7 How can the sustainability of irrigated agriculture be maximised?

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Chapter 7 examines the question ‘How can the sustainability of irrigated agriculture be maximised?’. It provides fundamental information about the risk of rise of watertable level, the effects of surface water drainage, and the likely ecological responses to altered flow regimes in the Flinders catchment. While there are many ecological changes that could occur as a result of irrigation development in the Flinders catchment, these are three key considerations. Key components and concepts are shown in Figure 7.1.

Figure 7.1 Schematic diagram of key components and concepts in the establishment of a greenfield irrigation development
7.1 Summary

The sustainability of agriculture in the Flinders catchment is in part related to the long-term impact of agricultural development on the natural environment. This chapter explores the scale of ecosystem response that would be expected based on the likely impacts of potential agricultural development.

The complexity of natural systems – and the many factors that impact on ecosystem response – mean that more detailed investigation would be required to inform specific developments. Many environmental changes may not be anticipated at the outset of a development, or could take many years to manifest, so these changes would require adaptive management and a thorough, well-documented understanding of baseline (pre-development) conditions.

7.1.1 RISK OF IRRIGATION-INDUCED SALINISATION

Rising groundwater can mobilise salts in the soils and substrata, bringing them close to the surface and discharging them into nearby rivers. The Flinders catchment has large areas of cracking clays with subsoils that are high in salt and susceptible to irrigation-induced secondary salinity. The alluvial soils, while smaller in extent, have lower salt levels. The watertable level depends on the initial depth to the watertable, recharge from rain and irrigation, the size of the irrigation area, management practices and distance to the river. The Assessment indicates that watertable levels under small neighbouring irrigation developments (less than 500 ha in area) are not likely to interact in the next 100 years if the developments are placed at least 1 km apart.

The watertable level is most likely to rise with high recharge rates and in soils with low saturated hydraulic conductivity. Proximity to rivers considerably reduces irrigation-induced rise in watertable level by increasing groundwater discharge. It may take many decades for watertable levels to respond fully to irrigation development, especially if the cultivated area is large or far from the river.

7.1.2 MANAGING IRRIGATION DRAINAGE

Surface drainage, or runoff, from irrigated land can arise from irrigation water itself and from rain falling on already wet soils. Transport of suspended sediments is unlikely to be significantly increased by continuous irrigated fodder production, but nitrogen and phosphorus accession to waterways could increase by up to 10% and 20%, respectively, depending on the area planted. For grain and other crops, nutrient and sediment loads depend largely on crop management.

Increases in phosphorus in runoff water are likely to affect downstream ecosystems, including Gulf of Carpentaria fisheries. Persistent waterholes in the Flinders catchment are critical refugia for biota during the dry season and most are naturally turbid. While vulnerable to changes in water quality, these ecosystems are adapted to high sediment loads.

7.1.3 ECOLOGICAL IMPLICATIONS OF ALTERED FLOW REGIMES

The responses of aquatic ecosystems to irrigation developments are varied and depend not just on the amount of water extracted, but also on the way in which water is extracted, stored and distributed through the landscape; the types of crops grown and irrigation systems used; the management systems in place; and local climate and environmental conditions.

Significant weed and water quality issues can arise from loss of riparian function following irrigation development. These impacts can be minimised through retention of riparian zones and appropriate and adaptive farm and riparian management.

Irrigation development changes the flow regime of a river system via extraction or diversion of water, instream barriers, and return of potentially contaminated irrigation tailwater. Natural flows in the Flinders catchment are low, or non-existent, during the dry season and waterholes become essential refugia for...
biota. They are vulnerable to changes in the dry-season flows that affect waterhole number, water volume and quality. Changes to flows can reduce the wet-season ‘first flush’ essential for refreshing waterholes after the dry season. Changes to flow regime also affect fish migration and recruitment as well as the delivery of nutrients to coastal waters, each of which is important in determining commercial and recreational fishing catches.

7.2 Risk of irrigation-induced salinisation

For salt to become an environmental problem there are three basic requirements: (i) a source of salt, (ii) a source of water in which to mobilise the salt, and (iii) mechanisms by which the salt is redistributed to locations in the landscape where it causes damage.

Soil and airborne electromagnetic data and analytical modelling results acquired as part of the Assessment highlight the importance of carefully selecting the location of irrigation development in the landscape and managing potential groundwater impacts.

Soil and airborne electromagnetic data indicate that the Flinders catchment has large areas of cracking clay soils, many of which are likely to be high in salt (particularly associated with the Rolling Downs Group) and susceptible to irrigation-induced salinity. There are also, however, large areas of recent alluvial and high level alluvial soils that pose a lower and potentially more localised salinity risk.

Analytical modelling results highlight that increased groundwater accessions from an irrigation development increases the watertable level beneath the development and also potentially increases groundwater discharge to the river system. If the watertable approaches within a couple of metres of the surface and there is a source of salt, irrigation-induced salinisation may occur. Watertable levels within irrigation developments can be managed using engineering approaches such as artificial drainage systems (Christen et al., 2003), used in conjunction with careful management (Hornbuckle et al., 2005). However, these systems are generally expensive and in most cases need a viable disposal method for drainage effluent (Ayars et al., 2006).

The results of this analysis show that irrigation developments close to rivers benefit from the river acting as a natural drainage point for the increase in groundwater accessions. This reduces the potential for land salinisation to occur. However, groundwater discharge to rivers can result in environmental problems. Unlike surface water drainage (Section 7.3) the flow of groundwater accessions to the river system is difficult to control through engineering approaches. In the Flinders catchment, groundwater discharge to the river system following irrigation development is likely to have elevated salt levels and greatest groundwater discharge is likely to occur in periods of low river flow, when the hydraulic gradient between the groundwater mound and the river system is highest. Careful consideration of the location of irrigation developments and likely environmental impacts will be needed to minimise potential non-beneficial impacts – i.e. changed flow regimes and saline discharge to the river system.

Controlling and minimising accessions to the groundwater system is critical and efforts to maximise the efficiency of irrigation systems to minimise groundwater accessions should be considered a key priority when developing new irrigation areas in the Flinders catchment.

Section 7.2 is structured as follows. An introduction to irrigation-induced salinisation is provided in Section 7.2.1. In Section 7.2.2 a new analytical modelling approach is used to evaluate the rise in watertable level and changes in groundwater discharge due to irrigation development. Section 7.2.3 uses this modelling approach to explore interactions of groundwater as a result of neighbouring irrigation developments. For irrigation-induced salinisation to occur there needs to be a source of salt. Section 7.2.4 discusses potential salt stores in the Flinders catchment.

This section presents generalised results. The risk of salinisation at a specific location in the Flinders catchment can only be properly assessed by undertaking detailed field investigation.
7.2.1 INTRODUCTION

Prior to European settlement, Australia was dotted with naturally-occurring brackish creeks, salt pans and salt marshes (Ghassemi et al., 1995). In these areas of ‘primary salinity’, ecosystems evolved that were adapted to the high concentrations of salt in the water and soil. Areas where the effects of salinity are now evident as a consequence of European settlement, are referred to as ‘secondary salinity’. Secondary salinity manifests itself in two main forms: that which occurs in irrigation regions and salinity occurring in dryland regions. The Assessment is concerned with irrigation-induced (secondary) salinity.

Three basic requirements for salt to become an environmental problem are: (i) a source of salt, (ii) a source of water in which to mobilise the salt, and (iii) mechanisms by which the salt is redistributed to locations in the landscape where it causes damage.

Rainfall contains small quantities of salt. Over many hundreds of years, salts from rainfall can become concentrated in the soil, through evaporation. Areas most susceptible typically have relatively low annual rainfall (i.e. less than 800 mm/year) and low soil permeability. An example in the Flinders catchment is the cracking clay soils formed on the Rolling Downs Group. Areas with higher rainfall (i.e. more than 1200 mm/year) and/or highly permeable soils tend to have lower concentrations of salts in the soil profile, because the salts are leached down to the watertable and flushed out of the groundwater system. Examples include the sand or loam over friable or earth clay and friable non-cracking clay or clay loam soils on the alluvial soils adjacent to the Flinders River. Salts can also be concentrated by in-situ weathering of rock and minerals in the soil.

In many irrigation developments around the world, poor-quality irrigation water is the source of salt in salinisation. In the Flinders catchment, however, the river water is relatively fresh (less than 500 EC), so is unlikely to be a source of salt. This is because the low levels of salt in the river water would be leached through the soil profile before they could accumulate in the root zone to levels that adversely affect crop development.

However, the increase in root zone drainage following applications of irrigation water can provide the source of water to mobilise soluble salts stored in the soil. Root zone drainage rates tend to be higher under coarser-textured soils (Petheram et al., 2002) and poor irrigation practices. In Australia, excessive root zone drainage through poor irrigation practices, together with leakage of water from irrigation distribution networks and drainage channels has caused watertable levels to rise under many intensive irrigated areas. Significant parts of all major intensive irrigation areas in Australia are currently either in a shallow watertable equilibrium condition or approaching it (Christen and Ayars, 2001). Where shallow watertables containing salts approach the land surface (in the vicinity of 2 to 3 m from the land surface), salts can concentrate in the root zone over time through evaporation. The process by which salts accumulate in the root zone is accelerated if the groundwater also has high salt concentrations. There are few groundwater data for the Flinders catchment, although it is known that groundwater quality is highly variable across the landscape (Chapter 3).

The extent to which watertable levels rise close to the surface depends on: (i) the initial depth to the watertable, (ii) the amount of recharge (originating from root zone drainage), (iii) the size of the irrigation area (thus dictating the total volume added to the landscape), (iv) the lateral distance to the river (which acts as a drainage boundary, thus reducing the height of the groundwater mound under irrigation), and (v) aquifer parameters, including the saturated hydraulic conductivity, aquifer thickness and specific yield. The hydraulic conductivity and specific yield are hydraulic properties of a soil’s ability to transmit water when submitted to a hydraulic gradient (e.g. difference in watertable level between two locations). The specific yield is the volume of water that could be allowed to drain from an aquifer under the forces of gravity and expressed as a proportion of the total volume of material in the aquifer.

In the Flinders catchment, there are few groundwater data and aquifer parameters typically need to be estimated from bore log information and generic relationships in the literature. The use of such relationships is made particularly challenging by the fact that saturated hydraulic conductivity is the most variable environmental parameter, its range varying by over 11 orders of magnitude. Typical values of saturated hydraulic conductivity and specific yield are provided in Table 7.1.
### Table 7.1 Typical values of specific yield and saturated hydraulic conductivity

<table>
<thead>
<tr>
<th>SOIL TEXTURE</th>
<th>SPECIFIC YIELD*</th>
<th>SATURATED HYDRAULIC CONDUCTIVITY** (m/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gravel</td>
<td>0.25</td>
<td>3 to 30,000</td>
</tr>
<tr>
<td>Sand</td>
<td>0.20</td>
<td>0.3 to 300</td>
</tr>
<tr>
<td>Silt</td>
<td>0.18</td>
<td>0.00003 to 3</td>
</tr>
<tr>
<td>Clay</td>
<td>0.02</td>
<td>0.00000003 to 0.00003</td>
</tr>
</tbody>
</table>

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

### 7.2.2 RISE IN WATERTABLE LEVEL AND CHANGES IN GROUNDWATER DISCHARGE DUE TO IRRIGATION DEVELOPMENT

A new analytical modelling approach (Jolly et al., 2013) was developed to evaluate the maximum (steady-state) rise in watertable level likely as a result of introducing new irrigation developments of varying areas situated at various distances from a river. A separate analysis was undertaken to investigate the time it takes the watertable to rise to its maximum level and how changes in groundwater discharge occur.

To investigate the sensitivity of the results to these parameters a range of likely $w^2$ values was selected (Table 7.2). Irrigation developments between 100 and 1000 ha in size are representative of irrigation developments on individual properties. Results of these farm-scale developments are presented in this section. Irrigation developments between 6,000 and 12,000 ha are representative of the size of scheme-scale irrigation developments and these results are presented in the case study analysis in chapters 8 to 10.

### Table 7.2 Likely range of values for parameters in the Flinders catchment

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>SYMBOL</th>
<th>UNIT</th>
<th>VALUES</th>
<th>COMMENT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Distance from centre of irrigation area to river</td>
<td>d</td>
<td>km</td>
<td>0.5, 1.0, 2.0, 5.0, 10.0</td>
<td>River assumed to be straight</td>
</tr>
<tr>
<td>Circular irrigation area</td>
<td>A</td>
<td>ha</td>
<td>100, 250, 500, 1000</td>
<td>For radii of 564, 892, 1262 and 1784 m</td>
</tr>
<tr>
<td>Recharge rate</td>
<td>R</td>
<td>mm/y</td>
<td>1, 10, 20, 50, 100, 200, 500</td>
<td>Recharge rate is related to the amount of water applied and the permeability of the soil. A recharge rate of 500 mm/y (or more) could occur under a ring tank.</td>
</tr>
<tr>
<td>Aquifer transmissivity (saturated hydraulic conductivity multiplied by aquifer thickness)</td>
<td>T</td>
<td>m$^2$/day</td>
<td>200, 500, 2000</td>
<td>Represents a constant saturated aquifer thickness ($h = 10$ m), and hydraulic conductivities ($K$) of 20, 50 and 200 m/day</td>
</tr>
<tr>
<td>Specific yield</td>
<td>Sy</td>
<td></td>
<td>0.10 to 0.20</td>
<td>Specific yield does not alter the maximum height of the watertable. It affects the time over which the watertable rise occurs.</td>
</tr>
</tbody>
</table>

### Maximum rise in watertable level

The maximum rise in watertable level increases with higher recharge rates and decreases with higher saturated hydraulic conductivity. Figure 7.2a shows that the effects of saturated hydraulic conductivity (and hence aquifer transmissivity) and recharge rates are linear but opposite and perfectly correlated. Hence, to simplify the presentation of results and reduce the number of variables, it is possible to report watertable level against recharge rate divided by the aquifer transmissivity.
Figure 7.2b shows the maximum watertable level expected for an irrigation area of 100 ha. This maximum level decreases as the distance to the river decreases. This is because, with the irrigation development located closer to the river, groundwater can be discharged to the river at a greater rate.

Figure 7.3 shows that the maximum watertable level increases in a non-linear manner as the distance to the river increases.

Figure 7.4a, Figure 7.4b and Figure 7.5a show that the maximum watertable level increases as the irrigation area increases, and also as the recharge rate increases. For the combination of parameters considered for the Flinders catchment, the highest point on the red line in Figure 7.5a shows the upper bound for watertable rise \( h_{\text{max}} = 41.8 - 10 = 31.8 \text{ m} \), where 10 m is the initial watertable level, which represents the largest irrigation area \( A = 1000 \text{ ha} \) located furthest from the river \( d \), with an aquifer having the lowest drainage capacity (highest R/T). Figure 7.5b presents the same results in a different way to highlight the effect of increasing the recharge area and distance to the river on watertable level.

**Figure 7.2** Steady-state watertable level for (a) various recharge rates and hydraulic conductivities \( K \) and (b) an irrigation area of 100 ha, at varying distances to the river

**Figure 7.3** Steady-state watertable level for an irrigation area of 1000 ha, plotted against distance to the river
Figure 7.4 Steady-state watertable level at varying distances to the river for an irrigation area of (a) 250 ha and (b) 500 ha

Figure 7.5 Steady-state watertable level at varying distances (d) to the river for (a) an irrigation area of 1000 ha and (b) various irrigation area and distance combinations

Changes in rise in watertable level over time

One of the challenges in managing groundwater is the time lag between a change in management and the response of the groundwater system. This analysis demonstrates how the watertable level rises over time until it achieves its maximum height. A key parameter for undertaking this analysis is the specific yield of the groundwater system. Figure 7.6 provides an example where the irrigation area is 100 ha and the recharge rate is 100 mm/year. For a given aquifer diffusivity (D) (i.e. aquifer transmissivity divided by specific yield), Figure 7.6 shows that after a change in recharge, the initial response of the groundwater system is identical regardless of the distance to the river (d). This is because the groundwater mound under an irrigation development forms before groundwater discharge to the river increases. When the groundwater mound reaches the river, the rate of the rise in watertable level starts to decline until the level reaches its maximum (i.e. under steady-state conditions). The watertable level takes longer to reach its maximum when the irrigation development is further from the river (Figure 7.6).

In Figure 7.6, for an aquifer diffusivity (D) with a high value of 200,000 m$^2$/day, the maximum watertable level is reached within about 13 years, whereas for the low value of 20,000 m$^2$/day, the maximum watertable level is reached in 30 to 100 years. The watertable level takes longer to reach its maximum when irrigation areas are larger and are located a greater distance from the river. With the most extreme combination of parameters from Table 7.2 (d = 10 km, A = 1000 ha, R = 500 mm/year, and D = 2000 m$^2$/day), it takes the watertable level about 270 years to reach approximately 90% of its rise.
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Changes in groundwater discharge over time

Groundwater mounds under irrigation developments can result in increased groundwater discharge to nearby rivers. This can have important ecological implications.

The time taken for a groundwater mound to discharge to a nearby river depends on the aquifer diffusivity and the distance to the river. In Figure 7.7 the groundwater discharge to the river (i.e. flux response) is expressed as a fraction of the recharge. The increase in groundwater discharge to a river following an irrigation development can take many years to occur, particularly where the irrigation development is located a long distance from the river.

7.2.3 INTERACTIONS OF GROUNDWATER AS A RESULT OF NEIGHBOURING IRRIGATION DEVELOPMENTS

The groundwater mounds that form under neighbouring irrigation developments have the potential to superimpose upon each other resulting in higher groundwater levels than may otherwise occur. Figure 7.8 illustrates the variation in groundwater level beneath two small (500 ha) neighbouring irrigation developments at different distances of separation. Two sets of parameters are examined. The first assumes a saturated hydraulic conductivity of 1 m/day and a recharge rate of 65 mm/year (Figure 7.8a). The second assumes a saturated hydraulic conductivity of 20 m/day and a recharge rate of 130 mm/year (Figure 7.8b).
Both assume the irrigation developments are 1 km from a river. The former is considered to be more representative of mosaic irrigation developments associated with offstream storages in the Flinders catchment. The results indicate that small size irrigation developments (i.e. 500 ha) exhibit very little interaction during the first 10 years after development. Small interactions occur within a 100-year time frame, but interactions can be avoided when the developments are placed 1 km apart. Placing 500-ha irrigation developments at least 5 km apart excludes any interaction (i.e. under steady-state conditions).

Figure 7.8 Variation in watertable level beneath two neighbouring 500-ha irrigation developments at different distances of separation
(a) Saturated hydraulic conductivity of 1 m/day, recharge of 65 mm/year and 1 km from river. (b) Saturated hydraulic conductivity of 20 m/day, recharge of 130 mm/year and 1 km from river.

7.2.4 SALT STORES IN THE FLINDERS CATCHMENT

There is limited information on salt stores below 1.5 m in the Flinders catchment. Soil sampling to 1.5 m indicates that clay soils associated with the Rolling Downs Group have high levels of salts close to the ground surface and that the recent alluvium and high flood free alluvium have low concentrations of salts. This is confirmed by conductivity–depth sections derived from airborne electromagnetic data. Figure 7.9 indicates that where the Rolling Downs Group is not overlain by alluvium elevated concentrations of salts occur close to the ground surface. These situations are likely to pose a salinity risk under irrigation and careful management would be required. In Figure 7.9 the high level alluvium (e.g. at 6200 m and 9000 m to 14,000 m distance along profile) has a low conductivity and if these areas were developed for irrigation secondary salinisation would most likely occur at the break of slope (e.g. at 5900 m and 8500 m distance along profile). The recent alluvial material adjacent to the Flinders River (e.g. at about 2750 m distance along profile) also contain low levels of salt and, as shown in Section 7.2.2, the watertable level is unlikely to approach the ground surface as groundwater will discharge to the river.
Chapter 7 How can the sustainability of irrigated agriculture be maximised?

7.3 Managing irrigation drainage

Surface drainage water is water that runs off irrigation developments as a result of over-irrigation or rainfall.

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

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

The quality of drainage water will vary depending upon a range of factors including water management and method of application, soil properties, method and timing of fertiliser and pesticide application, hydrogeology, climate and drainage system (Tanji and Kielen, 2002). These factors need to be taken into consideration when implementing drainage system water recycling and also when disposing of drainage water to natural environments.
A major concern with tailwater drainage is the agro-pollutants derived from pesticides and fertilisers that are generally associated with intensive cropping, derived from pesticides and fertilisers and are found in the tailwater from irrigated fields. Crop chemicals can enter surface drainage water if poor water application practices or significant rainfall events occur, after pesticide or fertiliser application (Tanji and Kielen, 2002). Tailwater runoff from pesticides and fertilisers can contain phosphate, organic nitrogen and pesticides that have the potential to adversely affect flora and fauna and ecosystem health, on land waterways, estuaries or marine environments. Tailwater runoff may also contain elevated levels of salts, particularly if the runoff has been generated on saline surface soils. Training of irrigators in responsible application of both water and agro-chemicals is therefore an essential component of sustainable management of irrigation.

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

7.3.1 IMPACTS OF SEDIMENT, NUTRIENTS AND AGROPOLLUTANTS TO RECEIVING WATERS OF THE FLINDERS CATCHMENT

Little information is available on the current or historical water quality of the Flinders River, its associated estuaries and coastal areas. Previous agricultural irrigation developments in tropical Australia have been associated with decreased river and offshore water quality (Brodie et al., 2010; Brodie et al., 2013; Lewis et al., 2009). These reductions in water quality are directly related to the removal of pre-existing ground cover and consequent high sediment loads, as well as the application of fertilisers and pesticides. Fertiliser and pesticide applications are in part absorbed and used by crops, but during rain events, unused nutrients and other chemicals, along with sediments eroded from exposed soils, are washed into rivers. These are released as pollutants into natural ecosystems as river flows spread out and slow in downstream reaches of the river, estuary and coastal receiving areas.

Downstream receiving areas effectively collect material carried in agricultural runoff into habitats including wetlands, mangroves and seagrass meadows of ecological, economic and social importance. Some of these habitats can be sensitive to increased levels of sediments, nutrients and pesticides from agricultural runoff.

The Assessment evaluated the potential for water quality change resulting from agricultural development for freshwater, estuarine and marine receiving areas of the Flinders catchment for four crop types: irrigated fodder, cotton, sorghum and rice (for more details see the companion technical report about waterhole ecology (Waltham et al., 2013)). The effects of crop type, management practice and total area cropped on suspended sediment, nitrogen and phosphorus were evaluated using the Export Coefficient Model (Cuddy et al., 1994; Johnes, 1996; Letcher et al., 2002). Because reliable information describing likely runoff behaviour of pesticides from the crops proposed for the Flinders catchments was not available the potential impacts of pesticides were not directly evaluated for the catchment. The effects of likely pesticide regimes associated with any proposed agricultural development will need to be thoroughly investigated before irrigation development takes place.

The Export Coefficient Model allows broad estimates of the amount of sediment, and the proportion of fertilisers, that wash into rivers in surface runoff. These load estimates can be broadly categorised as small (1 to 10%), moderate (10 to 50%) and large (greater than 50%) relative to baseline estimates, where a 50% increase in loads is equivalent to a 1.5-fold increase. Experience shows that small (1 to 10%) load increases are likely to have minimal ecological impact and moderate (10 to 50%) load increases are likely to have some degree of downstream impact, but without more information accurate prediction of impact is impossible. Large (greater than 50%) increases in loads are considered likely to have major impacts downstream.
For irrigated fodder, the results suggest that negligible change will occur in suspended sediment loads as a result of irrigated production, although moderate increases in phosphorus loads (between 11 and 21% depending on the size of the area planted) are considered likely to have some impact downstream. Increases in nitrogen loads are likely to be small (5 to 10%), with negligible ecological downstream impact. Pesticides are not expected to be used in substantial amounts for irrigated fodder, so are unlikely to negatively affect water quality.

Suspended sediment and nitrogen loads in the Flinders River are not predicted to increase to a large extent under irrigated sorghum, and predicted phosphorus load increases would be moderate (13 to 17%), with little difference between management practices. There is likely to be some downstream impact from increased phosphorus loads. It was not possible to model likely losses of pesticides because of the lack of data.

Poorly-managed ground cover (including no retention of stubble), combined with intensive tillage, can lead to substantial erosion during intense rainfall events in dry tropics cotton production, as shown in the Fitzroy catchment in Queensland (Silburn and Hunter, 2009). When these practices are used, the model predicts losses of 200,000 to 400,000 tonnes/year (from 10,000 ha to 20,000 ha cropped). In contrast, with minimum or zero tillage, stubble retention and contour bank practices, suspended sediment loads can be reduced to near baseline levels. Increases in phosphorus loads above baseline are small, except for a moderate (11%) increase for larger areas under more intensive practices. Predicted increases above baseline for nitrogen range are small (1 to 7%); however, moderate (13%) increases are predicted for larger areas when management practices focus on maximising yields rather than cost efficiency. Moderate increases in nitrogen and phosphorus loads are likely to have some impact downstream. It was not possible to model likely losses of pesticides due to lack of data.

For rice, predicted increases in suspended sediment, nitrogen and phosphorus loads are small for all scenarios modelled. Downstream impacts are unlikely.

Note that cropping areas of between 5,000 and 20,000 ha were used in the modelling reported above. Should agricultural development lead to much larger areas of crops planted, the impact from sediment, fertiliser and pesticide loads will be higher.

### 7.4 Ecological implications of altered flow regimes

Irrigation development necessarily alters the flow regime of streams and rivers, via extraction or diversion of water, the construction of levee banks, or return of irrigation tailwater. Each of these can affect the magnitude, frequency, duration, timing, rate of change and predictability of water flow (Poff et al., 1997).

Flow regime is the dominant driver of water quality, biotic community assemblages, aquatic productivity and the physical form of streams (their bank and channel structures, instream sand bars). It is therefore a critical determinant of the physical and ecological character of streams and rivers.

While some ecological responses to changed flow regimes are gradual and cumulative, for others there are often thresholds above which small changes in flow regime can have large and rapid impacts on ecosystem function and process.

The responses of aquatic ecosystems to irrigation developments are varied and depend not just on the amount of water extracted, but also on the way in which it is extracted, stored and distributed through the landscape; the types of crops grown and irrigation systems used; the management systems in place; and local climate and environmental conditions. Where a dam or weir is built across a stream, large areas of aquatic and terrestrial habitat may be inundated, and formerly shallow (even flowing) aquatic habitat is converted to a deep, lake-like environment that favours different species and ecological processes. Major structures – such as dams or weirs – impair or completely halt the critical passage of fish and other aquatic creatures. Such impediments have resulted in the localised extinction of many fish species from many thousands of kilometres of waterways. While such instream structures can be avoided by using water harvesting schemes that pump water directly from the river into offstream storages, these tend to reduce stored water yield.
Generally, where water is extracted from a river, the reduced flow volume results in reduced habitat availability and poorer water quality. This reduction in flow is the change most often associated with irrigation development, and the Assessment has shown that water extraction would affect the number, standing volume and flushing frequency of waterholes in affected streams. The use of off-farm storage in the Flinders catchment is likely to result in water extraction that is distributed across the catchment, rather than concentrated as with instream storage and, as a consequence, not all reaches of the river system would experience reduced water flows. Some reaches may even experience increased flows, as water is moved from one river or stream to another, either directly or via tailwater flow.

Where a dam or weir is constructed, overall annual flow volume often declines, but where water is released for distribution to downstream irrigators, the stream reaches below the dam or weir may receive higher dry-season flow than prior to development – a process known as supplementation. In such situations, whether streamflow decreases or increases depends on the season and the way in which water is extracted and distributed. Given that the character of any stream depends on its flow regime, increases in flow – including seasonal increases – also greatly alter the character of streams, possibly as much as the outcomes of decreasing flow. There are many studies that have examined the ecological impacts of supplementation and, in some cases, considerable management interventions are being enacted to reduce elevated flow volumes. Relevant north Queensland examples include the Barron/Walsh Rivers (Brizga et al., 2001a; Butler et al., 2008), the Pioneer River (Brizga et al., 2001b) and the Burdekin River irrigation area (Perna, 2003; Butler, 2006; Burrows et al., 2012).

The range of environmental changes that could potentially occur as a result of irrigation development is as varied as the number of developments that could be proposed. Thus, there are limitations to the specific advice that can be provided in the absence of specific development proposals. Even where a specific proposal is being evaluated, many environmental changes associated with irrigation developments are not easily predicted before or during development, and an adaptive management process is required to deal with each as they arise.

For instance, prior to the construction of the Burdekin Falls Dam, the Burdekin Project Committee (1977) and Burdekin Ecological Study (Fleming et al., 1981) concluded that the dam would improve water quality and clarity in the lower river and that para grass, an invasive weed from Africa, then present at relatively low levels, could become a useful ecological element as a result of increased water delivery to the floodplain. However, the Burdekin Falls Dam has remained persistently turbid since construction in 1987, greatly altering the water quality and ecological processes of the river below the dam, and the many streams and wetlands into which that water is pumped on the floodplain (Burrows and Butler, 2007). Para grass (and more recently hymenachne, an ecologically similar plant from South America) have become serious weeds of the floodplain wetlands, rendering innumerable wetlands unviable as habitat for most aquatic biota that formerly occurred there (Tait and Perna, 2000; Perna, 2003, 2004). Several elements are important in the pervasive impact of weeds within the irrigation area. The flow regime has been greatly altered, with seasonal or ephemeral streams becoming essentially perennial. Dry-season conditions restrict the growth of introduced plants, and favour local native species. Perennial flow, especially where nutrient levels are elevated, enables fast-growing weeds to proliferate. In the Burdekin River irrigation area, these have come to dominate most wetlands and – in many cases – to entirely cover the water surface of deepwater lagoons and the margins of stream channels. Similar, though not as widespread and devastating, effects are seen in the Mareeba-Dimbulah Irrigation Area (Butler et al., 2008).

Apart from the processes for extraction, storage and delivery of irrigation water, the way in which irrigation is practised is very important to environmental outcomes. Flood irrigation is commonly practised, resulting in large losses of water from the paddock to nearby streams and wetlands. More efficient irrigation systems will prevent such losses, but these are very expensive to install and maintain. Use of detention basins to capture runoff from farms is becoming more common. Again, this adds cost to farm establishment and operations. Ironically, as Burrows and Butler (2007) point out, some creek systems within intensive agricultural areas are actually maintained in a healthy, though unnatural, state through receiving supplemented year-round flow, rather than more natural seasonal or ephemeral flow regimes.

For example, Barratta Creek (within the Burdekin River irrigation area) was naturally seasonal but now runs year-round, through a combination of tailwater return from flood-irrigated farms (Burrows and Butler,
2007) and elevated watertable levels (DERM, 2013). The increased runoff contains elevated levels of nutrients and pesticides (Davis et al., 2013), but the persistent flow and the resulting aeration prevents the creek from worse environmental outcomes that would result if it ceased to flow and became stagnant – as has happened in nearby streams and lower downstream on the floodplain of Barratta Creek (Burrows and Butler, 2007; Burrows et al., 2012).

The receiving environments (natural streams) in the Assessment area are often of low volume and with limited opportunities for natural dilution. In the absence of diluting flows, contaminants and elevated nutrients may result in poor ecological health outcomes for these low-volume streams. Ecologically, poor quality stormwater runoff from farms is one of the biggest risks to aquatic health. Typically, runoff from farms drains initially to small creeks rather than larger rivers. This is because most rivers are naturally leveed. Even farms close to major rivers tend to drain away from the river bank into smaller creeks that have lesser dilution capacity and that are more susceptible to the impacts of poor quality runoff or the elevated baseflows it may generate.

Wetlands and streams in intensive irrigation districts of coastal north Queensland are severely compromised by altered flow regimes, poor water quality, invasive weeds, and loss of riparian integrity and fish passage barriers (Burrows, 1998; Tait and Perna, 2000; Perna, 2003; Godfrey and Pearson, 2012). Plant diversity is diminished and most water bodies support a lower abundance and diversity of fish species than they previously did (Hogan and Graham 1994a, 1994b; Burrows, 1998). Fish kills and other high profile displays of poor health are common, although in some wetlands fish kills are now rare because all the moderately sensitive fish species have already been eliminated, and only the more tolerant species remain (Butler and Crossland, 2003).

Nutrients are commonly implicated in water quality decline in irrigation districts (Tait and Perna, 2000; Perna, 2003, 2004) and more recent sampling, including in several different north Queensland irrigation districts, has shown that pesticides from cropped farms are regularly present in natural waters (Davis et al., 2012). Less commonly recognised, though, is that most wetlands and watercourses examined in these same irrigated districts have dangerously low levels of dissolved oxygen (Pearson et al., 2003; Butler and Burrows, 2007), a critical element for the survival of aquatic fauna. Farm runoff has been conclusively linked to these low dissolved oxygen levels (Butler and Crossland, 2003; Butler et al., 2007; Perna and Burrows, 2005; Veitch et al., 2008), especially in sugarcane farming areas where sugarcane juice itself has a high oxygen demand and can, when washed into adjacent waterways, rapidly consume all the available oxygen (Pearson et al., 2003; Butler and Crossland, 2003).

Another key component is the role of riparian management. Riparian zones in the dry tropics require active management. For graziers, riparian zones are part of their productive landscape and they therefore manage them as best they can. Irrigators do not use riparian zones as part of their productive landscape and thus do not tend to actively manage them. In the case of the Burdekin River irrigation area, this lack of management has resulted in significant degradation of riparian zones when the land management changed from grazing to cropping (Tait and Perna, 2000). A visually obvious example is the manner in which para grass has proliferated and dominated riparian zones since cessation of grazing and conversion to cropping. In the Burdekin, this has resulted in recent attempts to reintroduce grazing (and fire management) to riparian zones (Tait and Veitch, 2007). Some of these efforts have successfully rehabilitated wetlands, but often riparian corridors are too small for the reintroduction of grazing, fire, or other forms of active management. In Barratta Creek in the Burdekin River irrigation area, an undeveloped buffer of varying width, but up to 1 km wide, was retained when the area was developed for irrigated sugarcane in the mid-1990s. This has served well for some purposes but is gradually declining due to invasive weeds, poor regeneration of riparian trees, and a general inability to graze and burn key locations along the corridor (Tait and Veitch, 2007). Where cattle grazing and/or fire management has been implemented under environmental management funding, it has been successful in controlling weeds and restoring habitat functions. Its wider application, however, is limited by insufficient corridor width to make such operations truly financially viable.

For a variety of reasons, it is preferable to locate farms further from the riparian zones and banks of major streams. The recommended widths vary but for the purposes of maintaining riparian functions, distances of a few hundred metres to 1 km are commonly suggested. However, for viable grazing and/or fire...
management regimes to be implemented, much greater distances may be required. This may create difficulties, as suitable irrigable soils are usually located closer to major streams. Pumping costs from the streams are also a major consideration.

Overall, the condition of wetlands and streams affected by major irrigation development is often poor, especially for those located within the irrigation development itself. The degree to which flow has been reduced is one factor, but often smaller streams that have increased flow are also greatly altered. Some of the impacts result from the development itself (e.g. construction of impoundments and fish passage barriers) and are difficult to rectify, but many significant weed and water quality issues are the result of operations and environmental management after the development has begun operation. These issues can, with appropriate effort and willingness, be managed to a degree. Many of the environmental changes may not be predicted at the outset and may take years to manifest. Therefore, adaptive management is required, as well as a thorough, well-documented understanding of baseline pre-development conditions. In the context of a catchment where data and descriptions of existing conditions are very poor, this is a major gap that needs to be redressed.

7.5 References


Chapter 7 How can the sustainability of irrigated agriculture be maximised?


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