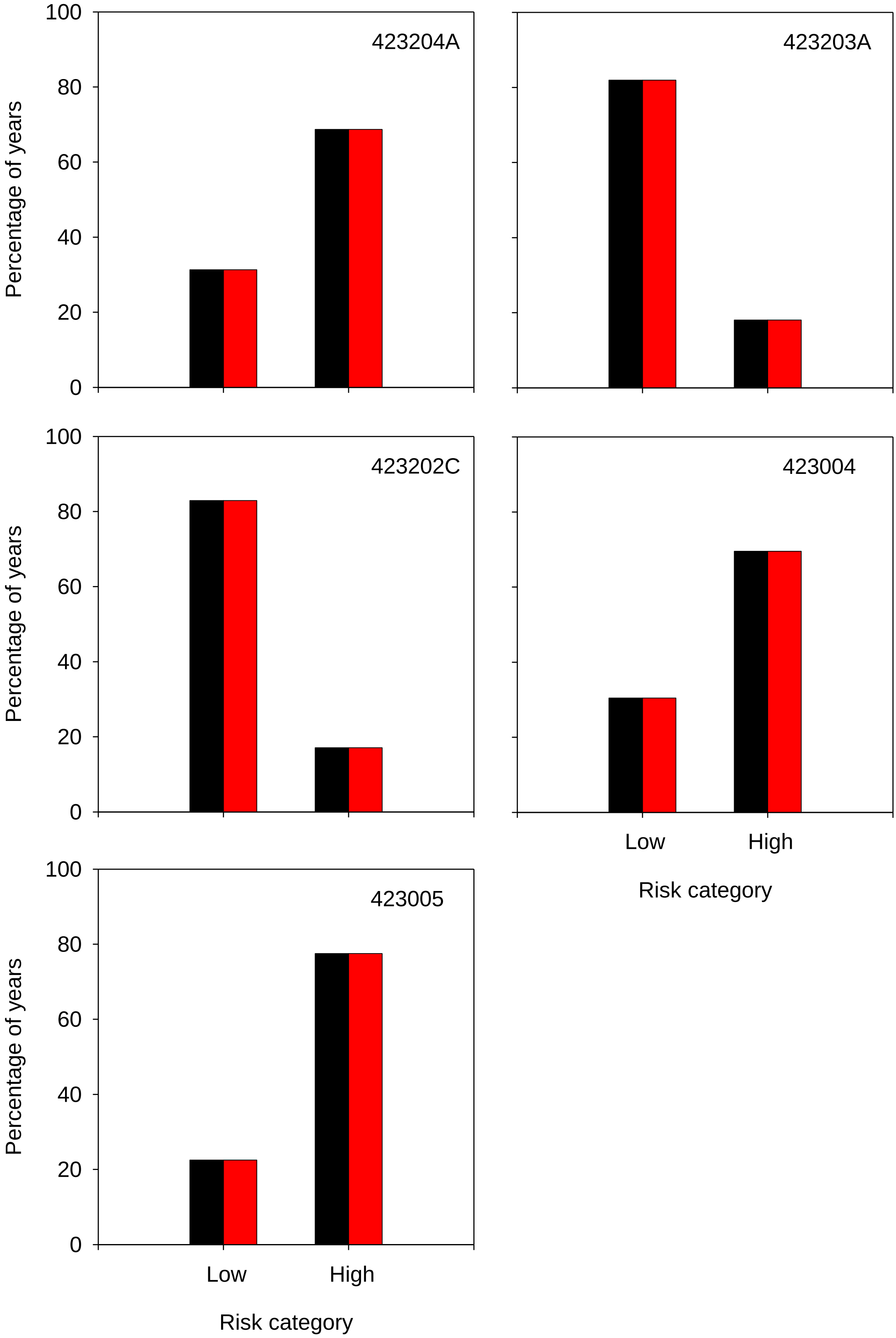

Figure 13: Risk profiles for *Chelodina longicollis*, as the percentage of years in the simulation period in each risk category (pre-development = black, full entitlement = red).

Absence of exotic fish species

Recruitment opportunities for European carp (*C. carpio)* were modelled at five environmental assessment nodes in the Warrego catchment (Table 20).

Table 20: Environmental assessment nodes were carp recruitment opportunities were modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423004, 423005, 423202C, 423203A, 423204A

There was no difference between the scenarios in the percentage of years where strong recruitment was modelled to occur at three of the five environmental assessment nodes (Table 21). However there were small reductions in the duration of these events under the full entitlement scenario at Warrego River at Cunnamulla Weir (423202C) and Warrego River at Turra (423005) of 9.9% and 5.5% respectively. These small reductions slightly reduce the total opportunity for European carp recruitment compared with the pre-development scenario at these nodes, so do not increase hazard to the ecosystem from these exotic fish.

At the remaining two assessment nodes there were small reductions under the full entitlement scenario in both the percentage of years where strong recruitment was modelled to occur and the duration of these events. At Warrego River at Wyandra (423203A) there was a 1.6% reduction of strong recruitment years with the total duration of these events reduced by 9.9%. At the Warrego River at Barringun (423004) there was a 2.5% reduction of strong recruitment years with the total duration of these events reduced by 12.4%. Once again, these reductions reduce the total opportunity for European carp recruitment compared with the pre-development scenario at these nodes, so do not represent a hazard to the ecosystem.

Table 21: Percentage of years in the simulation with *C. carpio* spawning and recruitment opportunities and the total duration over the simulation period of such opportunities

Environmental assessment node	Percentage of years in the simulation with spawning and recruitment opportunities	
	Pre-development	Full entitlement*
Warrego River at Augathella (423204A)	15.6 (2467 days)	15.6 (0%) (2467 days)
Warrego River at Wyandra (423203A)	49.2 (8402 days)	48.4 (–1.6%) (7570 days)
Warrego River at Cunnamulla Weir (423202C)	50.8 (8672 days)	50.8 (0%) (8192 days)
Warrego River at Barringun (423004)	92.6 (17344 days)	90.2 (–2.5%) (15197 days)
Warrego River at Turra (423005)	12.3 (1458 days)	12.3 (0%) (1377 days)

* number in brackets indicate % change from pre-development

Floodplain vegetation and wetlands

The frequency of floodplain vegetation and wetland inundation was modelled at six environmental assessment nodes in the Warrego catchment (Table 22, Figure 8).

Table 22: Environmental assessment nodes were floodplain vegetation inundation was modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423004, 423005, 423202C, 423206A, 423203A, 423204A

The results of floodplain vegetation and wetland inundation event analyses are shown in Tables 23–28 and Figures 14–19. Floodplain vegetation and wetland inundation frequencies under the full entitlement scenario were unchanged from pre-development for floodplain areas associated with five of the six assessment nodes: Warrego River at Augathella (423204A), Warrego River at Wyandra (423203A), Warrego River at Wallen (423206A), Warrego River at Cunnamulla (423202C), and Warrego River at Turra (423005).

At Warrego River at Barringun (423004), spells between inundation events of floodplain vegetation patches and wetlands at the small flood threshold (5000 ML/day) have been shortened under the full entitlement scenario. Under pre-development 74% of spells between floods lasted one year or less, whereas under full entitlement 82% of spells were shorter than a year. Spells between inundation events of floodplain vegetation patches and wetlands at medium-flood thresholds (5431, 7970 and 8470 ML/day) have been lengthened under full entitlement. This indicates that these floodplain features are inundated less often under full entitlement than under pre-development. These results may reflect the influence of flood event harvesting upstream of this node; some flood events that would inundate these features under pre-development may not do so under full entitlement due to reduced river levels.

Consequently, the effected floodplain wetlands and vegetation patches may be under water stress more often under full entitlement than pre-development, however the duration of spells between floods under full entitlement is still within the range experienced under pre-development. There is no difference between pre-development and full entitlement in the frequency of flooding at the large-flood threshold (19140 ML/day).

Table 23: 423204A Floodplain vegetation and wetlands

Species and wetlands	6827 ML/day		11 400 ML/day		30 557 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Eucalyptus camaldulensis	1	8.1	1	40.6	1	0
Eucalyptus coolabah	1	8.1	1	40.6	1	0
Floodplain wetlands	0	0	0	0	1	0.03

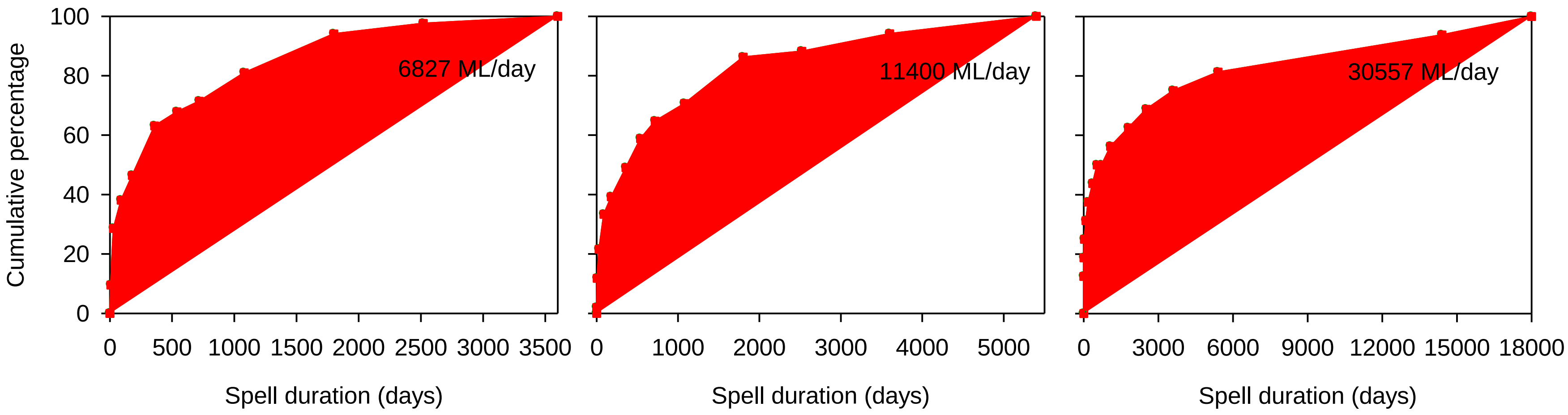

Figure 14: 423304A Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Table 24: 423203A Floodplain vegetation and wetlands

Species and wetlands	3431 ML/day		7935 ML/day		20 572 ML/day		64 244 ML/day		244 770 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	4	58	0	0	0	0	1	77	29	138.9
Eucalyptus camaldulensis	4	103.1	2	89.3	0	0	2	0.1	4	1.1
Eucalyptus ochrophloia	2	53	2	89.4	0	0	0	0	39	214.2
Eucalyptus coolabah	5	116.5	3	143.8	2	111.4	4	137.9	36	273.1
Muehlenbeckia florulenta	0	0	0	0	0	0	1	0.02	0	0
Floodplain wetlands	0	0	0	0	0	0	3	0.17	2	0.09

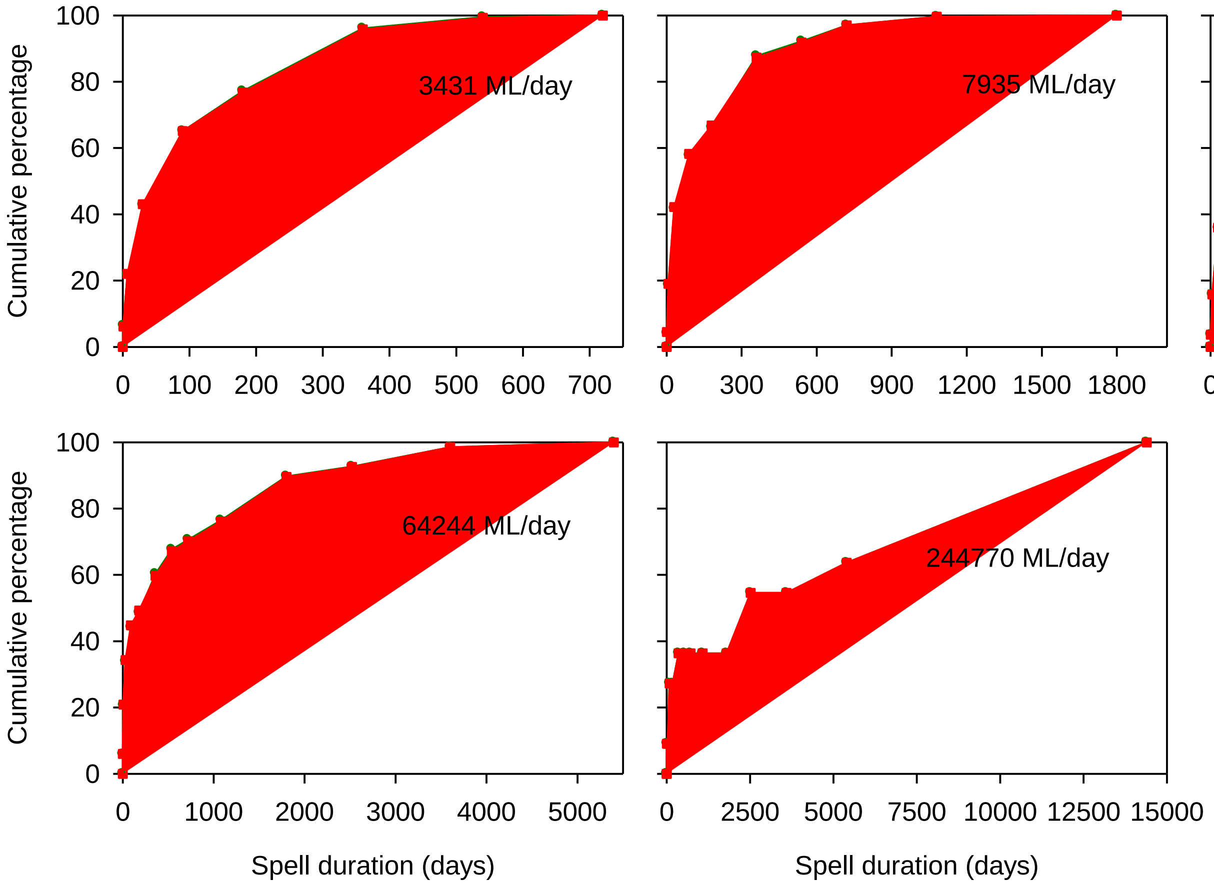

Figure 15: 423203A Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Table 25: 423206A Floodplain vegetation and wetlands

Species	3044 ML/day		20 487 ML/day		39 152 ML/day		69 317 ML/day		208 015 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	3	49.1	1	8.1	0	0	0	0	1	0.1
Eucalyptus camaldulensis	1	21.6	0	0	0	0	0	0	0	0
Eucalyptus ochrophloia	1	12.8	5	56.4	0	0	2	10.2	1	0.1
Eucalyptus coolabah	4	116.2	5	56.4	1	3.6	2	18.7	2	10.5
Floodplain wetlands	0	0	0	0	0	0	0	0	0	0

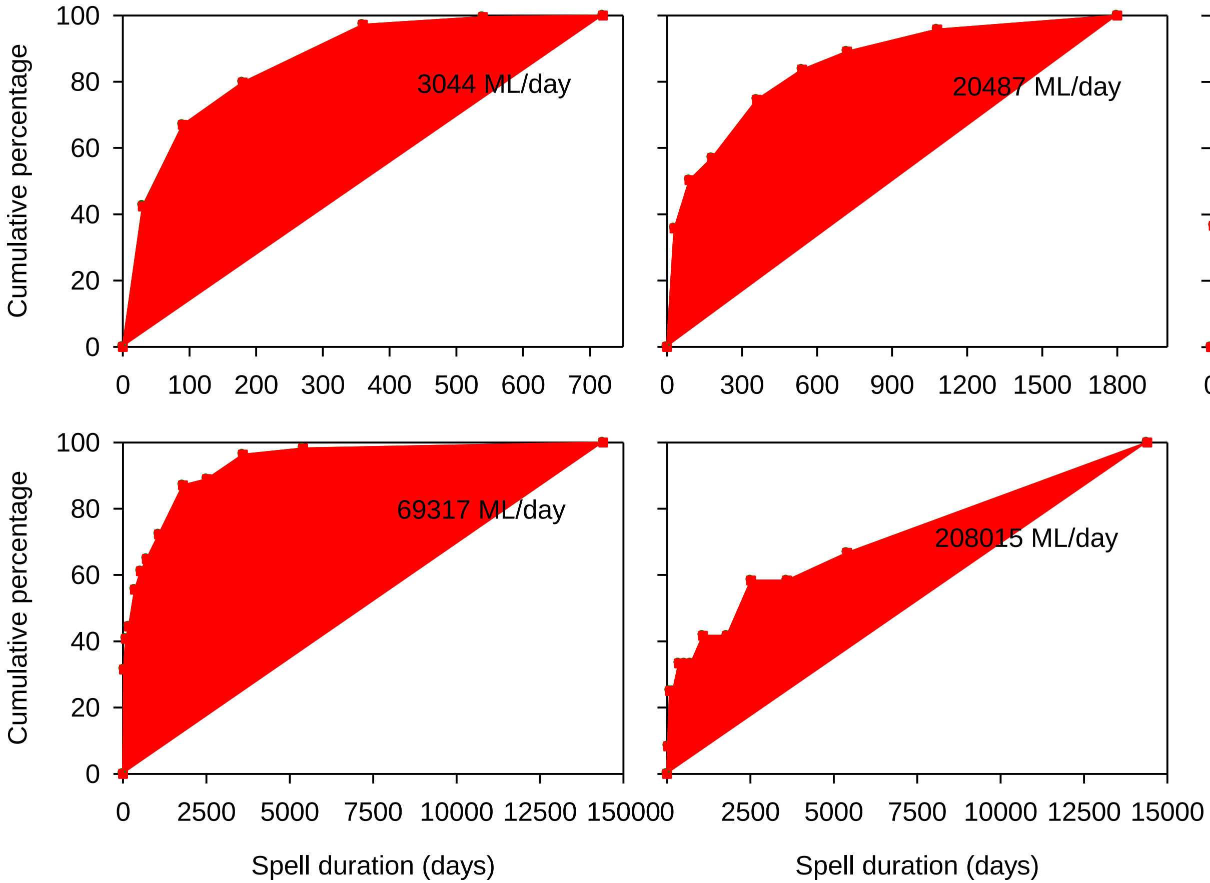

Figure 16: 423206A Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Table 26: 423202C Floodplain vegetation and wetlands

Species	20 000 ML/day		47 000 ML/day		50 498 ML/day		68 209 ML/day		126 466 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	71	2870.9	35	408.5	5	195.4	13	224.7	32	171.7
Eucalyptus camaldulensis	18	347.7	11	90.4	0	0	5	29	9	56.9
Eucalyptus ochrophloia	17	343.9	16	164.9	0	0	6	30.9	8	26.7
Eucalyptus coolabah	74	3023.4	35	408.5	5	195.4	13	224.7	33	173.4
Eucalyptus largiflorens	0	0	0	0	0	0	0	0	1	7.5
Muehlenbeckia florulenta	0	0	4	1.4	0	0	0	0	3	7.6
Floodplain wetlands	0	0	5	1.84	0	0	0	0	2	9.47

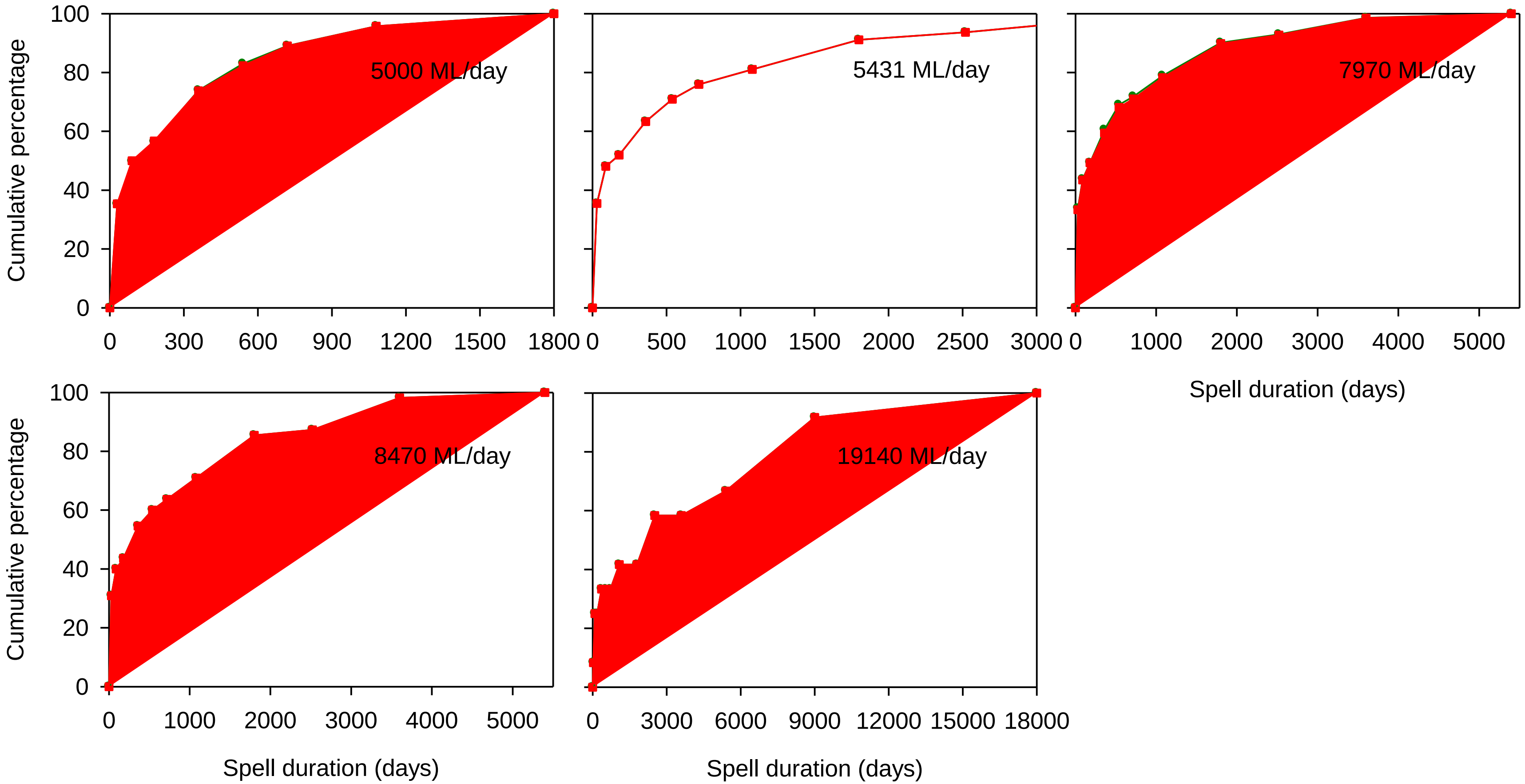

Figure 17: 423202C Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Table 27: 423004 Floodplain vegetation and wetlands

Species	5000 ML/day		5431 ML/day		7970 ML/day		8470 ML/day		19 140 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	71	2870.9	35	408.5	5	195.4	13	224.7	32	171.7
Eucalyptus camaldulensis	18	347.7	11	90.4	0	0	5	29	9	56.9
Eucalyptus ochrophloia	17	343.9	16	164.9	0	0	6	30.9	8	26.7
Eucalyptus coolabah	74	3023.4	35	408.5	5	195.4	13	224.7	33	173.4
Eucalyptus largiflorens	0	0	0	0	0	0	0	0	1	7.5
Muehlenbeckia florulenta	0	0	4	1.4	0	0	0	0	3	7.6
Floodplain wetlands	0	0	5	1.84	0	0	0	0	2	9.47

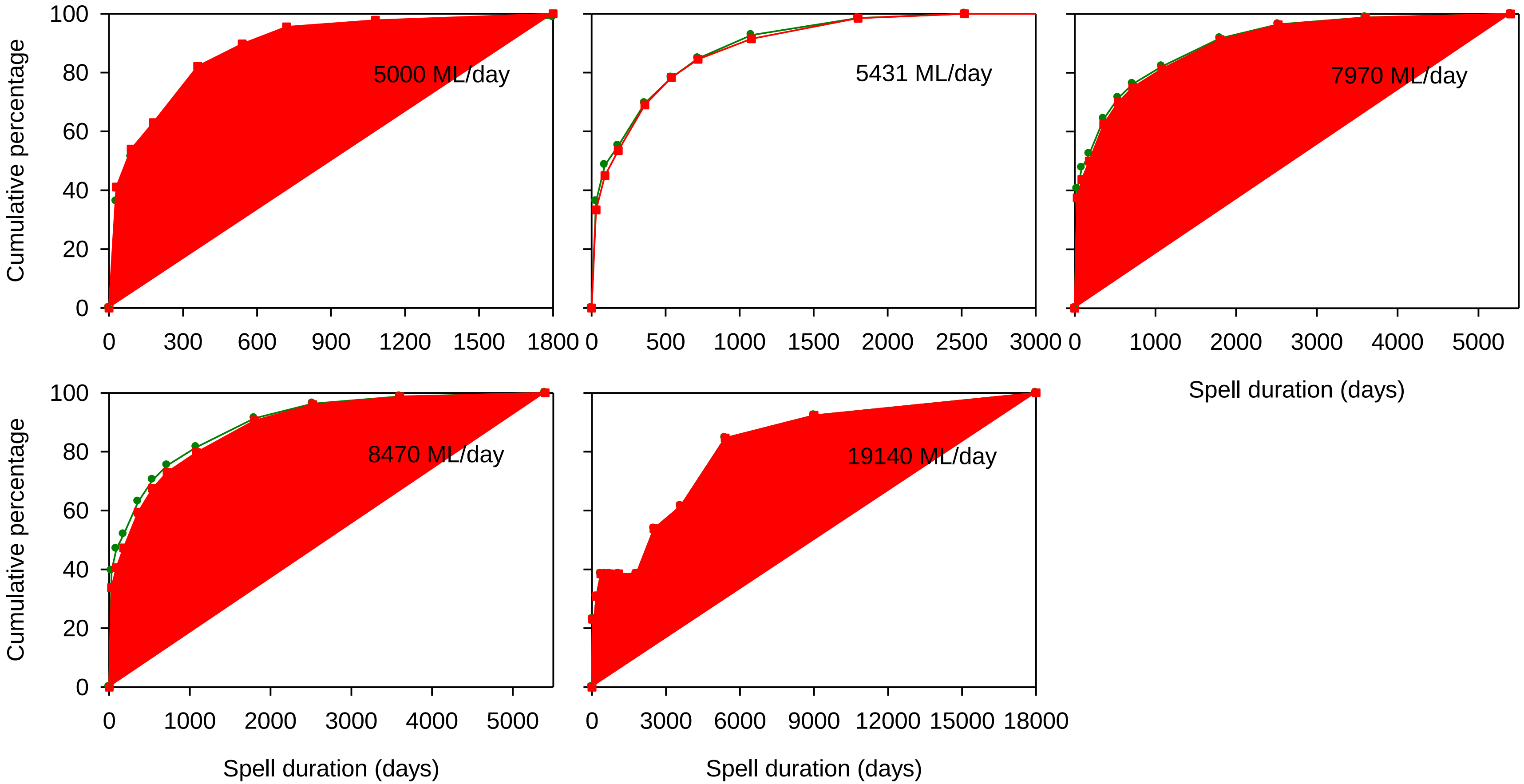

Figure 18: 423004 Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Table 28: 423005 Floodplain vegetation and wetlands

Species	1322 ML/day		6691 ML/day		16 115 ML/day		46 215 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	0	0.00	2	11.30	0	0.00	0	0.00
Eucalyptus camaldulensis	0	0.00	3	25.20	0	0.00	0	0.00
Eucalyptus ochrophloia	0	0.00	4	28.60	0	0.00	0	0.00
Eucalyptus coolabah	1	18.50	8	49.60	4	28.60	2	1.30
Floodplain wetlands	0	0	0	0	0	0	1	0.03

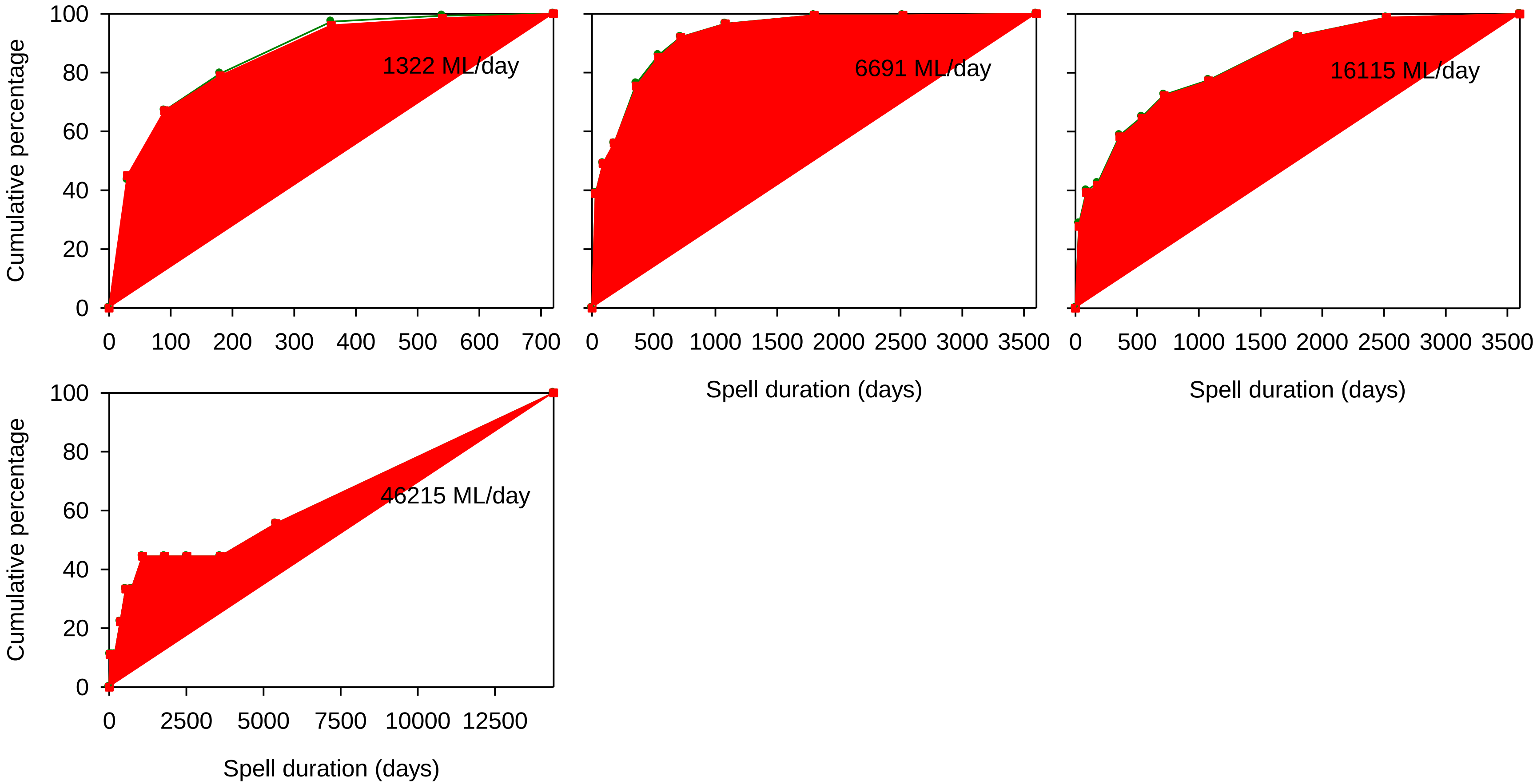

Figure 19: 423005 Floodplain area inundation duration plots (pre-development=green, full entitlement=red)

Waterholes as refugia

Waterholes as refugia were assessed at eight nodes in the Warrego catchment for no-flow spells (Table 29) and the entire catchment was assessed for waterhole pumping (i.e. allocations with a nil passing flow condition).

Table 29: Environmental assessment nodes where no- flow spells for refugial waterholes were modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423001, 423004, 423005, 423202C, 423203A, 423104A, 423205A, 423206A

No-flow spells

The number of no-flow spells over the simulation period was similar between scenarios at most environmental assessment nodes. Exceptions were Warrego River at Barringun (423004) where there were over 800 additional spells under full entitlement, and Warrego River at Turra (423005) where there were over 100 additional spells under full entitlement (Table 30). Cumulative frequency plots (Figure 20) show that this is due to more short-duration spells (typically lasting two to four months) at these two sites under the full entitlement scenario, but there are no other changes to the frequency distributions of spells between the scenarios at these or other Warrego environmental assessment nodes. These results suggest that there is no increased hazard to the function of waterholes as drought refuges for biota as most waterholes are likely to persist for longer than two to four months without flow (see Silcock 2009; DERM 2010b).

The maximum duration of no-flow spells over the simulation period was also similar between scenarios for most environmental assessment nodes, with increases of only a few days at most under full development (Table 30). There was a small decrease in the duration of the longest spell under full development at Warrego River at Barringun (423004). Warrego River at Cunnamulla Weir (423202C) was an exception as the maximum spell duration increased by 91 days under full entitlement, representing a 25% increase in duration over the pre-development maximum. This suggests that there is an increased probability of some waterholes that were permanent under pre-development conditions drying under full entitlement at this site, thus posing an increased hazard to biota that rely on waterholes as drought refuges.

Table 30: Number and maximum duration of no-flow spells over the simulation period

Environmental assessment node	Number and maximum duration of spells	
	Pre-development	Full entitlement*
Warrego River at Augathella (423204A)	562 spells, 1114 days	562 spells, 1114 days (0%)
Warrego River at Charleville (423201A)	691 spells, 714 days	663 spells, 714 days (0%)
Warrego River at Wyandra (423203A)	531 spells, 465 days	523 spells, 474 days (+2%)
Warrego River at Wallen (423206A)	491 spells, 465 days	513 spells, 474 days (+2%)
Warrego River at Cunnamulla Weir (423202C)	538 spells, 370 days	551 spells, 461 days (+25%)
Warrego River at Barringun (423004)	525 spells, 361 days	901 spells, 341 days (–6%)
Warrego River at Turra (423005)	624 spells, 430 days	735 spells, 430 days (0%)

* numbers in brackets indicate % change in maximum spell duration from pre-development

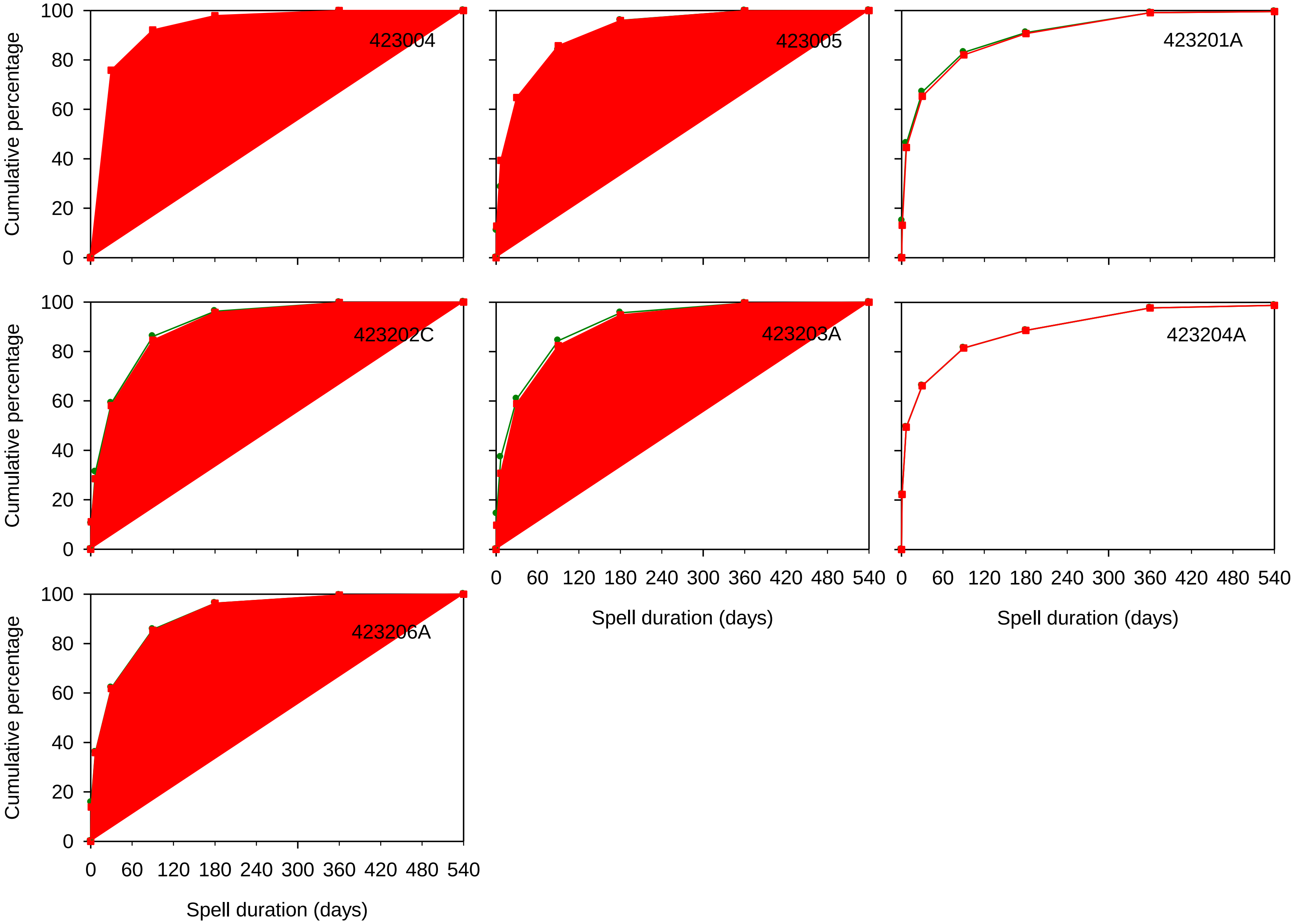

Figure 20: No-flow spell duration plots for each assessment node (pre-development=green, full entitlement=red). Plots show the proportion of spells which lasted at or longer than the number of days read off the x axes.

Waterhole pumping

Of 133 waterholes identified in the Warrego catchment (Silcock 2009), only 17 are subject to licensed extraction (Table 31; Appendix 3). Under pre-development conditions 15% of the distances between nearest neighbour waterholes were greater than 20 km, representing a natural limitation to fish dispersal in some reaches. Under the full entitlement scenario, four additional gaps of greater than 20 km were generated, representing a 2% increase in the number of nearest neighbour distances between waterholes posing a hazard to migratory species.

Table 31: Spatial distribution of refugial waterholes in the Warrego catchment in relation to the 20 km fish dispersal threshold, for pre-development and full entitlement scenarios.

Catchment	No. of waterholes (No. with extraction)	Gaps > 20 km under pre-development		Gaps > 20 km under full entitlement		Reaches where hazard increased	
		Number	%	Number	%	Number	%
Warrego	133 (17)	38	15	42	17	4	2

Fluvial geomorphology and river forming processes

River forming processes were modelled at seven environmental assessment nodes in the Warrego catchment (Table 32).

Table 32: Environmental assessment nodes where river forming processes were modelled

Catchment	Gauging Station(s) corresponding to IQQM Node(s)
Warrego	423001, 423004, 423005, 423202C, 423203A, 423104A, 423206A

There was no change in the number or total duration of bankfull events under the full entitlement scenario for three assessment nodes–Warrego River at Augathella (423204A), Warrego River at Charleville (423201A), Warrego River at Charleville (423201A), and Warrego River at Wyandra (423203A), and a 0.6% decrease in total duration at Warrego River at Wallen (423206A) (Table 33). The three nodes south of Charleville experienced a decrease in total duration of between 2.9% and 7.9% compared to pre-development, and there was also a very small decrease at Warrego River at Wallen (423206A). These decreases pose additional threats to the persistence of waterholes to act as refuges during no-flow spells, because they represent reductions in scouring of the waterholes to maintain their depth.

Table 33: Number of bankfull spells over the simulation period under both development scenarios

Environmental assessment node	Number and total duration of bankfull events over the simulation period	
	Pre-development	Full entitlement*
Warrego River at Augathella (423204A)	72 events, 187 days	72 events, 187 days (0%)
Warrego River at Charleville (423201A)	68 events, 215 days	68 events, 215 days (0%)
Warrego River at Wyandra (423203A)	72 events, 306 days	72 events, 306 days (0%)
Warrego River at Wallen (423206A)	195 events, 1184 days	194 events, 1177 days (–0.6%)
Warrego River at Cunnamulla Weir (423202C)	198 events,1218 days	193 events, 1182 days (–2.9%)
Warrego River at Barringun (423004)	137 events, 717 days	129 events, 660 days (–7.9%)
Warrego River at Turra (423005)	225 events, 1053 days	218 events, 998 days (–5.2%)

* numbers in brackets indicate % change in duration from pre-development

Paroo catchment ecological risk assessment results

General

The risk to two ecological asset indicators including ecosystem components and processes (Table 1) was modelled at two environmental assessment nodes in the Paroo catchment (Figure 21). Six ecological indicators were not modelled as there was no difference between the flow regimes under the pre-development and full entitlement IQQM scenarios in the Paroo catchment, thus reflecting no increased risk to these assets from the current water resource plan. Because there is no change in the occurrence of flood events, this also means that the full entitlement scenario meets outcome 9f(v) in the WRP, relating to the provision of bird-breeding events in the Paroo Overflow Lakes.

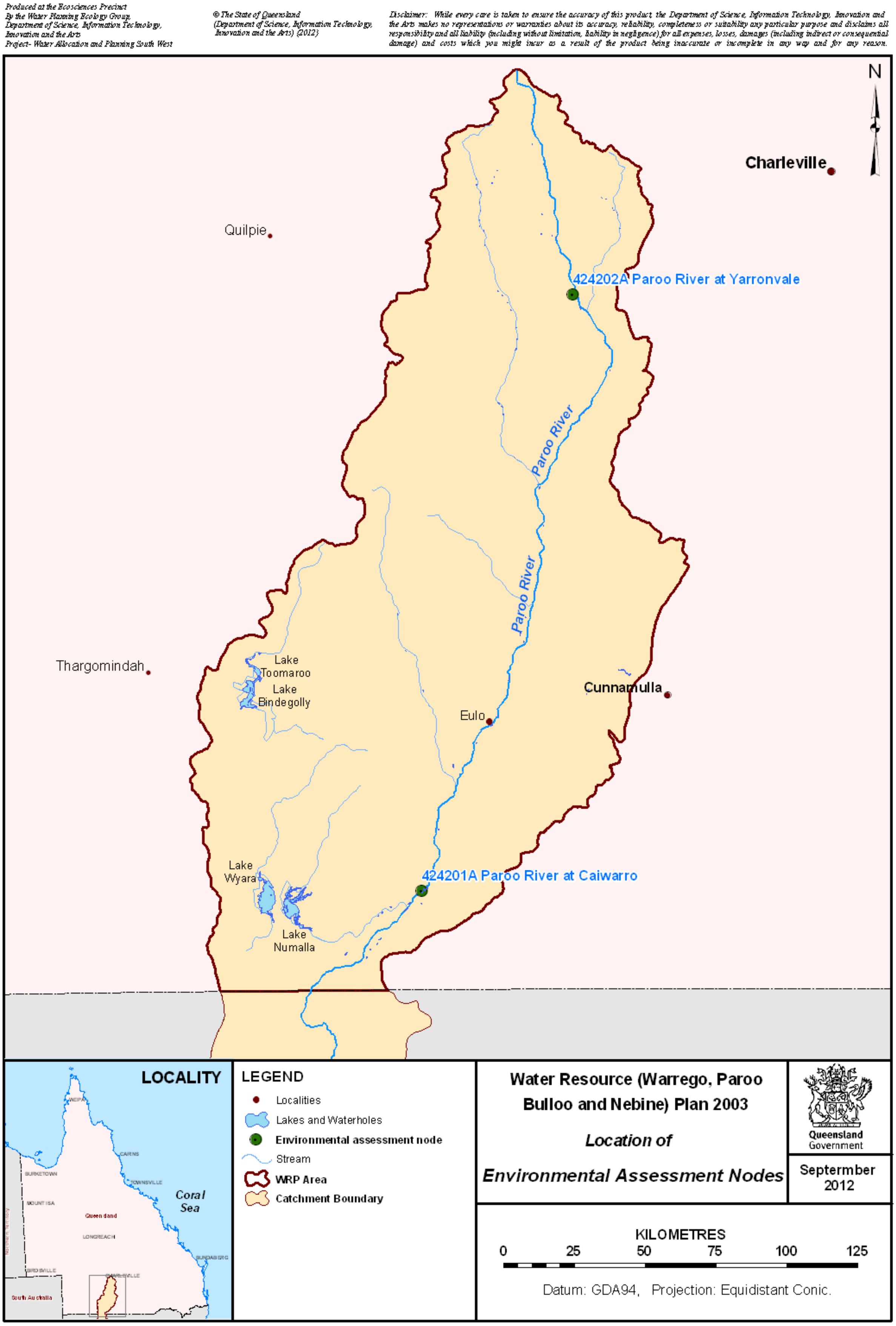

Figure 21: Location of environmental assessment nodes in the Paroo catchment

Flow spawning fish

Yellowbelly abundance was modelled at both environmental assessment nodes in the Paroo catchment. There were no periods in the simulation where the catchment-scale population abundance fell below the ToC of 5000 adults, indicating a low risk to Yellowbelly population viability in the Paroo catchment (Figure 22). There are up to three flood channels linking the Paroo population with the Warrego population of Golden Perch, which are providing regular gene flow and increase the effective population size.

Reduction in modelled abundance under the full entitlement scenario, at both catchment and individual environmental nodes was less than 10% from pre-development abundance (Figure 22, Table 34).

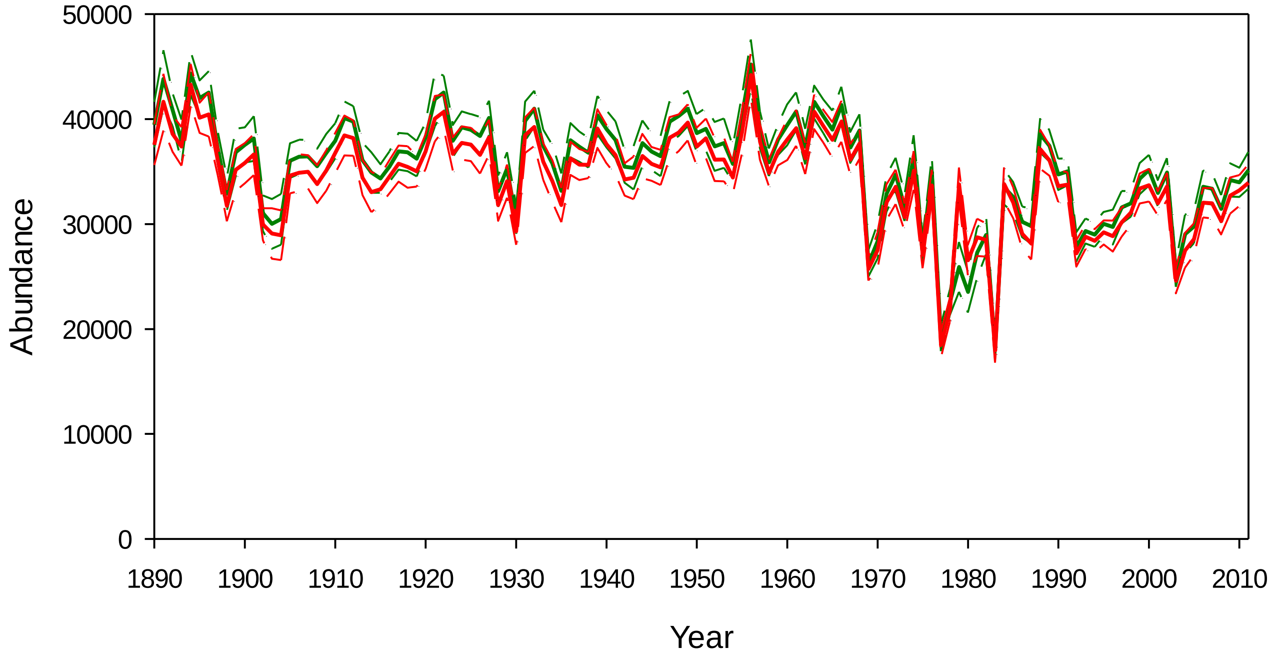

Figure 22: Yellowbelly population abundance (adults) in the Paroo catchment for the simulation period 1890–2011 [pre-development (green), full entitlement (red) ± std dev (dashed lines)].

Table 34: *Macquaria ambigua*–average population abundance and percentage change between development scenarios

Environmental assessment node	Population abundance[1]	
	Pre-development	Full entitlement[2]
Paroo catchment combined	**18,206**	**17,959(–3%)**
Paroo River at Yarronvale (424202A)	12,268	12,317 (0.4%)
Paroo River at Caiwarro (424201A)	12,542	12,169 (–3.0%)

[1] average abundance of adults in the population

[2] number in brackets indicate % change from pre-development

Waterholes as refugia

Waterhole pumping

There were two waterholes subject to licensed extraction of a total of 90 identified in the Paroo catchment (Table 35; Appendix 3). Under pre-development conditions 13% of the distances between nearest neighbour waterholes were greater than 20 km, representing a natural limitation to fish dispersal in some reaches. This was not changed under the full entitlement scenario, meaning that it posed no additional hazard to migratory species.

Table 35: Spatial distribution of refugial waterholes in the Paroo catchment in relation to the 20 km fish dispersal threshold, for pre-development and full entitlement scenarios.

No. of waterholes (No. with extraction)	Gaps > 20 km under pre-development		Gaps > 20 km under full entitlement		Reaches where hazard increased	
	Number	%	Number	%	Number	%
90 (2)	38	13	38	13	0	0

Bulloo catchment ecological risk assessment results

General

The risk to two ecological asset indicators including ecosystem components and processes (Table 1) was modelled at three environmental assessment nodes in the Bulloo catchment (Figure 23). Six ecological indicators were not modelled as there was no difference between the flow regimes under the pre-development and full entitlement IQQM scenarios in the Bulloo catchment, thus reflecting no increased risk to these assets from the current water resource plan. Because there is no change in the occurrence of flood events, this also means that the full entitlement scenario meets outcome 9f(v) in the WRP, relating to the provision of bird-breeding events in the Bulloo Lakes.

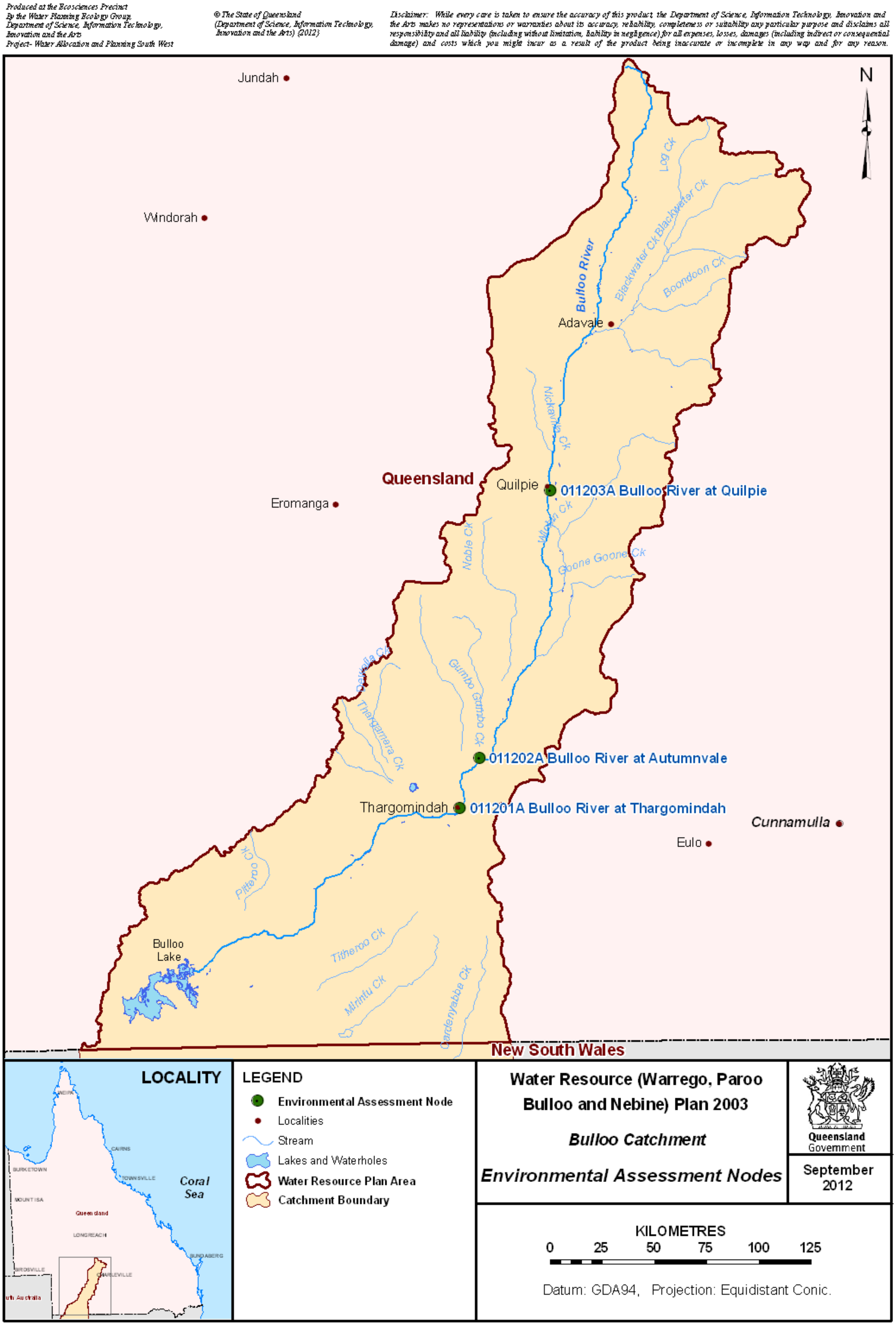

Figure 23: Location of environmental assessment nodes in the Bulloo catchment

Flow spawning fish

Yellowbelly abundance was modelled at two environmental assessment nodes in the Bulloo catchment. A catchment-scale ToC of 5000 adults was established based on a minimum population density threshold to support the long term viability of the Yellowbelly.

There were no periods in the simulation where the catchment-scale population abundance fell below the ToC, indicating low risk to Yellowbelly population viability in the Bulloo catchment (Figure 24). Reduction in modelled abundance under the full entitlement scenario, at both catchment and individual environmental nodes was less than 10 % from pre-development abundance (Table 36).

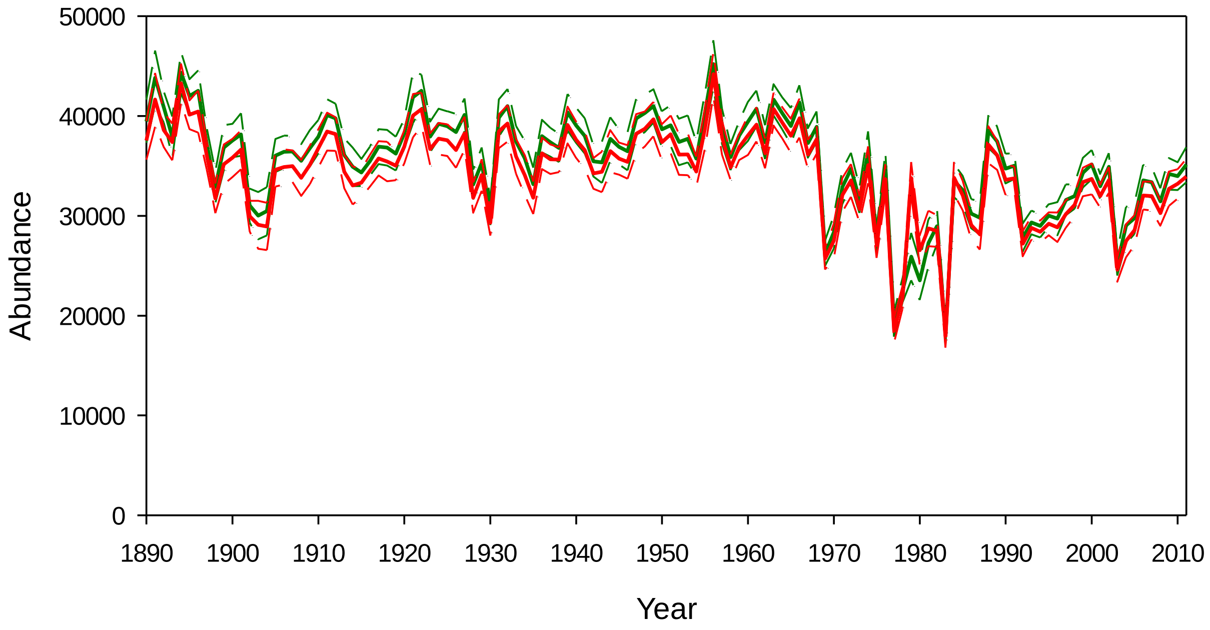

Figure 24: Yellowbelly population abundance (adults) in the Bulloo catchment for the simulation period 1890–2011 [pre-development (green), full entitlement (red) ± std dev (dashed lines)].

Table 36: *Macquaria ambigua*–average population abundance and percentage change between development scenarios

Environmental assessment node	Population abundance[1]	
	Pre-development	Full entitlement[2]
Bulloo catchment combined	34,361	33,353 (2.9%)
Bulloo River at Autumnvale (011202A)	5,682	5,684 (0%)
Bulloo River at Quilpie (011203A)	10,623	9,618 (–9.5%)

[1] average abundance of adults in the population

[2] number in brackets indicate % change from pre-development

Waterholes as refugia

Waterhole pumping

There were two waterholes subject to licensed extraction of a total of 90 identified in the Bulloo catchment (Table 37, Appendix 3). Under pre-development conditions 16% of the distances between nearest neighbour waterholes were greater than 20 km, representing a natural limitation to fish dispersal in some reaches. This was not changed under the full entitlement scenario, meaning that it posed no additional hazard to migratory species.

Table 37: Spatial distribution of refugial waterholes in the Paroo catchment in relation to the 20 km fish dispersal threshold, for pre-development and full entitlement scenarios.

No. of waterholes (No. with extraction)	Gaps > 20 km under pre-development		Gaps > 20 km under full entitlement		Reaches where hazard increased	
	Number	%	Number	%	Number	%
84 (2)	36	16	36	16	0	0

Genetic diversity of aquatic biota in the Bulloo catchment

There are two water allocations in the Bulloo, both of which are nil-flow (i.e. not water harvesting). The entitlements have a fixed location, well away from the catchment boundary and there are no provisions under the WRP for pumping across the catchment boundary. In the neighbouring Paroo catchment, there are two water allocations, and one unconverted water licence for urban use, all of which are located more than 30 km from the catchment boundary with the Bulloo. There are three properties authorised to take overland flow on the western side of the Paroo River, far from the catchment boundary and again, the WRP and ROP do not contain any provisions that would allow pumping of water between catchments (Warren Blackburn and Peter Brownhalls pers. comm.). Based on this information, the distribution and operation of entitlements does not increase risk to the genetic diversity of the Bulloo catchment.

Lakes Bindegolly and Wyara are large, shallow salt lakes that lie within the Dynevor Valley, nominally at the western edge of the Paroo catchment (Power et al. 2007). Geological evidence suggests that the Dynevor Valley once formed part of the flow path of the Bulloo River (Power et al. 2007). Today, the lakes are isolated from the Bulloo and are filled from relatively small, internally-draining catchments (DSEWPC 2010). While the lakes are known to have received waters from and contributed to floodwaters on the Paroo floodplain, hydrological connections to the Paroo River are relatively rare (Timms 1998; Power et al. 2007; Mark Handley pers. comm.). The Dynevor valley is considered an unlikely route for the transfer of aquatic biota between the Paroo and Bulloo because of the infrequent flow events, barriers formed by sand dunes and the inhospitable nature of the lakes due to their salinity (Power et al. 2007, Mark Handley pers. comm.). Further, because connections are only possible during very large floods, the frequency is not affected by current water resource management strategies.

While water transfer and hydrologic connectivity are unlikely to introduce new species or genotypes into the Bulloo, other anthropogenic and natural mechanisms exist for the transport of biota between catchments. For example, there are currently no European carp populations in Cooper Creek in the Lake Eyre Basin, however carp carcases have been found on the banks of the Creek, believed to be discarded bait used by anglers (Wager & Unmack 2000). It has been further proposed that a natural mechanism exists for the transport of carp between catchments, suggesting that the adhesive eggs of carp could become attached to the feet of waterbirds and carried between water bodies (Gilligan & Rayner 2007).

Nebine catchment ecological risk assessment results

General

The risk to six ecological asset indicators including ecosystem components and processes (Table 38) was modelled at one environmental assessment node in the Nebine catchment (Figure 25).

Table 38: Surface water ecological assets assessed in the Nebine catchment and their link to hydrology

Ecological asset	Assessment endpoint	Link to hydrology				Number of assessment nodes
		No-flow	Low flows	Medium flows	High flows	
Flow spawning fish	Population viability of Yellowbelly (*Macquaria ambigua*)	✓		✓	✓	2
Migratory fish species	Frequency of longitudinal dispersal opportunities	✓		✓		1
Absence of exotic fish species	Frequency of strong recruitment opportunities for the European carp (*Cyprinus carpio*)			✓		1
Floodplain vegetation	Length of spells between floodplain vegetation inundation events				✓	1
Floodplain wetlands	Length of spells between floodplain wetland inundation events				✓	1
Waterholes as refugia	Spells of no-flow isolation Distance between waterholes	✓	✓			1

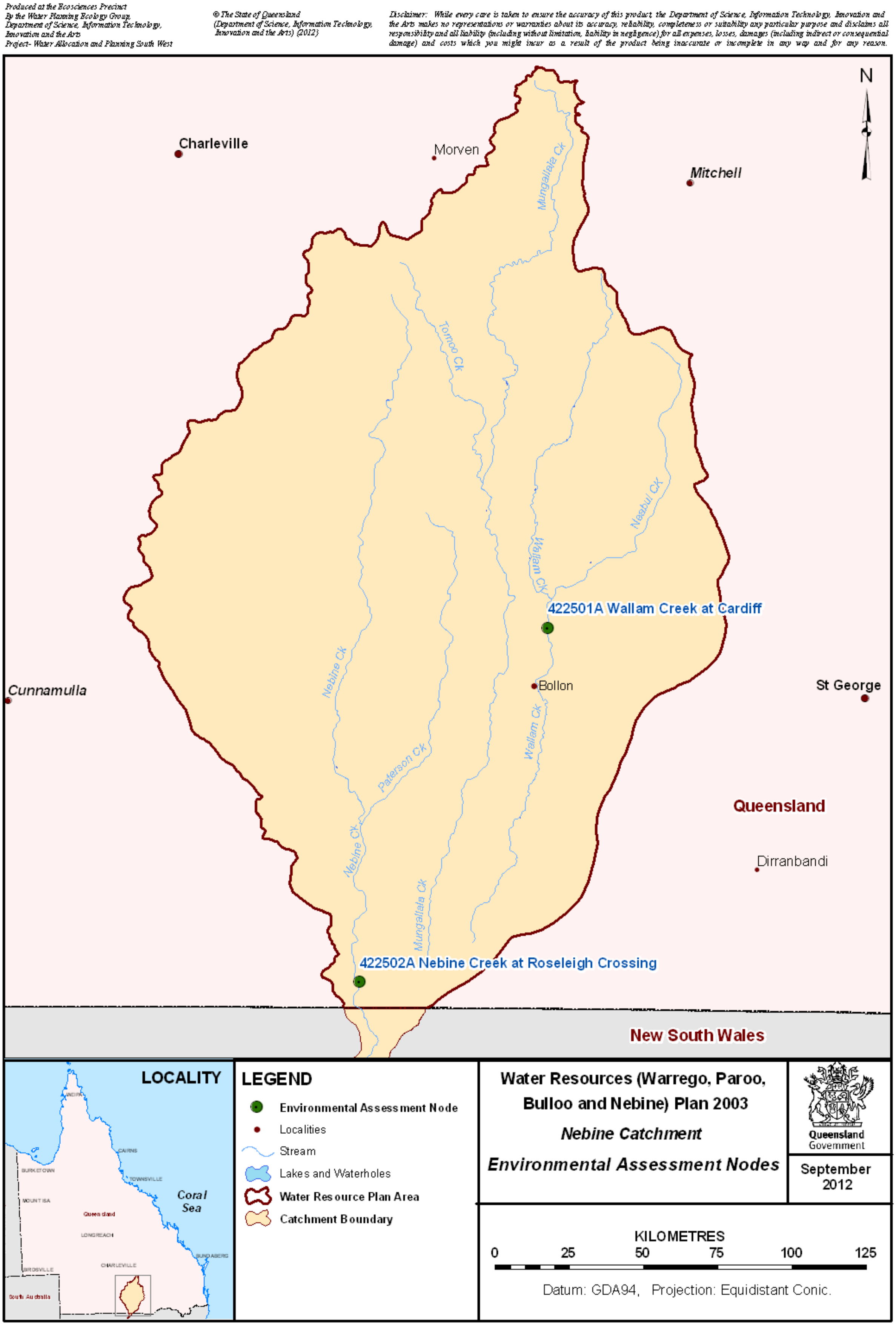

Figure 25: Location of environmental assessment nodes in the Nebine catchment

Flow spawning fish

Yellowbelly abundance was modelled at two environmental assessment nodes in the Nebine catchment. The annual catchment-scale population abundance of Yellowbelly did not fall below the ToC of 5000 adults under either scenario (Figure 26, Table 39) however abundances were low at times under both pre-development (minimum annual abundance 8670 individuals) and full development (6537 individuals) scenarios. These results suggest that the Nebine catchment may provide marginal habitat for Yellowbelly even under the pre-development flow regime. The population was however modelled together with the populations of the Culgoa and Balonne rivers to represent the sub-population exchanges during big flood events in the Culgoa flood plain, approximately every 15 years. These events are likely to provide sufficient gene flow and reduce the risk of low population numbers to population health even for lower densities.

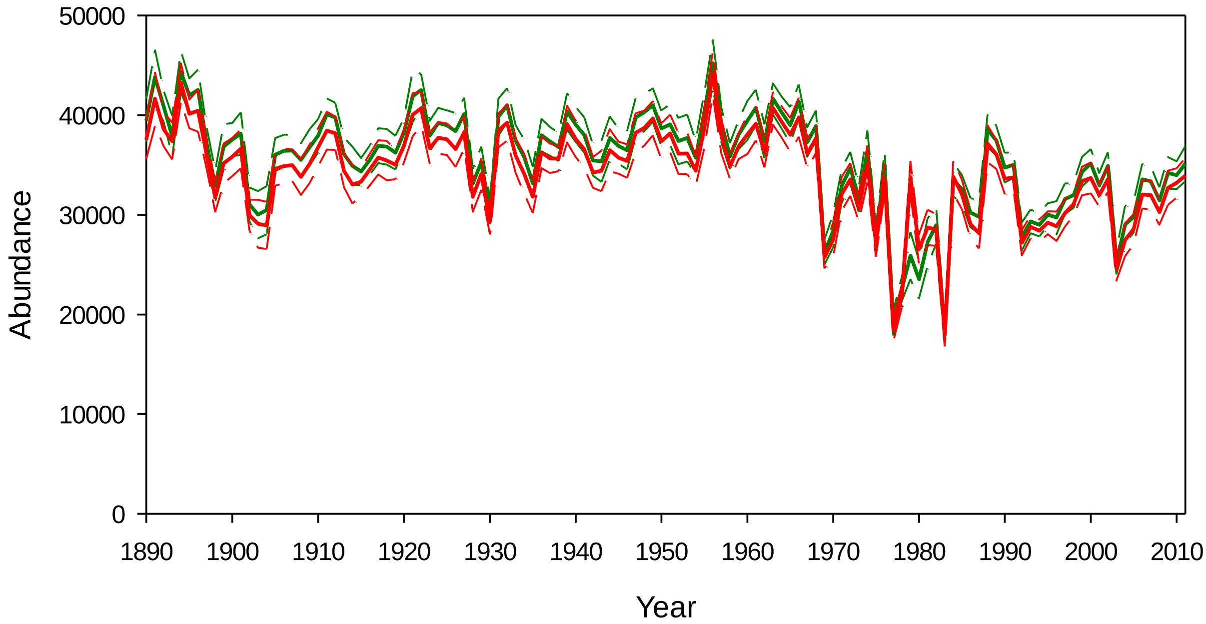

Figure 26: Yellowbelly population abundance (adults) in the Nebine catchment for the simulation period 1890–2011 [pre-development (green), full entitlement (red) ± std dev (dashed lines)].

Table 39: *Macquaria ambigua*–average population abundance and percentage change between development scenarios

Environmental assessment node	Population abundance[1]	
	Pre-development	Full entitlement[2]
Nebine catchment combined[3]	15 875	12 410 (–21.8%)
Wallam Creek at Cardiff (422501A)	7543	4284 (–43.2%)
Nebine Creek at Roseleigh Crossing (422502A)	4780	4778 (0%)

[1] average abundance of adults in the population;

[2] number in brackets indicate % change from pre-development

[3] population in the Nebine catchment within Queensland, which is a subset of the total modelled meta-population

Migratory fish species

The opportunities for migratory fish movement were modelled against three ToC risk categories at one node in the Nebine catchment (422502A).

There was no change in the number and duration of connection events for migratory fish under the full entitlement scenario compared to pre-development (Table 40).

Table 40: Number and duration of longitudinal connectivity events for migratory fish species over the simulation period

Environmental assessment node	Number and duration of connection events (days)	
	Pre-development	Full entitlement*
Nebine Creek at Roseleigh Crossing (422502A)	1057 events 13968 days	1057 events 13968 days (0%)

* number in brackets indicate % change of connection duration (days) from pre-development

Eastern snake-necked turtle (*Chelodina longicollis*)

The viability of *C. longicollis* populations was modelled at one environmental assessment node in the Nebine catchment.

There was no change in either the number or the duration of high stress periods due to the full entitlement scenario (Table 41). Although there were long periods of the simulation where the four year flood inundation return frequency ToC was exceeded at all assessment nodes under both flow scenarios, there was no increase in risk due to the full entitlement scenario compared with pre-development (Figure 27). These results suggest that the assessment node is marginal habitat for this species with high stress periods exceeding 30 years in duration under pre-development hydrology.

Table 41: High stress periods for the eastern snake-necked turtle

Environmental assessment node	Number of high stress periods	Total duration (days)	Longest spell (years) exceeding the ToC
Nebine Creek at Roseleigh Crossing (422502A)	14	34287	31.3

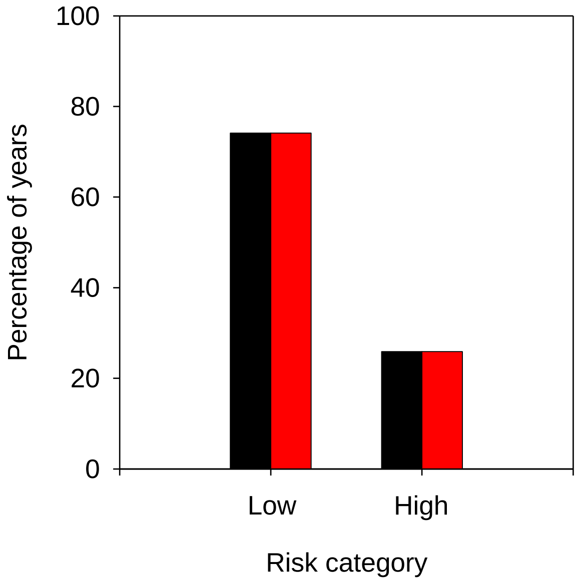

Figure 27: Risk profiles for *Chelodina longicollis*, as the percentage of years in the simulation period in each risk category (pre-development = black, full entitlement = red).

Absence of exotic fish species

Recruitment opportunities for *C. carpio* were modelled at one environmental assessment node in the Nebine catchment, Nebine Creek at Roseleigh Crossing (422502A).

There was no change in the percentage of years in the simulation period where strong recruitment was modelled to occur due to the full entitlement scenario (Table 42).

Table 42: Percentage of years and total number of days in the simulation with *C. carpio* spawning and recruitment opportunities

Environmental assessment node	Percentage of years in the simulation with spawning and recruitment opportunities	
	Pre-development	Full entitlement
Nebine Creek at Roseleigh Crossing (422502A)	28.7 (3483 days)	28.7 (3483 days)

Floodplain vegetation and wetlands

The frequency of floodplain vegetation and wetland inundation was modelled at one environmental assessment node in the Nebine catchment, Nebine Creek at Roseleigh Crossing (422502A) (Figure 25). Floodplain vegetation and wetland inundation frequencies under the full entitlement scenario were unchanged from pre-development for floodplain areas associated with this assessment node (Table 43, Figure 28).

Table 43: 422502A Floodplain vegetation and wetlands flooded at two inundation thresholds

Species and wetlands	2450 ML/day		7626 ML/day	
	Patches	Area (km^2)	Patches	Area (km^2)
Acacia stenophylla	0	0	1	51.58
Eucalyptus coolabah	0	0	1	51.58
Floodplain wetlands	2		0	

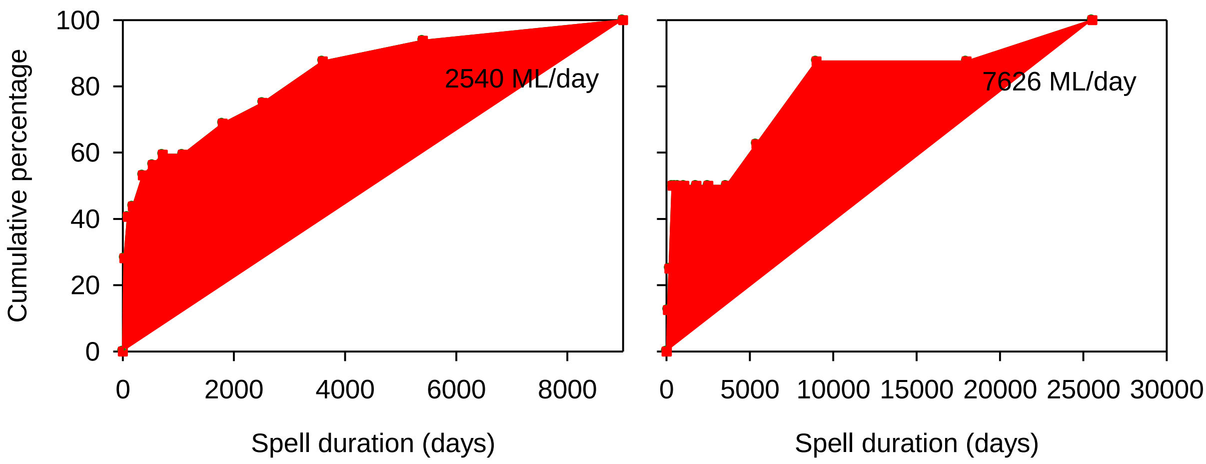

Figure 28: 422502A Floodplain area inundation duration plots (pre-development = green, full entitlement = red)

Waterholes as refugia

Waterhole isolation spells

Waterholes as refugia were assessed at one node in the Nebine catchment, Nebine Creek at Roseleigh Crossing (422305A) and there was no difference in the total number of spells, spell duration frequency distributions or the maximum spell duration between the scenarios (Table 44), indicating no increased hazard from full development.

Table 44: Number of no-flow spells over the simulation period

Environmental assessment node	Number and duration of spells (days)	
	Pre-development	Full entitlement*
Nebine Creek at Roseleigh Crossing (422305A)	1046 spells 331 days	1046 spells 331 days (0%)

* number in brackets indicate % change of maximum spell duration from pre-development

Waterhole pumping

There was one waterhole subject to licensed extraction of a total of 14 identified in the Nebine catchment (Table 45; Appendix 3). Under pre-development conditions 62% of the distances between nearest neighbour waterholes were greater than 20 km, representing a natural limitation to fish dispersal in some reaches. This was not changed under the full entitlement scenario, meaning that it posed no additional hazard to migratory species.

Table 45: Spatial distribution of refugial waterholes in the Warrego catchment in relation to the 20 km fish dispersal threshold, for pre-development and full entitlement scenarios.

No. of waterholes (No. with extraction)	Gaps > 20 km under pre-development		Gaps > 20 km under full entitlement		Reaches where hazard increased	
	Number	%	Number	%	Number	%
14 (1)	13	62	13	62	0	0

Knowledge gaps

In undertaking the ecological risk assessment, a number of knowledge gaps were identified. To improve future assessments and reduce the uncertainty in subsequent water allocation decisions, these knowledge gaps may inform the development and prioritisation of research and monitoring activities over the life of the new plan. Addressing the identified knowledge gaps will also fulfil outcome 9i in the WRP, which relates to improved understanding of matters affecting riverine ecological health.

Ecological asset selection

In stage one of the assessment process, 95 potential surface water ecological assets were identified from the ecosystem components and processes in the plan area. For all but nine of these, insufficient information was available in the published literature or from other research data to quantitatively characterise their critical dependencies in relation to the flow regime, meaning they could not be used in the assessment. For example, a diverse array of waterbirds is known to utilise inundated floodplain wetlands in the plan area. They are frequently cited as an important ecological value, however the magnitude of flows required to inundate wetland habitats, the period of inundation, the interaction between flow and habitat quality (e.g. vegetation for nesting, food availability), the scale of bird populations and the spatial patterns of habitat use cannot be adequately determined based on the current understanding, therefore eco-hydraulic rules linking aspects of the flow regime to waterbird population viability could not be developed.

Additionally, ecological outcome 9g relating to the maintenance of water quality levels could only be indirectly assessed (as a value supported by other ecological assets such as waterholes). Water quality could not be considered as an asset for the purposes of this assessment because quantitative relationships between hydrology and water quality are not well understood.

Further information about the critical aspects of the flow regime required by potential ecological assets would enable a more robust assessment utilising a wider variety of ecosystem components of the plan area.

Improved knowledge of eco-hydraulic requirements

While the nine prioritised assets did have sufficient information to be used in the quantitative risk assessment, in most cases knowledge gaps limited confidence in the results. In some instances, such as migratory fish, one well-studied species, Yellowbelly, was used to represent a number of species that exhibit similar behaviour, even though subtle differences in the flow requirements are likely to exist between species. For five assets (absence of exotic fish, floodplain wetlands, floodplain vegetation, refugial waterholes, and fluvial geomorphology and river forming processes), there was insufficient information about the consequence of changes from the pre-development flow regime to derive a ToC. In all cases, little information specific to the plan area was available, meaning knowledge from other regions had to be used to set eco-hydraulic rules and ToCs. The knowledge gaps identified and the types of information required to address them in order to improve future assessments are summarised in Table 46.

Table 46: Knowledge gaps identified from the WPBN WRP ecological assessment process

Ecological asset	Knowledge gap	Information required	Comments
Unassessed potential ecological assets	Information to set eco-hydraulic rules and ToCs representing the critical aspects of the flow regime required to maintain values represented by these potential assets.	The magnitude, duration and return interval of hydrologic conditions provided by the flow regime required for maintenance or reproduction and recruitment.	A number of high value species occupy regions within the plan area. Priority should be given to identify those species of highest value, highest dependency and restricted distribution. Basic information on how the life history of these species interacts with aspects of the flow regime should be collated throughout the life of the plan to ensure subsequent assessments can incorporate their eco-hydraulic requirements. Water quality is one of the WRP outcomes, but clear information about the effect of hydrology and other relevant environmental parameters is required in order to use water quality as an asset for risk assessment.
	Supporting habitat information for assessment	e.g. scale of populations and the distribution of habitats, specific wetlands/waterholes of importance, bathymetry and geomorphology, river channel cross-sections.	As above
Flow spawning fish species	Yellowbelly movement behaviour in the plan area	Rate, distance and flow cues of Yellowbelly movement in the plan area.	Yellowbelly is a high value ecological asset within the plan area, with recruitment behaviours linked to the flow regime. A number of knowledge gaps are highlighted here to improve the reliability of future modelling outputs. This is confirmed by the model sensitivity analysis, which identified migration parameters as the most influential factors for minimum population sizes.
	Contribution of juvenile recruits from floodplain and terminal wetlands to riverine channel populations	Patterns of juvenile migration between and within riverine channels and off stream and terminal wetlands.	The role of floodplains as juvenile habitat and as source of lateral nutrient transfer has high priority for basins with high levels of flood harvesting-while

		Quantify the hydrological requirements for these juvenile migrations to riverine channel populations. Measures of juvenile recruitment to the adult population. Information on the threats to these processes from water resource development.	the Warrego for instance would be a relatively natural system to research these processes in. If development is restricting movement between lower and middle reaches this could reduce the abundance of Yellowbelly in the Warrego (or other basins). Understanding of movement behaviour has a high priority as shown by model sensitivity analysis.
	Yellowbelly fecundity	Age-dependent recruitment rates	The recruitment rates were estimated for the meta-population model. While workable, this can be improved upon with actual observations of age-dependent recruitment. Such parameters are likely to have a big influence on realistic modelling of recovery from long dry spells.
	Yellowbelly utilisation of dam and weir pools	Knowledge of preferred habitat (edges, riparian qualities, and depths) which permit the estimation of carrying capacities of “artificial waterholes” such as dam and weir pools	Current estimates of habitat provided by weir pools may over represent their carrying capacity due to anoxic hypolimnetic water, or other characteristics. Most basins have more dams and weirs than the western basins; therefore this knowledge will have high priority for more developed basins.
	Persistence time of refugial waterholes in the plan area	Bathymetry of waterholes, rates and mechanisms of water loss during no-flow spells.	Refugial waterholes provide critical habitat to a range of biota during no-flow periods. In ephemeral systems such as the Warrego, Paroo, Bulloo and Nebine catchments, alteration to the provision of this refugial habitat (both in terms of overall persistence, and spatial patterns of provision) can have major consequence for many species. Therefore basic information on the distribution and bathymetric features of these waterholes and an inventory of threats to their persistence is a high priority knowledge gap to be addressed throughout the life of the plan.
	Minimum population viability threshold for Yellowbelly	Measures such as effective population size; information about the scale and connectivity of sub-	Important information to inform the derivation of thresholds of concern and the assessment of risk to population viability. This information determines

		populations in the northern Murray-Darling Basin.	minimum population thresholds for Yellowbelly and other species in the flow spawning guild against which to assess water allocation scenarios.
	Differences in flow requirements between Yellowbelly and other species in the flow spawning fish guild	Specifics of eco-hydraulic rules for use as model parameters e.g. duration, season and magnitude of flow cues, dispersal capacity, age distribution.	A number of assumptions were made in the risk modelling about how applicable the eco-hydraulic requirements of Yellowbelly are to other flow spawning fish species with similar requirements. These assumptions were based on scientific publications of findings from other catchments. It is a priority to validate these assumptions over the life of the plan to ensure subsequent assessment processes have improved certainty.
	Effect of changes in fish abundance on recreational fishing catchability	Relationship between target fish population size and catch success; interaction effect of increased exotic fish populations; ToC for recreational fishing value.	Currently the ToC for Yellowbelly is based on a minimum population to maintain long term viability. It has been recognised that recreational fishing is an important value associated with this species. Although there is likely to be a relationship between population abundance and catchability, there are also likely to be confounding factors such as abundance of European carp. These relationships can be further investigated throughout the life of the plan via structured stakeholder elicitation.
Migratory fish species	Fish movement behaviour in the plan area	Rate, distance and flow cues for fish movement in the plan area.	See comments on refugial waterholes
	Location and nature of barriers in the plan area	Location, size, drown-out flow threshold and the presence of fishways.	As above
Eastern snake-necked turtle	Accurate flow-habitat relationships for the plan area	Flow thresholds for inundation of floodplain wetlands from river-floodplain.	A number of ecological assets have either direct or indirect requirements relating to periodic floodplain inundation. Reliable digital elevation models (DEMs) linked to gauges; persistence time of floodplain wetlands (based on bathymetry, rates and sources of water loss) are critical knowledge requirements needed to progress future quantitative comparative analyses of water management scenarios.

	Nature and effect of "high stress" periods in the plan area	Maximum terrestrial aestivation time in the plan area; rate of mortality during post-aestivation migration; reproductive lag following aestivation and confinement to waterhole habitats.	The current ToC for the eastern snake-necked turtle is based on observations from eastern New South Wales. The assumptions which underpin this ToC should be tested on populations from the plan area.
Absence of exotic fish species	Flow requirements of carp at breeding sites	Locations, flow thresholds for inundation, persistence time and connectivity to the main channel of wetlands that act as breeding locations for carp.	European carp occupy all catchments in the plan area with the exception of the Bulloo. In these catchments, it is frequently the most dominant fish species as measured by biomass. Although not an ecological asset in its own right, the presence of this species represents a significant threat to native aquatic biota. Altered patterns of floodplain inundation and lateral and longitudinal connectivity due to water management have the potential to affect the recruitment success of this species. Basic information on the eco-hydraulic requirements of this species is required to better inform modelling to assess alternate water management scenarios and their impact on carp recruitment and dispersal. Additionally, information on the interaction between carp and native aquatic biota will provide the necessary context to fully understand the impact of this species.
	Carp movement behaviours in the plan area	Rate, distance and flow cues of carp movement in the plan area.	
	Carp mortality rates under different flow conditions	The rate of mortality of juvenile carp following a breeding opportunity; the relationship between time since flow and juvenile survival; the effect of flow pulses on carp recruitment and dispersal.	
	The flow requirements and impact of other exotic fish e.g. goldfish, *Gambusia*	Eco-hydraulic rules for breeding and dispersal; effects on habitats and native species.	As above
Floodplain vegetation	Accurate flow-habitat relationships for the plan area	Accurate flow thresholds for inundation of floodplain vegetation patches derived from river-floodplain DEMs linked to gauges	A number of ecological assets have either direct or indirect requirements relating to periodic floodplain inundation. Reliable digital elevation models (DEMs) linked to gauges; persistence time of floodplain wetlands (based on bathymetry, rates and sources of water loss) are critical knowledge requirements needed to progress future quantitative comparative analyses of water management scenarios.

	Watering requirements, especially groundwater vs. flood dependence	The relationship between groundwater, shallow aquifers, floodwater, soil moisture and vegetation condition.	The water regime supporting floodplain vegetation comprises soil moisture derived from local rainfall and floodplain inundation, and shallow groundwater sources. The role of floodplain inundation in recharging soil moisture and shallow aquifers is currently poorly understood. Modelling suggests that floodplain vegetation is persisting longer than our current understanding predicts based on water supplied by floods alone, even under pre-development conditions. Investigations are required to: (i) develop a water balance conceptualisation for floodplain systems incorporating all surface and groundwater components, (ii) identify the major water use pathways utilised by floodplain vegetation, (iii) identify water regimes which support vegetation recruitment (flowering, germination, and seedling establishment) and growth, and (iv) identify the threats to these regimes from water management.
	Flow requirements for vegetation recruitment (as opposed to maintenance)	The magnitude, timing and sequence of flow events required to trigger flowering and germination and ensure survival of seedlings and saplings to recruitment stage.	
	Role of floodplain production in supporting riverine food webs	Sources of carbon and fatty acids in aquatic food webs; relationship between flood size and frequency and the quality of food resources in aquatic systems.	The ecological values associated with floodplain vegetation include their contribution to aquatic food webs. However the value of this contribution remains unquantified.
Floodplain wetlands	Accurate flow thresholds for floodplain wetland inundation in the plan area	Accurate flow thresholds for inundation of floodplain wetlands from river-floodplain DEMs linked to gauges; historical wetting and drying patterns by aging and analysis of sediment cores.	A number of ecological assets have either direct or indirect requirements relating to periodic floodplain inundation. Reliable digital elevation models (DEMs) linked to gauges; persistence time of floodplain wetlands (based on bathymetry, rates and sources of water loss) are critical knowledge requirements needed to progress future quantitative comparative analyses of water management scenarios.
	Floodplain wetland persistence time in the plan area	Bathymetry of wetlands, rate and sources of water loss during no-flow spells.	
	The attributes of wetlands that support iconic species such as waterbirds and how these are influenced by flow	e.g. the role of flow in determining the richness and quality of food sources, the growth of nesting vegetation, water quality and temperature in wetlands.	See unassessed potential ecological assets.

Waterholes as refugia	Waterhole persistence in the plan area and the effect of waterhole pumping (i.e. allocations with a nil passing flow condition)	Bathymetry of waterholes; rate and mechanisms of water loss during no-flow spells; pumping rates; location and rate of stock and domestic pumping.	A number of assumptions were made in the assessment in relation to the persistence of refugial waterholes. To ensure a precautionary approach was used in the assessment, in all cases, a worst case persistence was adopted which assumed waterholes were lost from the system when stock and domestic pumping commenced. Given the critical importance of refugial waterholes in maintaining biota in these ephemeral systems during no-flow periods, a network understanding of waterhole locations, and persistence values and the location and nature of potential threats would greatly enhance future risk modelling and improve the capacity to evaluation alternate water management options.
	Depth thresholds for waterhole habitat quality	Relationships between waterhole depth and: water quality (e.g. DO), water temperature, primary production, food availability and quality, microhabitat availability, inter- and intra-specific competition and predation, disease transmission etc.	The current assessment assumes that waterholes remain suitable habitat until such time as it is dry. However as waterholes dry, changes in habitat quantity and quality occur. Information on the relationships between the depth of waterholes and their suitability to support dependent biota is required to set minimum depth thresholds required to maintain their refugial function.
Fluvial geomorphology and river forming processes	Rates of net sediment accumulation	Current sediment load based on sediment probing; historical/long-term rate of accumulation based on sediment coring and dating.	Proximal landuse activities may alter sediment transfer to streams. Deposited sediment loads have the potential to decrease waterhole depth, thus reducing their persistence over time. Although water management does not directly influence landuse practices, alteration to the river forming flows–particularly reduction in high flow events, may exacerbate this issue. Government monitoring programs such as SEAP should continue to investigate confounding stressors such as waterhole sedimentation to provide context to EFAP and the ongoing

			assessment of water management activities.
	Accurate bankfull flow levels for gauges in the plan area	River channel cross-sectional data for reaches around gauging stations	Basic cross-sectional information at gauges and important habitat areas will aid in the calculation of river forming events and their influence on sediment mobilisation.
	Sediment scouring properties of different flow events	The rate of sediment scouring at bankfull flow; the rate of sediment scouring from larger floods; the effect of in-channel and floodplain infrastructure; the scouring properties of different sediment types.	As above. Sediment profiling and cross-sectional information are essential information to inform the modelling of river forming processes across the state.
Unique genetic diversity of aquatic plants and animals within the Bulloo basin	Which elements of the biota are genetically unique	Phylogeography of aquatic biota in the Bulloo and surrounding regions.	Application of modern molecular methods to the determination of genetic structure of populations is now a routine component of many monitoring programs. It is recommended that opportunities be explored to add these genetic markers to existing EFAP activities utilising traditional population demographic methods.

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Appendix 1

Yellowbelly meta-population modelling parameters and data sources

Table 47: Population parameters as used in RAMAS Yellowbelly model. For additional background see Nicol & Todd (2004) and other contributions in Akçakaya et al. (2004).

Parameter	Observed value	Source	Model parameter input	Model effective parameter value*
Carrying capacity	0.052 Golden Perch > 2 yrs per m^2 of isolated waterhole; 0.021 adult Golden Perch for all electrofishing runs in Warrego and Condamine-Balonne	Depletion sampling of waterhole in Moonie River; single-pass electrofishing of Golden Perch in western rivers	Time series of waterhole surface area multiplied by observed density from depletion sampling	
Density dependence	Adults compete for multiple resources, with emphasis on deep pools (which contract into shrinking waterholes with diminishing food supply, consequently causing continued competition for habitat)	Pusey et al 2004	Ceiling density dependence, affecting survival rates for adults (3+ yrs)	
Sexual maturity	Between 3 and 4 (3 for males, 4 for females)	Pusey et al 2004	4 years	
Age Structure	Up to 26 years	Pusey et al 2004		11 (all fish older than 11 years are subjected to the 11 year age group parameters)
Recruitment	Recruitment rate is variable with flood levels; effective recruitment per adult 3+yrs	Warrego & Condamine-Balonne LTMP, 7 year survey data)	Threshold model overriding constant recruitment rate per adult	Regression equation for override trend; (effective) rates per year

Parameter	Observed value	Source	Model parameter input	Model effective parameter value*
Recruitment rate	Average observed recruitment rate is 1.34 per adult (however this is an incomplete observation, with high bias against detection of recruits)	LTMP 7 year survey data (Hagedoorn & Smallwood 2007)		Yr Recruits 4–5 3 6–11 4
Survival rate	Observed survival rate: year n / survival to year n+1 1 0.48 2 0.54 3 0.54 4 0.63 5 0.63 6 0.67 7 0.67 8 0.67 9 0.67	LTMP 7 year survey data (Hagedoorn & Smallwood 2007)	Parameter survival rate: year n / survival to year n+1 1 0.46 2 0.53 3 0.56 4 0.61 5 0.65 6 0.69 7 0.67 8 0.67 9 0.63	Effective survival rate: year n / survival to year n+1 1 0.46 2 0.53 3 0.56 4 0.61 5 0.65 6 0.69 7 0.67 8 0.67 9 0.63
Max. ann. adult growth rate R (effective recruitment)	(observations too limited due to high observation uncertainty)	(Nicol & Todd 2004)	1.2 (“doubling in 4 years”)	
Stochasticity	literature	(Nicol & Todd 2004)		Standard deviations of survival and recruitment matrix Uncertainty of population counts Uncertainty of dispersal Fecundity, survival &

Parameter	Observed value	Source	Model parameter input	Model effective parameter value*
				carrying capacity uncorrelated
Standard deviations of survival and recruitment matrix		(Nicol & Todd 2004)	Input matrix:	Effective (model-calculated) matrix: see below
Uncertainty of dispersal	cv for dispersal 0.2	(Nicol & Todd 2004)	Coefficient of variation (standard deviation / mean) of dispersing individuals is 0.2	
Fecundity, survival & carrying capacity			Set to uncorrelated	
Correlation distance function (Correlation between populations)		Literature values (Akçakaya et al. 2004)		a=1, b=200,c=1.5 in C_{ij} = a * exp(-D_{ij}^c/b); C_{ij} is correlation matrix, D_{ij} is distance matrix
Dispersal function	calculate dispersal matrix with parameters: a = 0.1, b = 20, c = 1, d = 200	Moonie dispersal	a = 0.1 b = 20 c = 1 Dmax=200	M_{ij} = a * exp(-D_{ij}^c/b) where a = 0.264 (effective figure) b = 20 c = 1 If distance > Dmax = 200 then M_{ij} = 0; M_{ij} = migration between nodes
Dispersal participation	33 % of population	Moonie fieldwork (DERM unpublished data)		0.33

Parameter	Observed value	Source	Model parameter input	Model effective parameter value*
Carrying capacity	0.052 Golden Perch > 2 yrs/m^2 of isolated waterhole; 0.021 adult Golden Perch for all electrofishing runs in Warrego and Condamine-Balonne	Depletion sampling of waterhole in Moonie River; single-pass electrofishing of Golden Perch in western rivers	Time series of waterhole surface area multiplied by observed density from depletion sampling	

* The calculation sequence overrode many initial parameter values, so the effective parameters are most like the actually observed values, i.e. the values the model had to be fitted to.

Table 48: Information and data sources utilised in the meta-population model

Information	Source
Channel connections between streams	GoogleEarth images (CNES SPOT and DigitalGlobe high resolution images), generally based both on stream channel geomorphology and location of major waterholes
Waterhole habitat	Location from Silcock (unpublished data) or GoogleEarth imagery in association with stream channels; Surface area from GoogleEarth imagery; Depths from SEAP or estimated according to comparison of geomorphological cues; General geometric rules of waterhole shape from Moonie River waterhole bathymetry (DERM unpulished);
Model time step	Temporal resolution of 1 year, modelled over 122 years (the availability of stream flow simulation data) plus a 15 year “burnin” period to provide stable model behaviour. Start date for a new year was set to October 1st.
Spatial model scale	Based on the assumption that distances of up to 20 km do not present a challenge for Yellowbelly migration, the sub-populations were aggregated and modelled at approximately 20 km resolution, and then assessed at the lower resolution coinciding with IQQM stream flow data nodes for changes to average density, or at river basin scale for population viability
Spatial model configuration	Used knowledge of stream flow connectivity, via channels, and as quantified by flow thresholds (Green & King 1993), to identify functionally related sub-populations, and define meta-population extent. For reporting purposes the assessment area was restricted to basin extents limited to

	Queensland. The generally accepted relationship between the basin-wide sub-populations in the Warrego, Paroo, Bulloo and Nebine basins and the overall Murray Darling basin population was not considered here.
Floodplain features included	Lateral floodplain features were not included, but terminal features including Lake Wombah and Yantabulla were, based on floodplain connectivity quantified for the Warrego (Green & King 1993), and for the Paroo (Timms 2008).
Connectivity	Connectivity was presented as a discrete 0/1 override in a connectivity matrix, and was based on the identification of the no-flow periods in the IQQM flow data. If no flow occurred in a biological year starting October 1st, no dispersal occurred in the model for that year. Connectivity between the lower Paroo and Warrego was based on the floodplain inundation thresholds linked to terminal lagoon inundation (see “Floodplain features”). Connectivity between the Nebine channels and the Culgoa was based on expert opinion (Khan, pers. comm.)
Interpretation of long term monitoring (LTMP) data for survival rates and annual recruitment	Using a length-age relationship from the Border Rivers (DERM unpublished data), fish lengths from the LTMP data were converted to ages, and the fish abundances were converted back to spawning dates.
Sensitivity analysis	Covers all model parameters (but not the steps leading up to preparing environmental data series for the models) – approach follows Curtis & Naujokaitis-Lewis (2008) with minor modifications.

Appendix 2

Regional Ecosystems (REs) dominated by each of the ecological asset vegetation species in the plan area

RE type	*Acacia stenophylla*	*Eucalyptus camaldulensis*	*Eucalyptus coolabah*	*Eucalyptus largiflorens*	*Eucalyptus ochrophloia*	*Meuhenbeckia florulenta*
1.3.7		+				
1.3.8		+				
1.3.9		+				
10.3.13		+	+			
10.3.14		+	+			
10.3.15		+	+			
10.3.23						
11.3.15	+		+			+
11.3.16				+		+
11.3.2		+				
11.3.25		+	+			
11.3.27		+	+			
11.3.28			+			
11.3.3			+			
11.3.37		+	+			
11.3.4		+	+			
13.3.5		+				
2.3.13	+					
2.3.14						+
2.3.17			+			
2.3.25		+				
2.3.26		+				
2.3.34		+				
4.3.1		+	+			
4.3.11		+	+			
4.3.2		+	+			
4.3.3		+	+			
4.3.4		+	+			
4.3.5		+	+			
4.3.6		+				
5.3.1		+	+			
5.3.13						+
5.3.18						+
5.3.2		+	+			
5.3.20		+	+			

RE type	*Acacia stenophylla*	*Eucalyptus camaldulensis*	*Eucalyptus coolabah*	*Eucalyptus largiflorens*	*Eucalyptus ochrophloia*	*Meuhenbeckia florulenta*
5.3.3		+				
5.3.4		+				
5.3.5			+			+
5.3.6			+			+
5.3.7			+			+
5.3.8	+		+			+
6.3.1		+				
6.3.11						+
6.3.12			+			
6.3.2	+	+	+			
6.3.24			+			
6.3.3	+	+	+			
6.3.4					+	
6.3.5			+		+	
6.3.7	+		+			
6.3.8				+		+
6.3.9			+			
9.3.1		+				
9.3.13		+				
9.3.17		+				
9.3.18			+			
9.3.19			+			
9.3.4		+				
9.3.7		+				

Appendix 3

Waterhole pumping spatial analysis methods

Unsupplemented water allocations with a nil passing flow condition (i.e. waterhole pumping licenses) were collated as a spreadsheet containing the Authority Reference number and associated positioning coordinates (Latitude and Longitude) for the Warrego, Paroo, Bulloo and Nebine subcatchment, current as at 18 July 2012.

The location and persistence character of refugial waterholes for the Murray Darling Basin were represented by a dataset (DERM, unpublished data) compiled using the methods detailed in Silcock (2009). Silcock (2009) mapped waterholes from wetland mapping produced by the Queensland Wetlands Mapping Program. The wetland mapping was compiled from time-series Landsat satellite imagery, Geodata and regional ecosystem mapping of water body features. Further details of the methodology and limitations of the dataset are detailed in EPA (2005a, b).

River and stream systems were represented by the Geofabric Surface Network in the Australian Hydrological Geospatial Fabric (Geofabric) Version 2.0 (BOM, 2011). The Geofabric covers the Australian Continent at a scale of 1:250 000. The Geofabric Surface Network was based on topologically connected flow direction streamlines, known as ANUDEM Streams V1.1.2 (ANUDEM Streams), as developed by the ANU using ANUDEM. The ANUDEM Streams V1.1.2 were cross-referenced and additionally informed by AusHydro V1.7.2 vector streamlines derived from GA GEODATA TOPO 250K Series 1 (GEODATA 1) and GEODATA TOPO 250 K Series 3 (GEODATA 3) (BOM, 2011). The Geofabric is currently considered the authoritative source data for hydrological geospatial entities.

Refugial waterhole and 250k Geofabric Surface Network data were extracted for the Warrego, Paroo, Bulloo and Nebine catchments and the nearest neighbour distances (upstream and downstream distances along the river channel to the next waterhole) calculated using an OD Cost Matrix in the Network Analyst extension of ArcGIS 9.3.1.

Refuge waterholes with a coinciding or water allocation were identified by overlaying the point location of unsupplemented water allocations with the refugial waterholes mapped by Silcock (2009). Nearest neighbour measures were then repeated with the waterholes linked to unsupplemented water allocations removed, to determine the effects of the full entitlement case on the spatial distribution of waterholes. Measurements were based on the distance between central points located inside the extent of polygon waterhole representations closest to the centroid of the polygon.

Some individual waterholes in the Warrego catchment had multiple water allocations. In these instances, as distances were calculated between waterhole centroids, those allocations will have the same nearest neighbour upstream and downstream distance results although the allocations may occur at different locations along the waterhole. The downstream distance could not be calculated for two allocations (27 AP13211 in the Warrego catchment and 31 AP13211 in the Paroo catchment) where no further waterholes were located downstream in the Silcock dataset within the catchment boundary.

Acoustic telemetry data from the Moonie River in the Upper Murray-Darling Basin (DERM 2010b; DSITIA, unpublished data) was analysed to determine the dispersal capacity of Yellowbelly, which identified 20 km as a fish dispersal threshold. Distances between permanent waterholes under pre-development and full-entitlement scenario (i.e. waterholes subject to extraction removed from the distribution), were then compared to the fish dispersal threshold to examine the effect of water resource development (Figure 29).

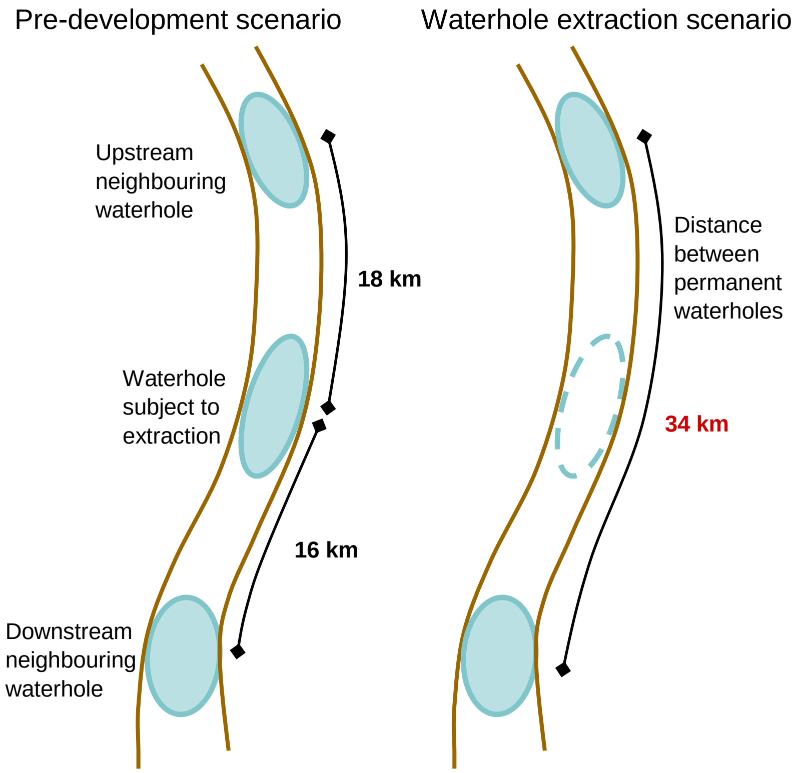

Figure 29: Approach for assessing the effect of extraction on waterhole refuge distribution

Appendix 4

Waterhole siltation in the plan area

A landholder survey confirmed that siltation of waterholes is recognised as a significant threatening process to waterhole persistence during no-flow spells in the plan area, and also that scouring by large floods can remedy this hazard (Silcock 2009). The following comments are from Silcock's (2009) database of permanent waterholes in the plan area and pertain to particular waterholes.

Warrego catchment
'waterholes are sanding-up'
'used to be a good permanent hole, but has silted up and only lasts 12 months- dries a few times a decade now'
'still permanent, but gets very low – has sanded up'
'has silted up – used to be able to jump off the bridge, now only a few feet of water in it, only a matter of time before it goes dry'
'still permanent, but is silting up now and now only a few feet deep
'prior to 1990 flood went dry every 2-4 years, but 1990 flood scoured it and is now more permanent and has not dried since'
'has probably silted up'
'used to be 17 foot deep, but silted up substantially and now only 8-9 foot deep'

Paroo catchment
'used to be more permanent but has silted up and now goes dry quite often during droughts'
'has silted up, was probably more permanent'
Not particularly deep – has silted up'

Bulloo catchment
'has silted up substantially'
'semi-permanent but silting up and only holds water for 6 months'
'has silted up and goes dry regularly'
'has silted up so may go dry in the future'

Nebine catchment
'permanence estimated pre-silting'
'more or less permanent, but may have silted up now'
'used to be permanent but has silted up now'

Appendix 5

Licensed waterhole pumping data

Waterhole pumping licenses (allocations with a nil passing flow condition) and the stream channel distance to the nearest neighbouring permanent waterholes

Licence Number	Details			Distance (m)		
	Waterhole	Catchment	Watercourse	Upstream	Downstream	Total
1 AP13211	27 Mile	Warrego	Warrego River	20659.13	15795.22	36454.34
2 AP13211	Charleville (North)	Warrego	Warrego River	6101.69	32246.48	38348.17
3 AP13211	Charleville (South)	Warrego	Warrego River	8654.77	30192.28	38847.05
4 AP13211	Charleville (South)	Warrego	Warrego River	8654.77	30192.28	38847.05
6 AP13211	Baker's Bend (North)	Warrego	Warrego River	13804.78	15583.50	29388.28
7 AP13211	Baker's Bend (North)	Warrego	Warrego River	13804.78	15583.50	29388.28
8 AP13211	Baker's Bend (North)	Warrego	Warrego River	13804.78	15583.50	29388.28
9 AP13211	Baker's Bend (North)	Warrego	Warrego River	13804.78	15583.50	29388.28
10 AP13211	Baker's Bend (North)	Warrego	Warrego River	13804.78	15583.50	29388.28
11 AP13211	Baker's Bend (South)	Warrego	Warrego River	17305.79	12082.50	29388.28
12 AP13211	Arapinta	Warrego	Warrego River	4241.10	5072.32	9313.42
13 AP13211	Dillalah/Barimornie	Warrego	Warrego River	2232.22	2159.82	4392.04
14 AP13211	Murweh	Warrego	Warrego River	1660.96	10238.72	11899.68
15 AP13211	Quilberry (South)	Warrego	Warrego River	13748.73	10393.01	24141.74
16 AP13211	Warrego R	Warrego	Warrego River	7650.79	6537.65	14188.44
17 AP13211	Claverton	Warrego	Warrego River	4479.54	3306.22	7785.77
18 AP13211	Coongoola	Warrego	Warrego River	1469.87	1902.88	3372.76
19 AP13211	Tickleman Garden Hole	Warrego	Warrego River	8216.95	2890.90	11107.85

20 AP13211	Cunnamulla	Warrego	Warrego River	17062.06	12573.15	29635.21
21 AP13211	Cunnamulla	Warrego	Warrego River	17062.06	12573.15	29635.21
22 AP13211	Ward	Warrego	Ward River	19851.77	13001.24	32853.01
23 AP13211	Ward	Warrego	Ward River	19851.77	13001.24	32853.01
24 AP13211	Ward	Warrego	Ward River	19851.77	13001.24	32853.01
25 AP13211	Ward	Warrego	Ward River	19851.77	13001.24	32853.01
76 AP13211	Baker's Bend (South)	Warrego	Warrego River	17305.79	12082.50	29388.28
26 AP13211	Tinnenburra	Warrego	Cuttaburra Creek	25318.51	9040.18	34358.69
27 AP13211	Kywong	Warrego	Cuttaburra Creek	11254.13	0.00	11254.13
30 AP13211	Eulo waterhole	Paroo	Paroo River	4688.91	9299.47	13988.38
31 AP13211	-[a]	Paroo	Paroo River	17389.99	0[b]	17389.99
28 AP13211	Wanko/Quilpie WH	Bulloo	Bulloo River	19282.66	33703.66	52986.31
29 AP13211	Adavale Waterholes	Bulloo	Blackwater Creek	10656.24	9068.56	72711.11
78 AP13211	Bollon	Nebine	Wallum Creek	59047.65	54361.69	113409.34

[a] No waterhole representation in Silcock’s dataset near the location provided for the water allocation. Water allocation location was used instead based on the assumption that an allocation would only be granted under the presence of a water source.

[b] No waterholes downstream of this location represented within the Silcock’s dataset within the WRP area of interest to calculate a nearest neighbour downstream distance.