New modelling capability to target stormwater interventions that support seagrass health along Adelaide's coast

Project U.2.5

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Contents

List of Figures

List of Tables

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The project team was a collaboration of five organisations:

SA Water

- Karen Rouse – project leader and primary report editor
- Milena Fernandes – water quality, historical seagrass distribution reconstruction, coastal water quality modelling (AREMp)
- Rob Daly – coastal water quality modelling (AREMp)

- CSIRO
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- EPA
  - Ying He – Source modelling
- Flinders University
  - Shiroma Maheepala – IUWM and ICUWM Source model design and model coupling

Terms and abbreviations

Term	Description
ACDC	Adelaide Coast and Discharging Catchment model – a coupled scenario analysis model described in this report
ACWQIP	Adelaide Coastal Waters Quality Improvement Plan
ACWS	Adelaide Coastal Waters Study
AMLR NRMB	Adelaide & Mt Lofty Ranges Natural Resources Management Board
AREM	Adelaide Receiving Environment Model – operational version
AREMp	Adelaide Receiving Environment Model – Pilot version
ASL	Area-specific loads – a term coined by the ACWS and used in this project
CDOM	Coloured dissolved organic matter
Cu	Copper
DOC	Dissolved organic carbon
DWC	Dry weather concentration
EC	Electrical conductivity, measured in
EMC	Event mean concentration (of water quality constituents such as total nitrogen)
EPA	Environment Protection Authority
HSI	Habitat Suitability Index
ICUWM	Integrated Catchment and Urban Water Management – model developed in this project by integrating the IUWM with its Catchment model
IUWM	Integrated Urban Water Management – model of metro Adelaide developed in earlier Goyder Institute Optimal Water Resources Mix project (Maheepala et al. 2014)
MAR	Managed Aquifer Recharge (an earlier research project)
NOx	Nitrous oxide
OII	Overall Indicator Index
OWRM	Optimal Water Resources Mix – a Goyder Institute for Water Research project which ran from 2013-2014
Pb	Lead
pH	Pure water has neutral pH, neither acid or alkaline
POC	Particulate organic carbon
RA	Running average
SedNet	Sediment Network – a sediment sourcing and delivery model developed by CSIRO
TOC	Total organic carbon
TKN	Total kjeldahl nitrogen, the total concentration of organic nitrogen and ammonia
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total suspended sediment and total suspended solids are often used interchangeably. No differentiation was made during analysis between sediment and other constituents, so we are using the term suspended solids in this report (e.g. sediment, algae). Sediment may be used in the context of referenced material
WDS	Water Data Services
WWTP	Wastewater treatment plant
Zn	Zinc

1 Project description

1.1 Outline

The purpose of this project was to develop tools and knowledge that could inform stormwater management policy and investment decisions to ensure that metropolitan Adelaide's coastal water quality is adequate to support desired environmental values, specifically the presence of seagrass meadows closer to the shore.

This was achieved by repurposing/building on the Integrated Urban Water Management (IUWM) and Catchment models of the Goyder Institute's Optimal Water Resources Mix Project (Maheepala et al. 2014) and the pilot of the Adelaide Receiving Environment model (AREMp[1]) developed by Deltares (for SA Water), as well as by exploring new lines of evidence, to assist government to:

- further develop its conceptual understanding of the urban catchment coastal system
- target stormwater interventions in time, space, and scale
- assess the likely absolute and/or comparative effectiveness of stormwater interventions to achieve/maintain coastal water quality necessary to sustain healthy seagrass.

Specific tasks were to:

- identify suitable data currently available to underpin a catchment-to-coast modelling capability for metropolitan Adelaide with a greater focus on suspended solids and nitrogen, with a lesser focus on phosphorus and coloured dissolved organic matter (CDOM)
- assess adequacy of available suitable data and data gaps for supporting a fully operational modelling capability
- develop an updated IUWM model using the latest public version of the eWater SOURCE platform, capable of integrating water quality and quantity aspects
- demonstrate 'proof of concept' that the IUWM and AREMp models could be coupled to
  - simulate catchment impacts on coastal water quality, under different stormwater management scenarios
  - identify the individual stormwater discharges contributing to coastal impact hotspots.

1.2 Policy context

Through ongoing policy initiatives commencing with the Water for Good Plan (DFW 2010), and continuing with the current drive to develop a Greater Adelaide Integrated Water Management Program, the South Australian government has outlined the role that integrated urban water management is expected to play in ensuring water security, liveability and environmental sustainability for metropolitan Adelaide in the face of growing urbanisation and climate change.

[1] Called AREMp to identify it as a pilot version of the AREM

A key consideration for both liveability and environmental sustainability is the relationship between metropolitan Adelaide and its adjoining coastal waters, and the impacts on coastal water quality of land based discharges comprising stormwater and industrial (Penrice Soda Products) and municipal wastewater. The Adelaide Coastal Waters Study (Fox et al. 2007) established that nutrients primarily from wastewater, and suspended sediment primarily from stormwater, were mainly responsible for the decline in ecological health of the coastal waters since Adelaide was established. South Australia's Environment Protection Act 1993 and subordinate policies and plans have successfully driven significant investment in environmental improvement programs for point source pollutants such as wastewater discharges over the last 20 years and much has been achieved. Stormwater as a diffuse pollutant source is less effectively regulated in this way and has been managed primarily under the framework of the former Catchment Water Management Act 1995 and the current Natural Resources Management Act 2004 through catchment management and natural resource management plans respectively. Investment under these plans initially focused on gross pollutant removal but subsequently expanded to embrace measures that target suspended sediment with stormwater harvesting prominent due to its concomitant water security benefits. Most recently other water sensitive urban design features for example rain gardens and swales, have been encouraged by the government in its 'Water sensitive urban design: Creating more liveable and water sensitive cities in South Australia'(DEWNR 2013).

The need to understand and manage the relative and cumulative impact of stormwater and wastewater discharges in an integrated way was advanced as a policy goal in 2011 with the release of South Australia's 'Stormwater Strategy – the future of Stormwater Management'. Subsequently the Adelaide Coastal Water Quality Improvement Plan (EPA 2013) included relevant strategies, in particular to promote integrated use of wastewater and stormwater across Adelaide (Strategy #2), and integrate monitoring for cumulative impact assessment (Strategy #4).

The theme of integrated urban water management was further developed in the Issues Paper 'Transitioning Adelaide to a Water Sensitive City: Towards an Urban Water Plan for Greater Adelaide' (DEWNR 2014), and a policy framework to facilitate integrated water management for Greater Adelaide is currently being developed taking into account responses to the Issues Paper. To satisfy the majority of stakeholders, one of the main goals of the framework needs to be the simultaneous achievement of a liveable city and healthy coast in a cost-effective, transparent and equitable way. Given the capital intensity of stormwater and wastewater infrastructure, integrated management and investment planning for wastewater and stormwater will inevitably be necessary to deliver such optimal environmental outcomes in the most cost effective way. The tools and information generated by this project will assist in achieving this.

1.3 Scientific and technical context

There is a wealth of available information relating to coastal ecology especially seagrass health, pollutant impacts of stormwater and wastewater, modelling of catchments and coasts, and stormwater interventions that contribute to water sensitive urban design, which is relevant to this project. This section however focuses only on key aspects of particular previous studies that underpin or have direct bearing on the research described herein.

The Adelaide Coastal Waters Study (ACWS) (Fox et al. 2007) primarily investigated the cause of nearshore seagrass decline along the Adelaide coast. These nearshore seagrass meadows are ecologically and economically important for providing habitat for fish and other marine species, and for trapping sediment and dissipating wave energy, and socially important as an icon of good water quality valued by the community. The ACWS concluded that their decline was mainly a result of poor water quality (characterised by high levels of wastewater-derived nitrogen and stormwater-derived suspended sediment) leading to excessive levels of epiphyte growth and direct and indirect shading of seagrass. To improve water quality to a level that would sustain healthy seagrass, a 75% reduction in total nitrogen (TN) and 50% reduction in total suspended solids (TSS) loads from 2003 levels was recommended for land-based discharges overall. For TN this equated to a target area load of 1 tonne per square kilometre. (A target area load for TSS was not provided.) The ACWS did not however provide any granularity to these reduction targets that might reflect more local conditions, distribution of nutrient and sediment discharge points or interactions between them. In addition to the broad findings of the ACWS, this project has drawn extensively on the component input studies of Wilkinson et al. (2005a, 2005b).

The Adelaide Coastal Water Quality Improvement Plan or ACWQIP (EPA 2013) was developed to implement the recommendations of the ACWS as well as to provide a way forward to improve water quality in the coastal waters. The headline achievement of the ACWQIP has been to reduce TSS load target to around 4200 tonnes/year and TN load target to 600 tonnes/year by 2030. The ACWQIP not only draws on the findings of the ACWS but also summarises research undertaken in subsequent years to provide a more refined and geographically nuanced assessment of where efforts should be focused. For example it picks up on the ACWS advice that CDOM in stormwater should be reduced to improve light transmission; acknowledges recent variation in water quality along the coast (fair to good in northern and central portions, and poor in the South); and promotes the application of WSUD to reduce stormwater flows and sediment inputs. Most importantly for the purposes of this project, the ACWQIP recognises that based on location-specific studies and application of the principles of adaptive management, the load targets may be further revised.

In response to findings of the ACWS and to support achievement and refinement of the ACWQIP strategies, the South Australian Water Corporation (SA Water) engaged consultants Deltares and DAMCO Consulting to work with them to develop a biogeochemical model of the Adelaide Coastal Waters. This model to be known as the Adelaide Receiving Environment Model (AREM), would use the Delft 3D model platform and be based around identifying the physical and ecological parameters that characterise habitat suitable for healthy seagrass growth. This pilot version of the AREM (AREMp) was completed in 2014 and has been used in this project. Since that time, SA Water has collected further monitoring data and used this to support the development of an improved and fully operational version of the AREM. Some of these data relating to particle size distribution in stormwater flows, and findings of early model runs regarding the importance of sediment resuspension, have been used as inputs to this Goyder Institute project.

The other main scientific and technical work integral to this project comprises studies undertaken to advance integrated urban water management and water sensitive urban design in Adelaide. Specifically this project builds on the catchment expertise and Goyder Institute OWRM project models[2] (Maheepala et al. 2014). In the OWRM project these models were standalone. A key task of this project was to integrate these two models to enable stormwater quality and quantity to be considered in a holistic manner. The integration approach is described in Chapter 2 of this report.

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[2] These are the rainfall-runoff (catchment) model which generates inflows, and the river system model that describes the urban water system (supply and demands) that were built using different versions of Source http://ewater.org.au/products/ewater-source.

To a lesser extent this project considers approaches developed in the Goyder Improved Water Quality Model for South Australian Catchments project (Kuhnert et al. 2015, Freebairn et al. 2015).

For specific information regarding stormwater interventions the project has drawn heavily on the Urban Stormwater Harvesting Options Study of Wallbridge and Gilbert (2009) that assessed the potential of Adelaide catchments for stormwater harvesting. Their study identified 72 feasible schemes with harvesting potential >250 ML/year across 19 catchments. These schemes have been used as the basis for a modelling scenario that examines the potential for stormwater interventions based on harvesting to alleviate 'hot spots' of poor water quality in coastal waters. The findings of earlier studies by the Goyder Institute and others relating to water sensitive urban design, have been used to inform modelling scenarios that examine the potential for stormwater interventions based on 'filtration' or 'settling' to reduce suspended sediment and thereby alleviate coastal hotspots.

1.4 Description of the project area

The project area (Figure 1-1) covers most of the Metropolitan Adelaide, including its coastal waters, and the major growth areas located outside the Gawler local government area, i.e. Concordia and Roseworthy growth areas (DPTI 2010). It describes the area north of the town of Gawler in the north to Sellicks beach in the City of Onkaparinga in the south, and from east of the towns of Bridgewater and One Tree Hill in the east, to the coast of the Gulf St. Vincent. It excludes a portion of the area governed by Adelaide Hills Local government, between Kangaroo Creek and Mount Bold reservoirs.

The area enjoys a generally Mediterranean climate with mild winters characterised by moderate rainfall and hot, dry summers. The mean maximum summer (December-February) temperature is 29˚C, with some days going over 40˚C. Mean minimum winter (June-August) temperature is 15˚C. Mean annual rainfall is 544 mm, with monthly rainfall varying from 15 mm in February to 79 mm in June and varying considerably from east to west[3].

The project area contains more than 20 local government areas (Figure 1-1) with each Council responsible for local stormwater management within its boundaries. For the purposes of natural resources management and especially the management of rivers and creeks, the whole area falls within the remit of the Adelaide & Mt Lofty Ranges Natural Resources Management Board (AMLR NRMB). There are 20 different catchments (or parts of catchments) within the project area ranging from large stormwater drains, through creeks to portions of the large catchments of the Gawler, Torrens, and Onkaparinga Rivers, and the whole of the catchment of the Patawalonga River (Figure 1-1). These catchments all ultimately drain from the hills in the east to the coastal waters in the west.

Other major land-based discharges to the coastal waters currently comprise wastewater flows from three wastewater treatment plants (WWTPs) at Bolivar in the north, Glenelg in the central zone and Christies Beach in the South. There are also a number of significant historical discharges relevant to this project –wastewater rich in nitrogen and suspended sediment discharged by the Penrice Soda plant at Osborne (discharge ceased in 2013), and sewage sludge outfalls at Semaphore and Glenelg which both ceased discharging in 1993 (Figure 1-2).

[3] Bureau of Meteorology, station 023090 Kent Town, observed records from January 1977 to May 2014

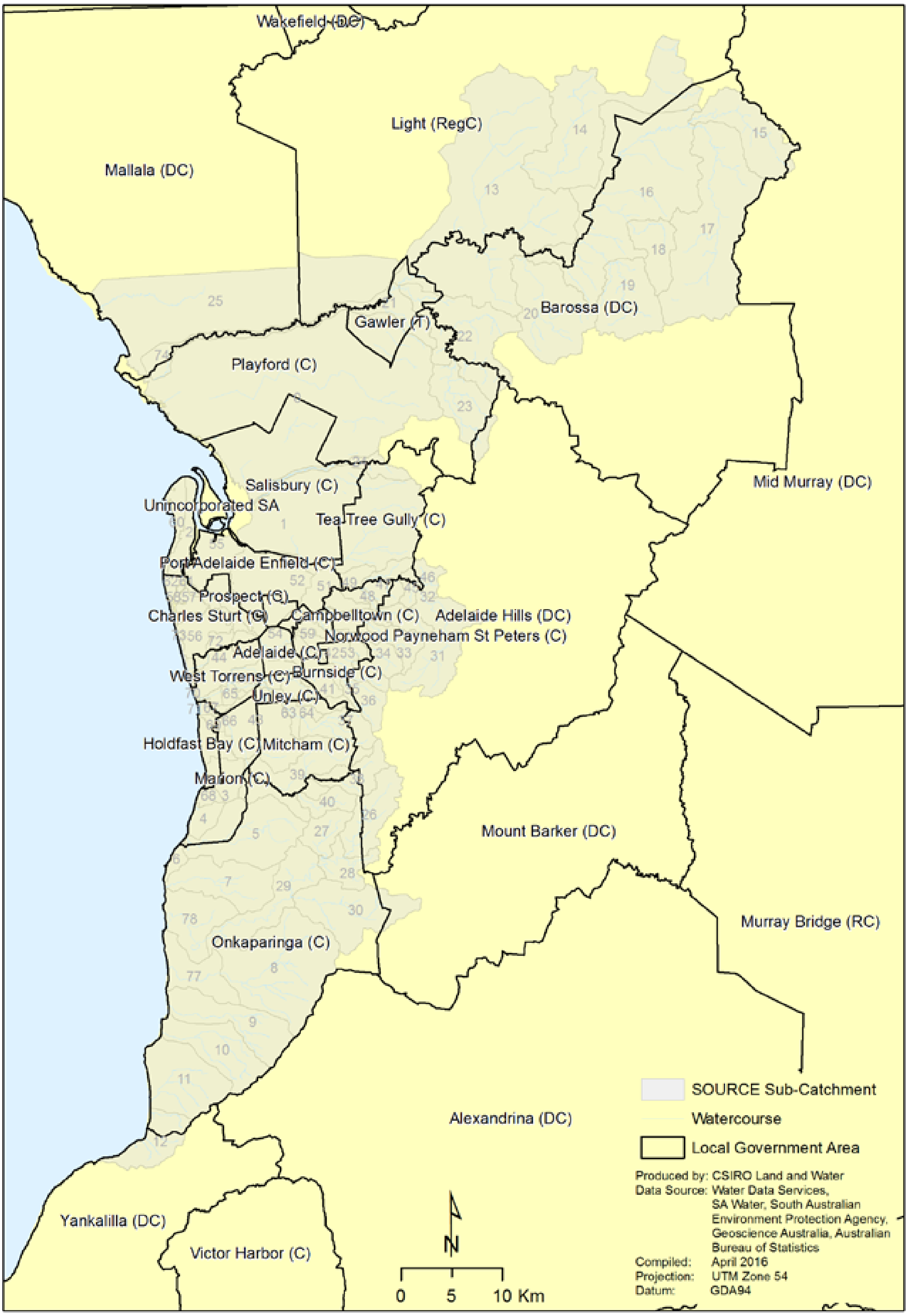


Figure 1-1 Geographic extent of the project area (the Source sub-catchments), noting that the project area includes the waters adjoining the coastline

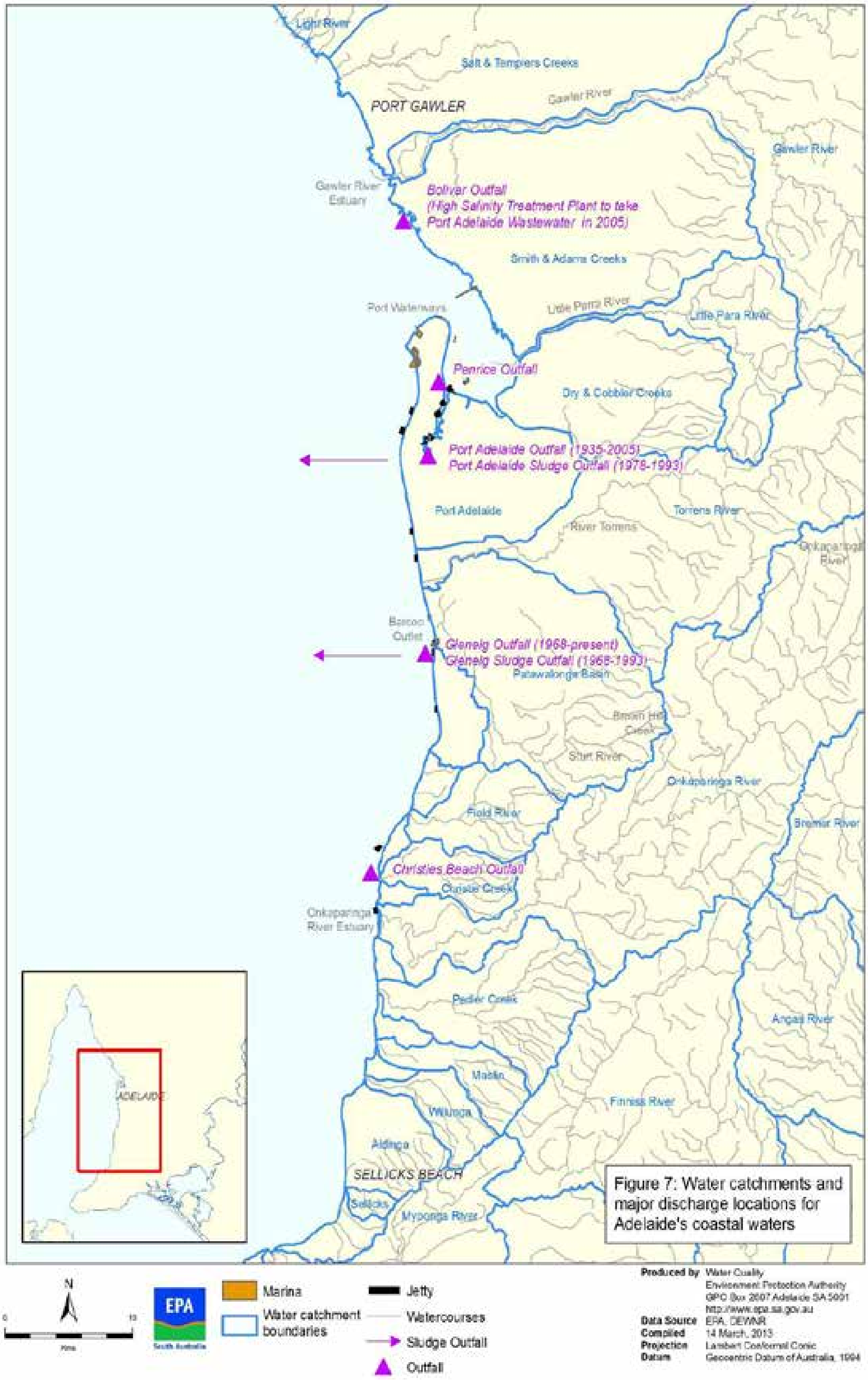


Figure 1-2 The project area showing water catchment areas and major discharge locations to Adelaide's coastal waters (reproduced from Figure 7, McDowell and Pfennig 2013 © Copyright Environment Protection Authority 2013)

The metropolitan coastal waters which are part of the larger Gulf St Vincent, are characterised by seagrass meadows and patches mainly in the north and central parts, with reef systems more prevalent to the south. The latest available mapping of the distribution of seagrass was undertaken by the Department of Environment, Water and Natural Resources in 2013 (Figure 1-3).

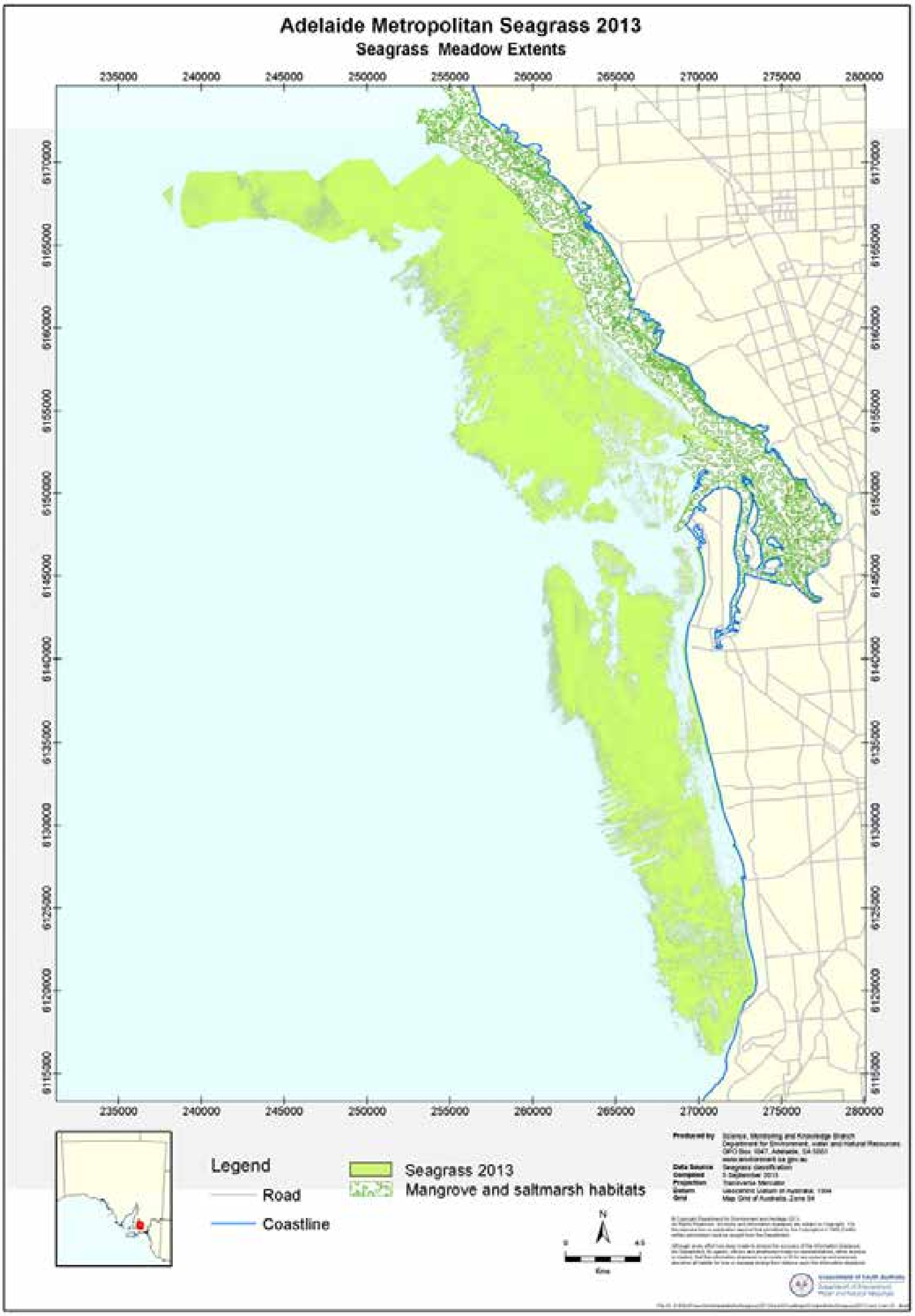


Figure 1-3 Extent of seagrass meadows as mapped by DEWNR in 2013 (reproduced from Fig. 2, Hart 2013)

2 Methodology

This project is based on the development and use of computational models to explore the relationships between environmental characteristics of the project area particularly seagrass health, water quality (mainly nitrogen and suspended solids, but also phosphorus) and stormwater discharges, in the context of conceptual models advanced by the Adelaide Coastal Waters Study (Fox et al. 2007) and subsequently by Cheshire (2015).

Data identified as suitable through considerations described in §2.1 were used as inputs to the models used by the project – the constituent generation model (EMC/DWC), the Integrated Catchment and Urban Water Management (ICUWM) model and the AREMp coastal receiving waters model, a pilot of the AREM model being developed jointly by SA Water and Deltares. For the ICUWM model, a key methodological decision was the choice of constituent generation method. The project team investigated several approaches to determine their suitability. This is covered in §2.1.5, with detail on the alternate approaches in Appendix B .

This project's methodological approach to the ICUWM model, AREMp and EPA conceptual model, are described in §2.2.

Coupling of the ICUWM and AREMp models was undertaken to produce the proof-of-concept Adelaide Coast and Discharging Catchments (ACDC) model according to the methodology described in §2.3.

The models developed in this project were used to consider different scenarios of stormwater interventions described in §2.4 with a view to improving water quality at hotspots of impact determined according to the approach described in §2.5. The different lines of evidence that were used to explore thresholds of impact, such as historic reconstruction of seagrass loss, are also described in §2.5.

In addition to the scenario modelling, another potential applications of the models was trialled as 'proof of concept' that the models can provide support to targeting stormwater interventions. This comprised the identification of sub-catchments contributing most to sediment loads as described in §2.6.

2.1 Sourcing and assessment of general suitability of data to underpin modelling

The quality of the outputs of computational models is inevitably limited by the availability and suitability of data and hence evaluation of the flow and water quality data available to support this project was a key foundational task as described in §2.1.1 and §2.1.3. This analysis revealed that some project-critical new data were required and their derivation is discussed in §2.1.4 and §2.1.5.

2.1.1 Existing catchment water quality data

A range of SA State agencies and institutions have collected flow and water quality data for a range of sites in urban and rural streams and stormwater drainage lines across the project area. Details of these potential data sources are in Appendix A and summarised in Table 2-1. The locations of the selected datasets and their custodians are given in Figure 2-1. Suitability of the datasets for the purpose of constituent generation modelling is discussed in §2.1.2.

Table 2-1 Catchment water quality potential data sources (Note: only composite sample data were used)

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
Water Information	AMLR NRMB	Public	33	1994-present	Integrated composite	Y	Y	Y*	Y
Water Information	AMLR NRMB	Public	5	1972-present	Flow gauge	Y	N	N	N
Water information	AMLR NRMB	By agreement	2	2008-2013	Integrated composite	Y	Y	Y	Y
Goyder MLR WQ Modelling project	EPA	Public	8	1971-2007	Grab	N	N	Y	Y
Goyder MLR WQ Modelling project	EPA	By agreement	27	2008-2011	Integrated Composite	Y	N	Y	Y
Goyder MLR WQ Modelling project	EPA	By agreement	21	1973-2008	Integrated Composite	Y	N	N	Y
Goyder MLR WQ Modelling project	EPA	By agreement	1	2011-2015	Grab	Y	Y	N	N
AREM Project	SA Water	By agreement	3	2010-2014	Discrete composite	Y	Y	Y*	Y
Goyder MLR WQ Modelling project	SA Water	By agreement	1	1996-2013	Integrated composite	Y	Y	Y	Y
WaterConnect	DEWNR	Public	59	1968-present	Flow gauge	Y	N	N	N
MAR Research Projects	CSIRO	By agreement	1	2006	Composite	Y	Y	Y	Y
MAR Research Projects	CSIRO	By agreement	1	2010-2012	Integrated composite	Y	Y	Y	Y
MAR Research Projects	CSIRO	By Agreement	9	2010-2012	Grab	Y/N	Y	Y	Y
Drain 18	Uni SA	Not available at time of study	1	1994-1997	Auto	N	Y	Y	Y
Council projects	City of Salisbury	By agreement	3	2003-2008	Integrated Composite	Y/N	Y	Y	Y
Council projects	City of Playford	By agreement	3	2007-2012	Integrated Composite	Y	Y	Y*	Y

* Calculated as TKN+Nox

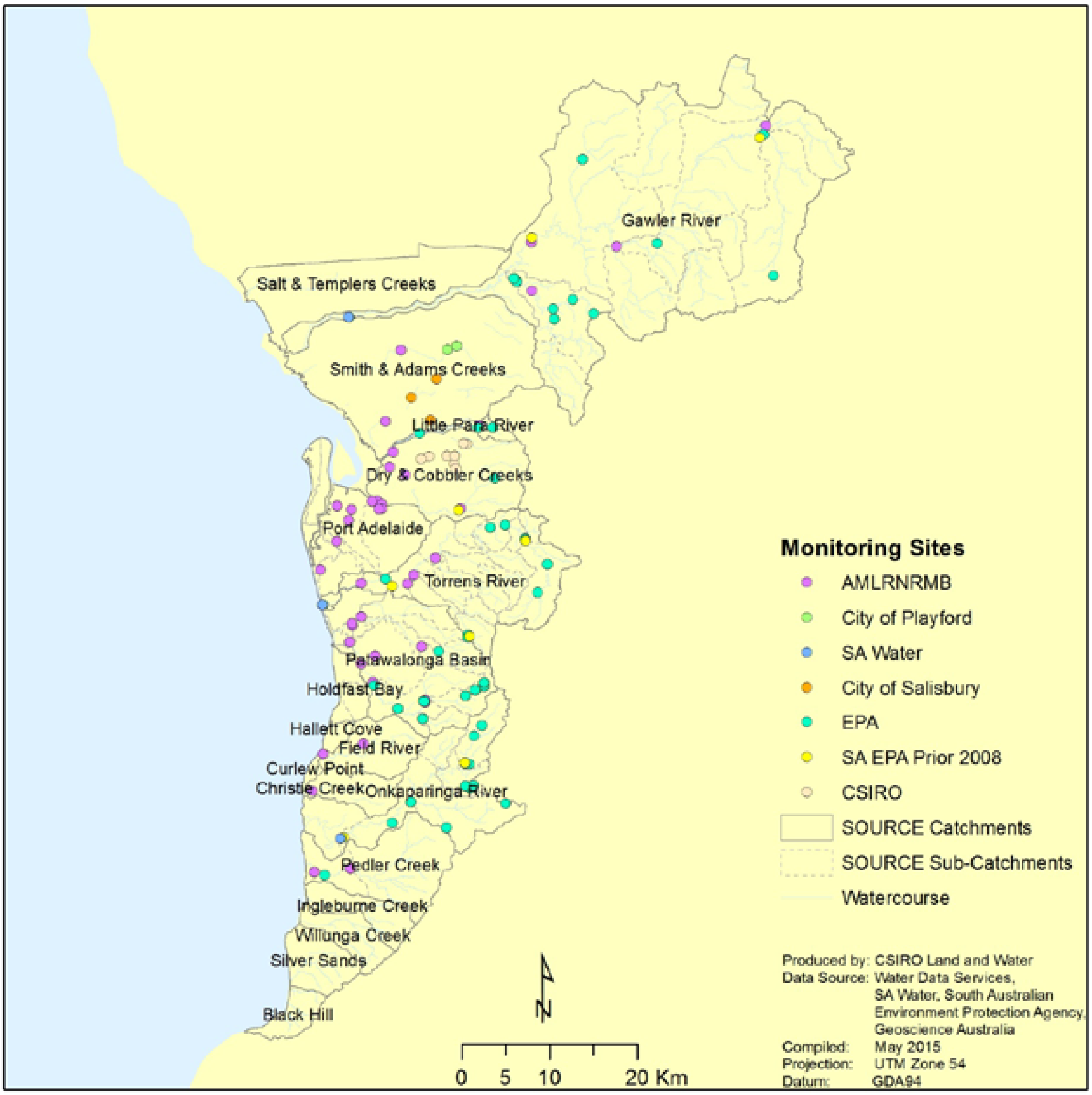


Figure 2-1 The water quality and flow monitoring sites considered in this project and their locations within the modelled sub-catchments. The legend identifies the custodians of the sites

2.1.2 Suitability of available catchment data for constituent generation modelling

A major consideration for the suitability of data relates to its intended use. For the purposes of water quality constituent generation modelling in this project, grab sampling data (e.g. pre-2008 EPA sampling and CSIRO catchment monitoring) were considered not suitable for calculation of loads as they represent only a single point on a hydrograph. Without precise time stamps and highly resolved flow data, relationships between water quality and flow are tenuous.

The EPA provided water quality and flow data collected downstream of a quarry that contained estimates of TSS load that were derived from daily flow and turbidity monitoring combined with event-based TSS grab sampling. These data were suitable for constituent (TSS) modelling and could be conservatively considered representative of extractive industry land use (i.e. mine/quarry functional unit class in SOURCE model) across the project area.

Integrated flow weighted composite sampling data were ideal for use in constituent generation modelling in the current project where load estimates of individual events were required. These included data owned by the AMLR NRMB for 33 sites within the project area, the Parafield Drain site monitored by CSIRO, the Scott Creek site monitored by SA Water, the 27 sites monitored by the EPA, and the 5 stations monitored by the Salisbury and Playford Councils. In addition, the flow weighted discrete composite sampling data collected near the outlets of the Gawler, Onkaparinga and Torrens rivers by SA Water at a sub-daily interval were integrated at a daily time step for use in the model.

2.1.3 Existing coastal data

Potential coastal data is described in Appendix A and summarised in Table 2-2. Use was made of data and methods developed as part of the earlier Adelaide Coastal Waters Study (Fox et al. 2007).

Table 2-2 Coastal data requirements and their potential sources

Data requirement	Source	Period of record	Note
Seagrass distribution	DEWNR	Studies from 1949 to 2013	Maps of the change in spatial extent of seagrass produced from imagery
Seagrass water quality thresholds	ACWS	Best available information as at 2007-2008	Mapping compared to Adelaide Coastal Waters study suspended solids and total nitrogen area-specific loads to verify thresholds
Freshwater inputs	ACWS; UWAOI	2003-2005	Daily flows from rivers and stormwater drains
Wastewater inputs	ACWS	Dependent on life of WWTP	Monthly flows only
Land-based constituent generation data	ACWS	Stormwater 1972-	Used to calculate daily loads from land-based sources, including stormwater

2.1.4 New data

Data available at the start of this project proved insufficient for the needs of the project and hence additional data listed in Table 2-3 were obtained. For example, 'end of catchment' particle size and organic carbon data were collected to improve the quality of the inputs to the AREMp, and new data were generated by manipulation of existing data to yield EMC/DWC values to underpin the ICUWM model described in detail in S2.1.5.

Details of the collection and analysis methods are in Appendix A and summarised in Table 2-3.

Table 2-3 New data and its source/derivation

Data requirement	Source	Period of record	# of sites	Note
Stormwater particle sizes	SA Water	Jul 2014 to Sep 2015	3	Three fractions were reported from which a mean particle size was calculated using LISST particle size analyser, see Appendix A
Stormwater organic carbon (dissolved, particulate, total)	SA Water	Jul 2014 to Sep 2015	3	Same locations as for particle size analysis
Constituent inputs for ICUWM model	Literature & some local data, tabulated in Chapter 3		3	Several approaches were considered with the EMC/DWC approach selected

2.1.5 Constituent generation for input to catchment model

There are several approaches to constituent generation modelling. Selection of an appropriate approach is dependent on many factors, including data requirements, data availability, skillset, range of questions to be answered and the processes of interest, including build-up and washoff (dry time between events and rainfall intensity), catchment wetness, streambank condition, land use (stable or development), sediment type. Four approaches were considered:

- event mean concentration (EMC) and dry weather concentration (DWC), as implemented in the Source platform
- empirical relationships established using a power function, as implemented in the Source platform
- dynamic sediment budget river network (SedNet) model (Wilkinson et al. 2014)
- Loads Regression Estimator and Random Forests (Kuhnert et al. 2015)

All approaches have their strengths, and the decision on which approach to adopt was determined on availability of appropriate data. The power function relies on good flow/water quality relationships which were not generally observed for the project area. Sediment flux modelling using models such as SedNet has not been undertaken in highly urbanised impervious catchments as exist in the project area. Building of the statistical models required for regression modelling requires substantial database processing and analyses beyond the scope of the project. Additionally the Loads Regression Estimator approach may not be applicable for highly impervious urban catchments, given the often flashy, sub-daily nature of urban hydrology. As the flow-weighted water quality data required for the EMC/DWC approach were available, this approach was selected and is described in detail in this section. Details of the other approaches considered, but rejected, are in Appendix B .

Event Mean Concentration (EMC) / Dry Weather Concentration (DWC) approach

The Event Mean Concentration (EMC) and Dry Weather Concentration (DWC) model applies two fixed constituent concentrations (EMC/DWC) to calculate total constituent load as a product of concentration and flow. Within Source, different EMC and DWC values are mapped to areas (often land uses) that have similar hydrologic behaviour and similar rates of constituent generation. These are called Functional Units (FUs) within Source. EMCs and DWCs as implemented in Source are fixed through time. This approach lends itself to estimation of long term loads (useful from a policy perspective) with less confidence for short term estimates.

Two approaches for application of the EMC/DWC model were investigated:

- use published EMC/DWC values for land uses, compiled from a review of the literature, and apply at FU scale
- derive EMC/DWC values from locally collected data and apply at sub-catchment or catchment scale.

Fletcher (2004) in an extensive review of stormwater quality data from around Australia arrived at a list of recommended typical EMC and DWC values for TSS, TN and TP for a range of general land use types (Tables 2.43-2.45 in Fletcher 2004). These aligned well with the land use and FU types used in this project (Table 2-6). Fleming et al. (2010) in a study modelling total nutrients and suspended solids loads for catchment areas in the Adelaide Mount Lofty Ranges derived EMC values for a range of land uses and FU types applied in a SOURCE modelling platform. It was proposed to input both sets of values for respective FU types and compare resulting annual loads with previous estimates (Wilkinson et al. 2005b, Freebairn et al. 2015).

The distribution of suitable local water quality and flow data across the project area would allow derivation of EMC/DWC values at the sub-catchment scale for many areas. The SOURCE Scientific Reference Guide (Kelley & O'Brien 2012) calculates a constituent load (C_{load}) at each time step for an area (can be an FU or a sub-catchment) as the sum of 'slow flow' (SF) (or base flow) and 'quick flow' (QF) (or event flow):

$$C_{load} = (SF \times DWC) + (QF \times EMC)$$

Quick and slow flow conditions are commonly defined by analysing flow duration curves. This presents a challenge for hydrological modelling of impervious urban catchment areas using a daily time step as runoff initiation times and flow durations are frequently in the order of hours, i.e. on a sub-daily time scale. Higher order streams and river reaches are exceptions. Within the SOURCE model, SF and QF are defined through the Catchment hydrological model. In the case of the current project, the SIMHYD rainfall-runoff model is applied at a daily timestep and SF and QF are equivalent to base flow and surface flow respectively.

The following approach to use of data was adopted:

- where suitable local water quality data exist at sub-catchment scale, apply these data at sub-catchment scale – in later reporting these are referred to as the 'Gonzalez' values and/or scenario;
- where sub-catchment scale local water quality data were limited or entirely absent, but suitable data exist downstream within the catchment, apply these data at catchment scale;
- where local water quality data were absent at catchment scale , apply values from Fletcher (2004) at functional unit (land use) scale – in later reporting there are referred to as the 'Fletcher' values and/or scenario.

2.2 Model development

2.2.1 EPA conceptual model

Due to delays in contracting for this project, the EPA separately engaged Dr Anthony Cheshire to develop a conceptual model to support the ACWQIP. In October 2014 the project team participated in a stakeholder workshop designed 'to bring together existing knowledge and experience to support the further development of a series of conceptual models for the Adelaide Coastal Waters region' (Cheshire 2014). The workshop provided background to the investigations subsequently undertaken in this project and also prompted early consideration of how project outputs might be used to inform or refine the conceptual model.

The final report on the conceptual modelling (Cheshire 2015) was provided to the EPA in mid-2015 and while unpublished has been made available to the project team for consideration. It documents a 'series of conceptual models developed primarily to provide a tool that can be used to illustrate and interrogate our understanding of the inter-connections that exist between the catchment and coast'. Cheshire (2015) comments that there is now widespread recognition that the broad-scale degradation of the Adelaide coastal waters has resulted from the cumulative effect of multiple stressors and identifies conceptual models as ideally suited to focusing attention on the net effect of multiple processes rather than towards individual activities. Three types of conceptual model were presented to support the ACWQIP: State Transition, Relationship Diagram and Forester Diagram. It is anticipated that the results of the current project may provide additional information to refine the conceptual models developed by Cheshire (2015).

2.2.2 Adelaide Receiving Environment Model pilot (AREMp) coastal model

The Adelaide Receiving Environment Model (AREM) is a coastal model including hydrodynamic, wave and biogeochemical modules, used to simulate marine water quality and its suitability for seagrasses as a function of land-based inputs. The pilot version of AREM (AREMp) was developed as a proof-of-concept based on historical datasets and existing knowledge of the system (Zijl et al. 2014). Data requirements and their sources are listed in Table 2-4.

Table 2-4 Data sources for AREMp pilot hydrodynamic (FLOW), wave (WAVE) and biogeochemical (WAQ) modules

Module	Data	Data source
Pilot FLOW	Coastline boundary	DEWNR
	Depth data	Australian bathymetry and topography grid
Pilot FLOW	Meteorological data (precipitation, evaporation, winds, temperature, solar irradiance	Bureau of Meteorology, Adelaide airport gauge (selected as the gauge close to the coast and in the middle of the project area)
	Temperature and salinity profiles	ACWS (Kaempf 2006), Desalinisation plant monitoring
Pilot FLOW	Tidal constituents	TPXO 7.2 Global inverse tide model database (a product by the Oregon State University)
	Water level data	Bureau of Meteorology at Port Giles and Outer Harbour
Pilot FLOW	Freshwater inputs	AMLR NRMB (http://amlr.waterdata.com.au), SA Water, Penrice soda ash factory
Pilot WAVE	Wave height, peak wave period, mean wave direction, wind	Climate Forecast System Reanalysis (CFSR), recently developed by the National Center for Environmental Prediction (NCEP, US National Oceanic and Atmospheric Administration
Pilot WAQ	Meteorological data (precipitation, evaporation, winds, temperature, solar irradiance at Adelaide airport	Bureau of Meteorology Adelaide Airport selected as representative for the whole project area
	Nutrient and suspended solids	AMLR NRMB (http://amlr.waterdata.com.au), SA Water, Penrice soda ash factory
Pilot WAQ	Marine water quality data	Integrated Marine Observing System (<http://imos.org.au/saimos.html>), SA EPA, ACWS, SA Water
	Light in the water column, seagrass epiphyte cover	SARDI as part of the ACWS (Bryars et al. 2006, Collings et al. 2006)
Pilot WAQ	Spatial distribution of discharges	NearMap (http://au.nearmap.com/)

The AREMp was updated as part of this project – these are listed in Table 2-5.

Table 2-5 Improvements made to the AREMp model for this project

Improvement	Need
Inclusion of discharge location	Accommodate catchment model simulated discharges
Freshwater (or stormwater) inputs	Discharge points to the coastal receiving waters
Discharge point sources	Discharge points to the coastal receiving waters (e.g. WWTPs, Penrice soda ash factory)
Revised inputs of CDOM and constituent conversion rules	Include new data collected for AREMp calibration (Appendix C)
Updated loads information	Bring in data for rivers and stormwater, WWTPs and Penrice soda factory (Appendix C)
New simulations for 1940 and 1975	Take advantage of historical discharge data derived from ACWS (Pattiaratchi et al. 2007)
New simulations for 2005, 2006 and 2011	Take advantage of simulated discharges from this project's catchment modelling

Habitat suitability modelling

The outputs of AREMp have been used to investigate how water quality compares with suitable habitat thresholds for nine species of seagrass. Thresholds were derived from seven parameters (light, including epiphytes, temperature, salinity, tidal inundation, substrate type, flow velocity and wave dynamics) (Zijl et al. 2014). Habitat suitability curves were then constructed for these parameters from an extensive literature review, including Australian and international studies, on the relationship between seagrass health and the parameter (Erftemeijer 2014). The curves were transformed into suitability indices (values between 0 and 1) for each species and a species habitat suitability index (HSI) derived by taking the lowest value for each grid cell.

Comparison of predicted water quality with habitat suitability thresholds produces a map of seagrass suitability for the area of interest (Zijl et al. 2014). An overall seagrass distribution map was obtained for all species combined by merging the maps of all species together, taking the highest HSI value of any species for each grid cell.

2.2.3 Integrated catchment and urban water management (ICUWM) model

Hydrological modelling was conducted for the Adelaide Coastal Waters Study (ACWS) (Fox et al. 2007) input studies investigating stormwater volumes and pollutant loads (as well as inputs from other sources including wastewater discharge) to the Gulf St. Vincent. Stormwater modelling used the IHACRES approach (Littlewood & Jakeman 1993) and focused on two time periods representing historical (1940 to 2004) and 2005 conditions (Wilkinson et al. 2005b).

Another model of the ACWS area was developed for the Adelaide Coastal Water Quality Improvement Plan using the E2 modelling platform and hydrology was calibrated using its Rainfall Runoff Library (Podger 2004, BMT WBM 2008). This model was rebuilt in the eWater Source modelling platform for the Optimal Water Resource Mix (OWRM) Goyder Institute project (Maheepala et al. 2014). The OWRM project produced an Integrated Urban Water Management (IUWM) model that was developed to identify the most cost-effective and environmentally sustainable mix of water sources to meet potable and non-potable water demands in a given town or city. The OWRM project area had the same geographic boundary as this project, without extending into the coastal waters (Figure 1-2). Water quality modelling was not implemented in this model.

The hydrological components of the IUWM model (catchment runoff simulated using SIMHYD, calibrated to observed data) were used for this project, with rainfall and potential evapotranspiration data from 1/7/2003 to 30/06/2013 from Bureau of Meteorology gauge stations. These are as used in the OWRM project and described in Maheepala et al. (2014).

To support the water quality constituent generation modelling, the project area, as characterized in the Source model, was classified into 12 types of functional units (FUs) (Table 2-6). This classification was based solely on land use type (recoded from South Australian land use mapping, DPTI 2015) and did not account for other differences e.g. soil type, slope, climate or catchment management practices that could be expected to influence constituent generation (Chiew and Scanlon 2002).

Table 2-6 Land use classes and Functional Units in IUWM model mapped to land use classes in stormwater quality literature review by Fletcher (2004)

Land use class	Functional Unit	Land use class (Fletcher 2004)
Commercial	Commercial	Commercial
Education	Commercial	Commercial
Public Institution	Commercial	Commercial
Retail Commercial	Commercial	Commercial
Services	Commercial	Commercial
Forestry	Forestry	Forest/Natural
Reserve	Forestry	Forest/Natural
Agriculture	Horticulture/Agriculture	Agriculture
Horticulture	Horticulture/Agriculture	Agriculture
Food Industry	Industry	Industry
Industrial	Industry	Industry
Utility Industry	Industry	Industry
Livestock	Livestock	Agriculture
Mine/Quarry	Mining	NA
Golf	Open space	Forest/Natural
Recreation	Open space	Forest/Natural
Vacant	Open space	Forest/Natural
Residential Native Cover	Open space	Forest/Natural
Road	Road	Roads
Rural Residential	Rural living	Rural
Non-private Residential	Urban	Residential
Residential	Urban	Residential
Vacant Residential	Urban	Residential
WWTP	WWTP	NA
Beach	Water	NA
Reservoir	Water	NA
Water	Water	NA

WWTP = Wastewater treatment plant; NA = not applicable

Different FUs can be nested within sub-catchments so consideration must be given to scale; concentrations applied must be representative of appropriate scales, e.g. FU, sub-catchment or catchment scale. This is particularly relevant when considering mixed land use catchment areas and the spatial distribution of water quality data.

This project integrated the IUWM water resource planning and catchment models into the one Source model, the consolidated application being called the ICUWM (Integrated Catchment and Urban Water Management) model. The changes/improvements then made to the ICUWM model to service this project are listed in Table 2-7.

Table 2-7 Improvements made to the IUWM (Source) model for Adelaide to create the IUCWM model

Improvement	Need
Rebuild using more recent versions of Source	Internal consistency within the Source application (NSE >0.6 and total volume error <10% for calibration & validation periods, with exception of Onkaparinga River system with NSE of 0.565 (Maheepala et al. 2014) Calibration was on annual (rather than low or high) flows to match with the reporting of the water quality modelling. The EMC/DWC approach tends to overestimate loads from low flows and underestimate those from high flows. Interpretation of results then requires a longer time step (e.g. annual) to smooth potential intra-annual discrepancies.
Refined sub-catchment boundaries and node-link network	Improved representation of the project area to support coupling with coastal model
Full model build, calibrated against gauged flow data	Maintainability and currency of the model application
Land use updated using 2012 data	Improve currency of catchment representation
Addition of water quality (constituent) modelling	To identify locations for stormwater management intervention
Revised functional units (units of similar hydrologic behaviour)	Based on land use type, recoded from SA generalised land use mapping. Used for attributing constituent generation rates to land uses

2.3 Coupling of models to create Adelaide Coast and Discharging Catchments (ACDC) model

The AREMp is a coastal model which includes hydrodynamics, wave and biogeochemical modules to derive marine water quality and its suitability for seagrasses taking into account numerous land-based inputs along the Adelaide coast. The ICUWM model is a catchment-based water quantity and quality model to derive quantity and quality of stormwater discharging from Metropolitan Adelaide under different water and land management options and climatic conditions, to the Adelaide coast.

The ICUWM Catchment model produces output for all catchment areas in the Adelaide area, many of which are ungauged and so not included in the AREMp. Additional coastal discharges were added to represent these additional stormwater sources (Figure 2-2). For simplicity, one discharge point per catchment area was added. The points were located on the coast where discharges would be expected such as a river, creek or stormwater infrastructure. Where many potential locations were available, the largest or most central location was chosen.

The flow from the large northern Adelaide catchment (ICUWM sub-catchment #1) was split equally between two discharge points already in the AREMp: Smiths Creek and Helps Rd Drain.

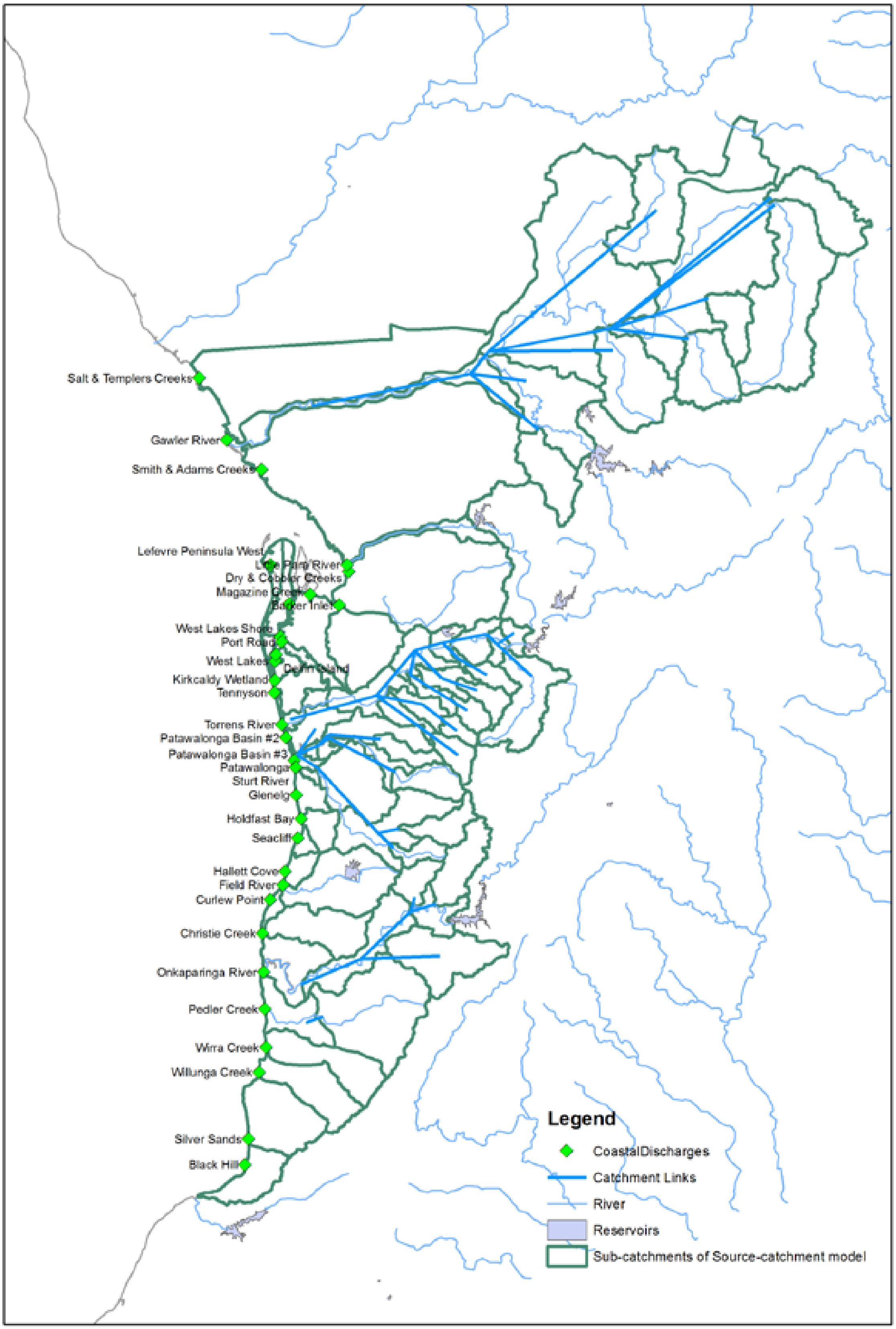


Figure 2-2 Locations of coastal discharge points derived from ICUWM model

ICUWM model output was reformatted to be suitable for input to the AREMp. This was done using a Matlab script. Downstream flow/loads from the ICUWM model were recorded for all of the 'node links' between catchments and saved in a tabular format text file (csv). The text file was read by Matlab and parsed into a data structure by catchment number and load type (flow, TSS, TN, TP). Catchments that discharge to the coast were matched up with the corresponding AREMp discharge point (Figure 2-2). Where several catchment areas discharged through one point (e.g. Westlakes and Patawalonga) their loads were summed. The Matlab script created entries for the volume of each discharge in a file for the AREMp FLOW module (hydrodynamics). Similarly, files for each discharge with flow and concentration values were created for the AREMp WAQ module (water quality). Concentrations were derived from modelled loads, on a daily basis, by dividing by the total flow.

Table 2-8 Assignment of ICUWM sub-catchments to AREMp discharge points, ordered (approximately) from north to south moving down the coastline

Catchment #	AREMp discharge point	Catchment #	AREMp discharge point
3	Lefevre Peninsula East	4	Holdfast Bay
61	Lefevre Peninsula West	69	Seacliff
1	Helps	73	Torrens River
57,63,58,59,62	Kirkcaldy-Westlakes	67,68	Sturt River / Patawalonga
26	Salt & Templers Creeks	5	Hallett Cove
56	Magazine Creek	6	Field River
53	Barker Inlet	7	Curlew Point
2	Dry & Cobbler Creeks	8	Christie Creek
25	Little Para River	78	Onkaparinga River
1	Smith & Adams Creeks	77	Pedler Creek
75	Gawler River	10	Wirra Creek
74	Tennyson	11	Willunga Creek
71	Patawalonga Basin 2	12	Silver Sands
72	Patawalonga Basin 3	13	Black Hill
70	Glenelg		

The total loads were apportioned between expected subtypes. For example modelled total nitrogen was split between particulate and dissolved phases, including organic and inorganic species (particulate organic nitrogen, dissolved organic nitrogen, nitrate, ammonia). This was done at run-time by specifying rules for dividing the total loads among the different species in the 'loads definition file'. Conversion rules were determined based on composite and grab sample data where available as well as literature values.

Interaction between the two models (i.e. the AREMp and the ICUWM model) occurs at the Adelaide coast, where stormwater discharging to the Adelaide coast becomes an input to the AREM model. The temporal scale of the AREMp is two minutes for its hydrodynamics module and 15 minutes for its biogeochemical module, in order to resolve the relevant processes that govern the fate of coastal discharges in the coastal waters", whereas the temporal scale of the ICUWM model is a day. Due to its high temporal resolution, the time-period of the AREMp simulation runs is generally 12 months whereas the time-period of ICUWM model simulation runs is generally more than 30 years. The longer simulation period used by ICUWM is to adequately understand the impact of variability in climate on the volume and quality of catchment runoff.

Two approaches were considered for coupling the models:

- dynamic coupling where the two models are linked in such a manner that execution of one model invokes the other model automatically
- manual coupling where outputs of one model are manually fed as input to the other model, i.e. through file transfer.

The second approach was adopted as the purpose of the exercise was to provide a proof-of-concept, and in general, dynamic coupling is more complex and resource-intensive than a manual approach. This approach also allows more flexibility for future research; a user may want to run scenarios in either model alone, or apply a process between the coupling occurring (e.g. for sensitivity analysis) without having to run a redundant second model.

The differences in temporal scales and simulation periods between the models posed technical challenges, even when using manual coupling. The solution was to reduce the problem space by selecting representative wet, dry and average years (based on rainfall at Adelaide Airport, Figure 2-3) in the recent past so that the urban landscape would be similar to the present, and coupling the models only for those years. The years selected were:

- 2005 (a representative wet year), annual rainfall 473.4 mm
- 2006 (a representative dry year), annual rainfall 234.6 mm
- 2011 (a representative average year), annual rainfall 444.2 mm.

While there were wetter and drier years during the period 1960 to present (Figure 2-3), the selection of years was bounded to the recent past to ensure similar development profiles.

Manual coupling then involved execution of the ICUWM model from 1982-2013 (30 years) on a daily basis and providing daily runoff and constituent loads (TP, TN and TSS) for 2005, 2006 and 2011, for catchments that discharge directly to the Adelaide coast as inputs to the AREMp. There were 33 coupling (or input) points (see Figure 2-4). The AREMp was modified to receive daily flow and constituent loads from these 33 input points. The manually coupled ICUWM–AREMp modelling suite (known as the Adelaide Coast and Discharging Catchments 'ACDC' model) was then available for scenario analysis.

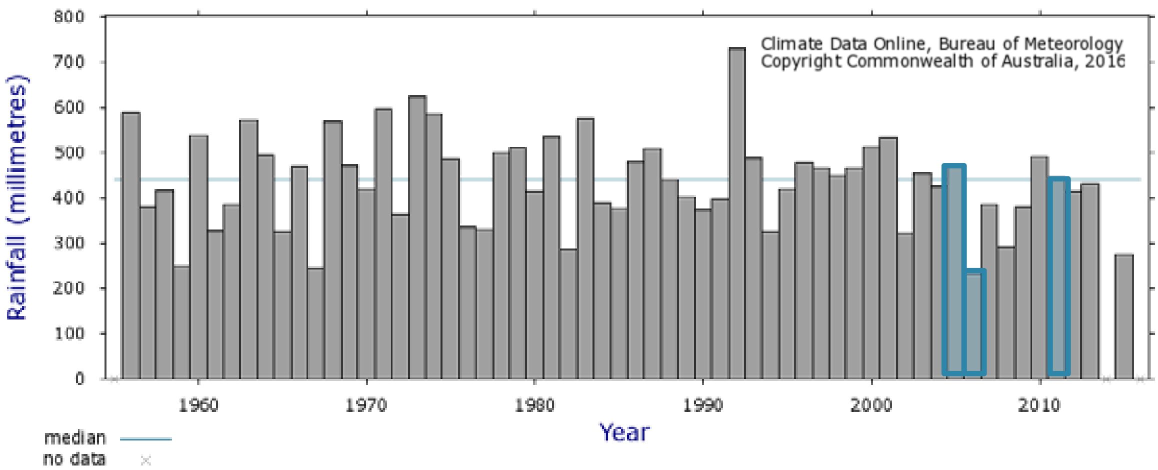


Figure 2-3 Annual rainfall at Adelaide Airport (023034) (Bureau of Meteorology Climate Data Services) (Adelaide airport selected as being representative of the project area)

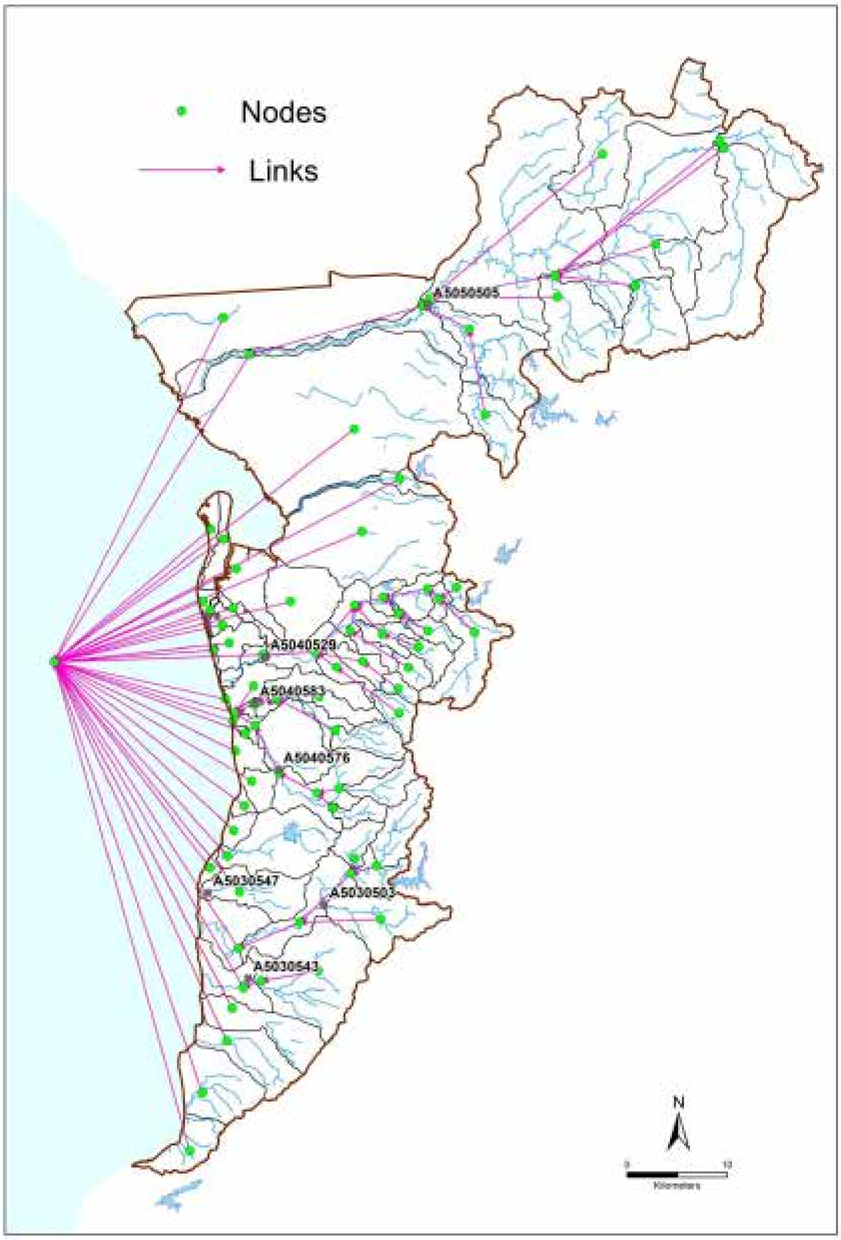


Figure 2-4 Catchments discharging to the Adelaide coast (Maheepala et al. 2014). Also shown is the ICUWM model node-link network

2.4 Scenario selection

A set of 'what-if' scenarios were formulated for ACDC, designed to demonstrate the power of the coupled models to investigate the impact of a range of stormwater interventions on stormwater flows and constituents discharging to the Adelaide coast. These scenarios are described in Table 2-9.

Table 2-9 Description of ACDC modelling scenarios

Scenario identifier	Short description	Purpose	Details
A	Base case	Understand differences in the impact on seagrass health due to stormwater discharging to the Adelaide coast under wet, dry and average climatic conditions, without any stormwater management interventions or stormwater harvesting schemes	Uses locally derived EMC/DWC values, and those from Fletcher (2004) where local data absent. Used to provide the baseline predictions against which to compare alternate scenarios Executed for 2005, 2006 and 2011 (representative wet, dry and average years) Used Area-Specific Loads (ASLs) as indicator for impact caused by coastal dischargers on seagrass
B	100% stormwater harvesting	Consider stormwater management methods aimed at harvesting stormwater for fit-for-purpose uses	Harvesting 100% of scheme capacities for sites identified by Wallbridge and Gilbert (2009) and an additional 2GL/year harvesting scheme proposed for Gawler River
C	50% stormwater harvesting	Consider stormwater management methods aimed at improving stormwater quality and impact of these methods on stormwater flows and constituents discharging to the Adelaide coast	Harvesting 50% of scheme capacities for sites identified by Wallbridge and Gilbert (2009) and an additional 2GL/year harvesting scheme proposed for Gawler River
D	100% stormwater harvesting + 50% reduction in urban TP, TN, TSS	Explore what could be achieved through a 50% reduction in urban constituent generation	Application of constituent model filter to 'urban' FUs using 50% load reduction for both quick (QF) and slow (SF) flow components
E	AREMp with no receiving discharges	Inform the magnitude of reduction in stormwater discharges required to achieve NO significant impact on seagrass health	The AREMp executed for 2011 (average climatic conditions) with no stormwater discharges to the Adelaide coast (i.e. 100% capture of stormwater). Compares suspended solids and nitrogen area-specific loads (ASLs) to those under Scenario A-2011. If significant differences in ASLs observed, then run for wet and dry climatic conditions

2.5 Approach to identifying coastal water quality targets and coastal impact hotspots

Using area-specific load (ASL) thresholds to determine seagrass susceptibility to impact was an approach used by the Adelaide Coastal Waters Study (ACWS) (Fox et al. 2007) and adopted within this project. ASLs are expected to be a measure of anthropogenic pressure. Details of our implementation are given in §2.5.1.

The ACWS estimated nitrogen ASLs to vary between 0.3 and 2.5 tonnes/km^2 in 2003 based on the load discharged (2740 tonnes), area of the Adelaide coastal zone (100 km^2), and residence time (5–30 days) (Fox et al. 2007). These values were compared to the critical nitrogen load limit of 1 tonne/km^2 observed for coastal lagoons in NSW (Scanes et al. 1998, Harris 2008) and a recommendation by Fox et al. (2007) to reduce nitrogen loads to the Adelaide coastal waters by 75% to 600 tonnes, to support the return of seagrasses. The ACWS recognised that nitrogen is only part of the problem, with discharges of suspended solids compounding the impact on seagrasses by reducing light availability. A conservative target of 12% surface irradiance at 9 m depth was used to recommend a 50% load reduction of suspended solids to 4200 tonnes (Fox et al. 2007). By comparing the recommended loads for nitrogen (600 tonnes) and suspended solids (4200 tonnes), this project estimated the corresponding acceptable ASL limit for suspended solids to be 7 tonnes/km^2.

It should be noted however that the thresholds recommended by the ACWS were not based on actual data for the Adelaide coast, but on the best available information for seagrass ecosystems in Australia at the time (Fox et al. 2007, Harris 2008). This project has used local data and models to evaluate these thresholds and consider refinements.

2.5.1 Method to calculate Area Specific Loads (ASLs)

This project explored how AREMp could be used to derive alternative lines of evidence for the relationship between land-based inputs and seagrass status. In this context, the ACWS method for ASLs was extended to include a temporal, as well as a spatial, dimension to account for pressure from nitrogen and suspended solids discharges.

The computation of ASLs is based on the assessment of tracer simulations using the AREMp. Tracer simulations are simulations calculating the transport in the coastal waters of substances that do not interact with other substances and that may, or may not, undergo decay. The computation makes use of the simultaneous release of one conservative (non-decaying) tracer and one decaying tracer from each coastal discharge point in the AREMp, being:

- rivers and stormwater discharges (Torrens, Gawler, Onkaparinga, etc)
- industrial wastewater discharge (Penrice)
- municipal wastewater discharges (Bolivar, Christies, Glenelg WWTPs).

The release of a conservative and a decaying tracer allows tracing of a source, while keeping track of the time elapsed since tracer release, by the concentration ratio of the two tracers. This is not unlike the C14 method used in archaeology. The ASL is made specific for nitrogen or suspended solids by choosing the time-dependent release rates of the two tracers at every discharge point to be exactly equal to the actual release rates of nitrogen and suspended solids respectively. The steps in the method developed by the project team to obtain ASLs from the AREMp tracer simulations was to:

1. calculate annual and depth averaged concentrations for every individual conservative tracer (representing one discharge point)
2. establish a series of relevant equal concentration contour lines C (grammes/m^3)
3. calculate the surface area within each concentration contour S (m^2)
4. calculate the mean concentration of the conservative (C_{mc}) and the decaying (C_{md}) tracers inside that contour
5. calculate the residence time T (days) of the tracer over the area inside the contour using the following equation, where k is the known decay rate (in days) of the decaying tracer

$$T = -ln(C_{md}/C_{mc})/k$$

6. calculate ASL (grammes/m^2 or tonnes/km^2) for each contour from the load discharged W (grammes/day), the surface area S (m^2) and the residence time T (days):

$$ASL = WT/S$$

7. Sum ASLs for individual sources or tracers to give totals per sector or overall totals.

This approach provides annually mean ASL values, but does not allow for assessing temporal variability.

ASLs were further developed as part of this project to include a time component. Seagrasses can tolerate shorter periods of reduced light availability and reduced photosynthesis by drawing from their reserves. Prolonged periods of reduced light availability will eventually affect their health and ultimately kill them. Species with relatively large reserves in the form of a well-developed root-rhizome system can withstand longer periods of reduced light availability than species with smaller reserves. Examples of the former are *Posidonia spp,* examples of the latter are *Amphibolis spp*.

To reflect these considerations, we assessed the variability of ASLs by calculating the maximum values of running averages (RA) over a defined period. We chose periods of one month (1mRA), three months (3mRA) and six months (6mRA) and determined maximum RA values at a location. Using multiple time periods in the calculation of RA provided a temporal integration of ASLs over ecologically significant periods.

The ASL 6mRA can be considered representative for pressure on more tolerant seagrass species *(Posidonia spp)*, while the ASL3m and the ASL1m can be considered representative for pressure on more sensitive seagrass species (*Amphibolis spp)*.

We further assumed that the temporal variability of the concentrations was proportional to the temporal variability of the ASL. For practical reasons, the temporal variability of the concentrations was characterised by tidally averaged concentrations calculated by the AREMp. The within-tide variation of the concentrations was thus neglected, as seagrass impacts are evidently not an issue of minutes to hours. This procedure maintained differences between springs and neaps – these differences are relevant for the residual transport time scales in the Adelaide coastal waters. The tide has a mixed (diurnal, semi-diurnal) character, but the largest component is the tide constituent, i.e. the physical forcing of tides, which has a period of 23.934 hours, so we used an averaging period of 24 hours.

8. In summary, to estimate the maximum ASL for a 1 month period ($ASL_{1mRAmax}$) using the following equation:

$$ASL_{1mRAmax} = ASL\frac{C_{1mRAmax}}{C_{mean}}$$

where *ASL* is the annual mean ASL, C_{mean} is the annual average of the daily mean concentrations, and $C_{1mRAmax}$ is the maximum of the 1 month running average (1mRA) of the daily mean concentrations. While calculating running averages, we made the simulated year circular, meaning that the average over 15-31 December and 1-15 January is also a valid one month running average (1mRA). In a similar way the three month (3mRA) and six month (6mRA) running average values were determined.

2.5.2 Deriving an Overall Impact Indicator (OII)

The fact that both pressures (nitrogen and suspended solids) cause their own form of turbidity which jointly affect the availability of light for seagrass motivated us to derive a single overall impact indicator (OII). This was achieved by normalising the ASLs using the ACWS threshold values for nitrogen (1 tonne/km^2) and suspended solids (7 tonnes/km^{-2}) (see relevant section in §2.1.3) and summing:

$$OII = \frac{ASL_N}{1} + \frac{ASL_{SS}}{7}$$

2.5.3 Historical seagrass loss reconstruction

As a reality check on the ACWS (modelled) nitrogen and suspended solids ASLs and thresholds, we undertook an analysis of seagrass loss mapped during discrete historical periods. In this project, the years chosen for the historical analysis of ASLs were 1940 (pre-dating major seagrass losses) and 1975 (characteristic of peak discharges occurring between 1970 and 1977, 1977 having the largest recorded loss).

Seagrass loss areas (1970–77) were converted to a 20m raster grid and intersected with modelled suspended solids and nitrogen ASL layers from the AREMp. Frequency distributions of pixels intersected with suspended solids and nitrogen ASLs were then created.

Attempts were made to further refine these thresholds by analysing the spatial correlation between nitrogen and suspended solids load contours (pressure) and seagrass loss (effect). The aim of the analysis was the validation of the concept of thresholds that can be linked to seagrass status at a local scale. The analysis thus provides an empirical approach to be used directly to derive hotspots of impact, or as a second line of evidence to support mechanistic modelling of the nutrients-epiphytes-seagrass effect chain.

The review of historical seagrass loss along the Adelaide coast indicates that while the system was in dynamic equilibrium in the 1930s (Westphalen et al. 2004), the rate of loss increased from the 1940s to peak in the 1970s. This historical reconstruction of loss used aerial photography, which varied in extent and location of area surveyed between years. The largest documented area of loss (918 ha) was the nearshore system between Largs Bay and Glenelg during 1971–1977. This temporal pattern of loss followed an increase in population from around 300,000 in the 1930s to >800,000 in the 1970s, a marked increase in urbanised impervious surfaces and several coastal developments (Williams 1974). The situation has stabilised since the late 2000s, with the most recent 2013 survey suggesting that areas affected by loss nowadays might be smaller than areas being recolonised.

2.6 Linking hotspots to catchments

Having used the ACDC to identify stormwater/river discharges contributing to hotspots of coastal impact, the ICUWM model was subsequently used to explore which sub-catchments should potentially be targeted for intervention. To do this, sub-catchment mean annual and mean annual areal TSS, TP and TN loads over 2003–2013 were examined. Mean annual areal values represent constituent discharge from a hectare of sub-catchment. Mean annual areal distribution is useful when comparing discharges and water quality constituents from different catchments because it allows for comparing flows and constituents discharging from a unit area of a catchment.

3 Results

3.1 Filling key data gaps – Particle size distribution

To better understand the nature of the stormwater sediment loads and the mechanism(s) by which they might contribute to coastal impacts, particle size distribution data were collected for the Torrens, Gawler and Onkaparinga discharges across different seasons and phases of the hydrograph as summarised in Table 3-1 and Table 3-2.

Table 3-1 Mean (SD) contribution of each size class to the total particle pool (% and number of samples), and mean particle size, in the Torrens River, Gawler River and Onkaparinga River

River	Season	No of samples	Mean particle size (µm)	SS1 < 16µm (%)	SS2 16-63µm (%)	SS3 >63 µm (%)
Torrens	Summer/autumn	37	66 (23)	16% (6)	38% (9)	46% (13)
Torrens	Winter	54	46 (17)	26% (9)	45% (8)	29% (11)
Gawler	Winter	42	39 (23)	35% (11)	36% (4)	29% (14)
Onkaparinga	Winter	84	77 (46)	20% (9)	35% (11)	45% (18)

Table 3-2 Mean (SD) contribution of each size class to the total particle pool (% and number of samples), and mean particle size, in the Torrens River (in summer/autumn, or winter), Gawler River (in winter) and Onkaparinga River (in winter), according to when in the hydrograph the sample was taken (stage)

River	Stage	n	Mean particle size (µM)	SS1 < 16µm (%)	SS2 16-63µm (%)	SS3 >63 µm (%)
Torrens S/A	Rising	9	68 (16)	17% (4)	38% (8)	45% (9)
Torrens S/A	Peak	12	66 (26)	15% (7)	39% (9)	46% (14)
Torrens S/A	Falling	16	65 (26)	16% (7)	38% (9)	46% (14)
Torrens W1	Rising	4	77 (14)	16% (4)	38% (6)	46% (5)
Torrens W1	Peak	9	47 (21)	23% (8)	47% (8)	30% (13)
Torrens W1	Falling	33	42 (14)	29% (9)	44% (9)	27% (10)
Gawler	Rising	15	58 (27)	27% (12)	35% (5)	38% (15)
Gawler	Peak	8	29 (11)	41% (11)	36% (5)	23% (12)
Gawler	Falling	19	29 (10)	39% (6)	36% (3)	25% (9)
Onkaparinga	Rising	38	92 (46)	17% (7)	32% (10)	51% (15)
Onkaparinga	Peak	15	83 (49)	17% (7)	35% (9)	48% (16)
Onkaparinga	Falling	31	55 (36)	26% (10)	39% (11)	35% (17)

[1]the event labelled 28/7/14 was not included as composed of a series of small events immediately after a large event, and therefore difficult to classify into phases of the hydrograph.

Mean particle size in the Torrens River was found to be higher in summer and autumn, when compared to winter (Figure 3-1, Table 3-1). The higher mean particle size during the dry months of the year coincided with a lower contribution of suspended solids <16µm (SS1) and between 16 and

63µm (SS2) to the total, generally around 16% and 38%, respectively (Figure 3-2). In winter, SS1 increased to ~26% of the total, and SS2 to 45%.

There was no distinction of size fractions between phases of the hydrograph in summer/autumn (Table 3-1), but in winter mean particle size was higher when flow was rising (50-100 µM) when compared to peak or falling flows (generally <60 µM) (Figure 3-2). While rising flows had a similar particle size distribution to summer/autumn, the peak and falling limbs of the hydrograph were associated with a higher percentage of fines (SS1 = 23-29%, SS2 = 44-47%).

The higher contribution of larger particles in the warmer months when soil is dry is possibly a consequence of a greater fraction deriving from channel bed resuspension rather than surface runoff (Walling et al. 2000). When soil moisture is low, infiltration capacity is larger than rainfall and runoff is limited (Barma & Varley 2012). The contribution of channel bed resuspension is likely to be compounded by phytoplankton in summer, as indicated by generally high particulate organic carbon (POC) values (Figure 3-3). In contrast, catchment soil is closer to saturation during winter, and a larger fraction of particles is likely to derive from catchment runoff during peak and falling flows. No clear correlation with POC was observed overall (Figure 3-3).

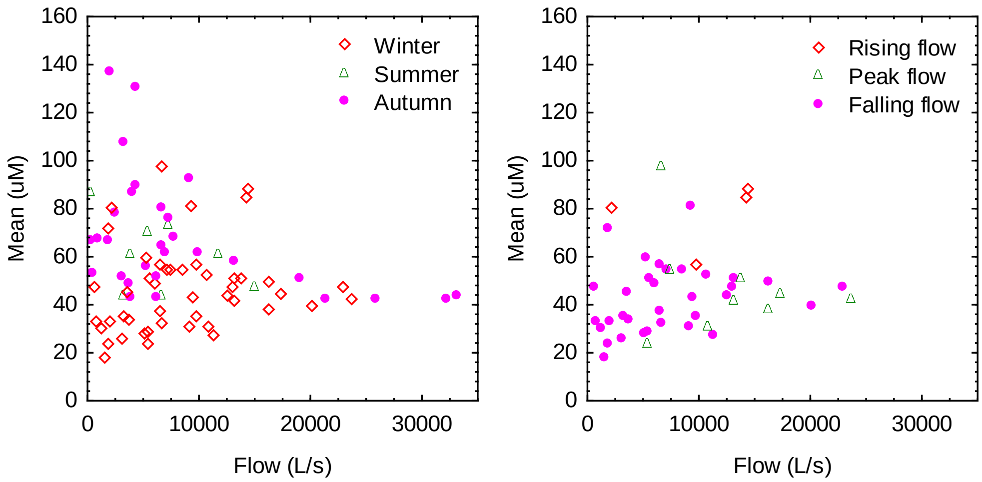


Figure 3-1 Mean particle size of suspended solids versus flow in the Torrens River according to season (left) and, for winter samples, when the sample was taken in the hydrograph (right)

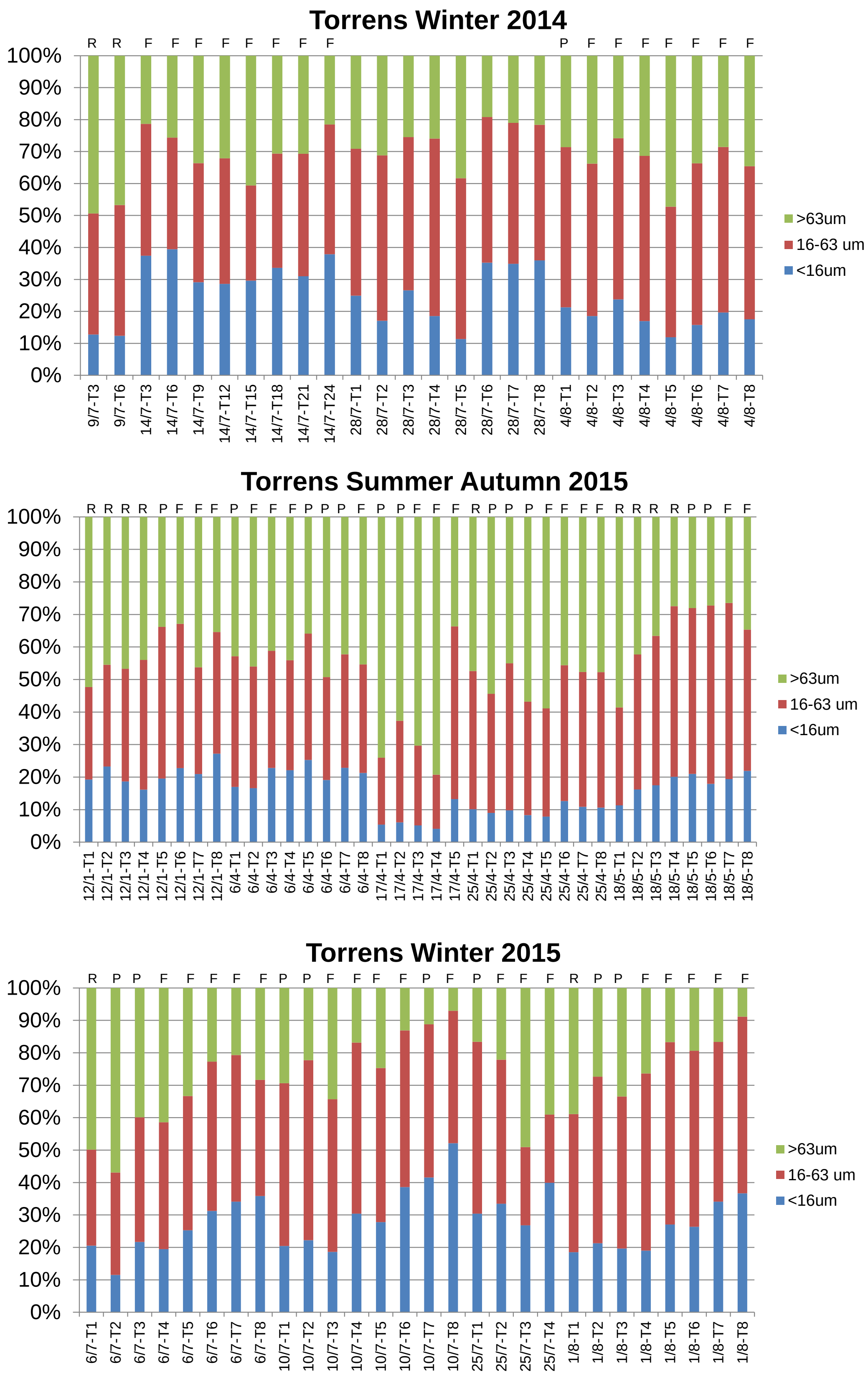


Figure 3-2 Distribution of particles in the Torrens River according to size in the winter of 2014 (top), summer/autumn of 2015 (middle) and winter of 2015 (bottom). The letters above the bars indicate when the samples were taken in the hydrograph: rising I, peak (P) and falling (F) flows. The 28/7/14 event is not labelled as composed of a series of small events immediately after a large event, and therefore difficult to classify into phases of the hydrograph

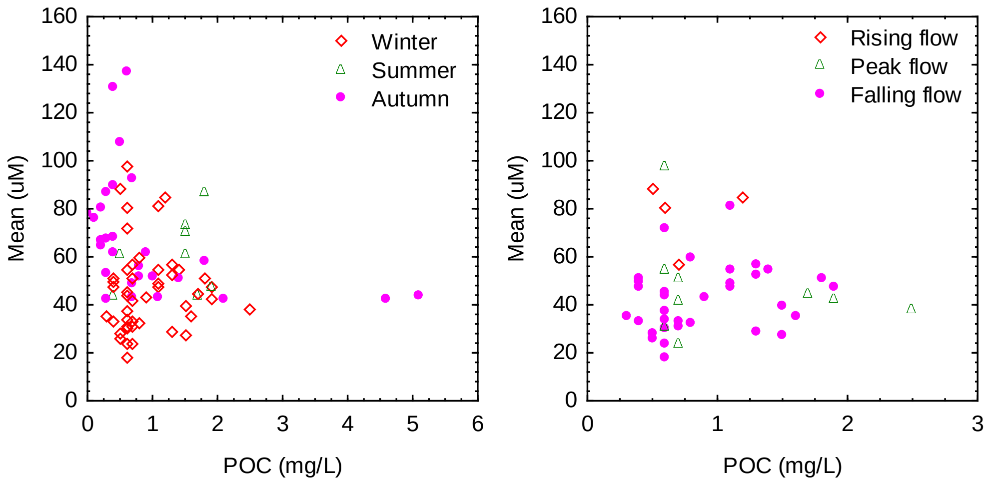


Figure 3-3 Mean particle size of suspended sediments versus particulate organic carbon in the Torrens River according to season (left) and, in winter, according to when the sample was taken in the hydrograph (right)

Particles <63 mM constituted approximately 70% of total sediments in the Gawler River, equally distributed between SS1 and SS2 (Figure 3-4, Figure 3-5). The separation between phases of the hydrograph was even more marked for this river, with rising flow having a mean particle size generally around 58 mM, decreasing to ~29 mM during peak and falling flows (Table 3-2, Figure 3-4). This trend is explained by the contribution of particles <63 mM increasing from ~62% of the total during rising flow to more than 75% during peak and falling flow. This increase was accompanied by an increase in POC; while ascending flows generally had POC concentrations <2.5 mg/L, values as high as 6 mg/L were recorded during peak and falling flows (Figure 3-4). As for the Torrens River, a balance between bed channel resuspension and surface runoff in the catchment might contribute to these trends.

Mean particle size in the Onkaparinga River also changed with phases in the hydrograph, rising flow generally carrying particles in the range 40–210 mM, and falling flow <80 mM (Figure 3-4). The mean particle size of peak flow was intermediate, typically varying between 50 and 130 mM. POC did not seem to be a major determinant in particle size (Figure 3-4), although the highest values of POC (>6 mg/L) were recorded during falling flow when the mean particle size was ~20 mM. Particles <63 mM represented just over half of the total (Table 3-2, Figure 3-5), with SS1 contributing around 17% of the total. This value increased during falling flow to ~65% of the total, with SS1 representing 26% of the total.

Overall, the Gawler River had the finest particles (mean particle size 39 mM), and the Onkaparinga River, the coarsest (77 mM) (Table 3-2). The finer particles in the Gawler might be a consequence of a more gentle topography and prevalence of horticultural land-use (Walling et al. 2000) in comparison to the Torrens and Onkaparinga, which receive inputs from rivers draining the Mount Lofty Ranges.

The particle size distribution in the rising limb of the hydrograph was similar for all 3 rivers (Table 3-2), with a mean particle size around 60-90 mM, particles <63 mM contributing about 50–60% of all particles, SS1 representing ~17% in the Torrens and Onkaparinga, and 27% in the Gawler. Particle size decreased during peak flows, remaining low during falling flows, SS2 dominating in the Torrens River (44–47%), and SS1 in the Gawler River (39–41%). This difference between dominating size classes explains why the Gawler River has a lower mean particle size during peak and falling flows. For the Onkaparinga River, particle size decline was only noticeable in the falling limb of the hydrograph, with a marked increase in SS1 to 26% of the total.

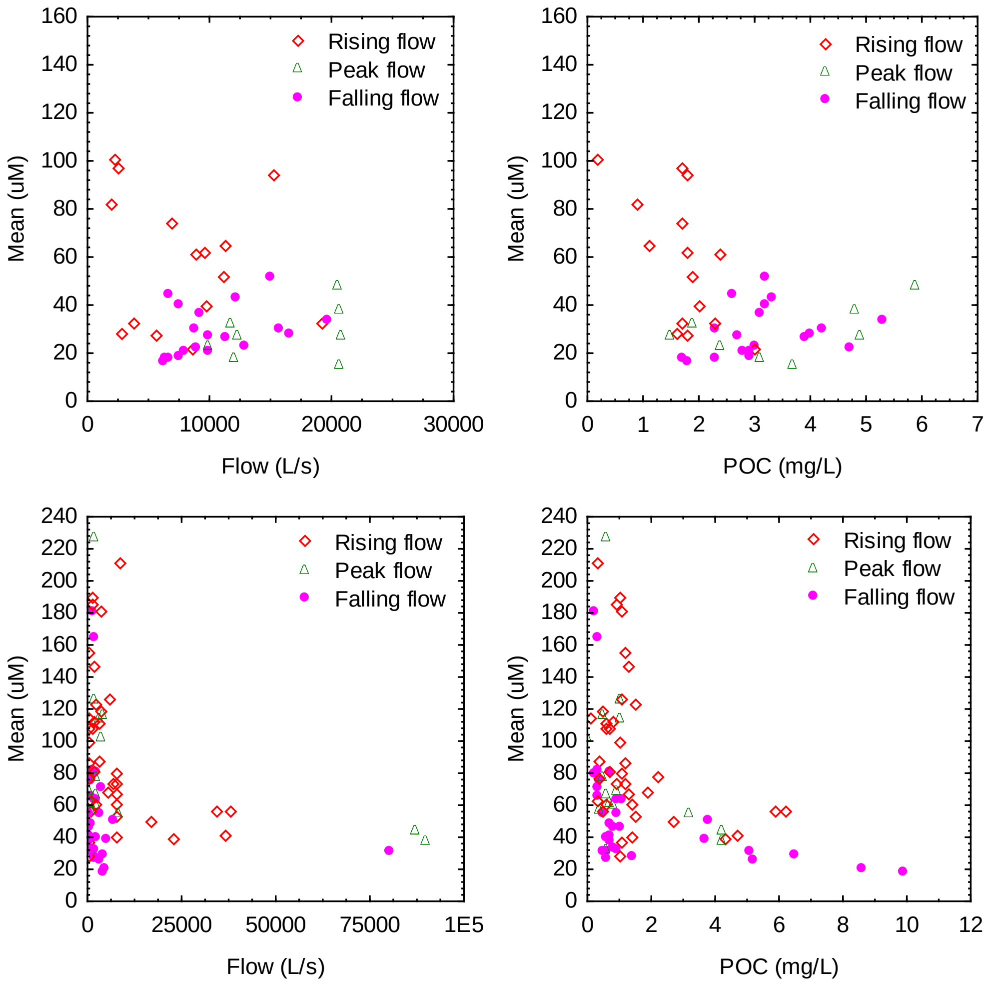


Figure 3-4 Mean particle size of suspended sediments in the Gawler (top) and Onkaparinga (bottom) rivers according to flow (left) and particulate organic carbon (right), and when the sample was taken in the hydrograph

There were no significant relationships between particle size distribution (mean particle size, percentage of each fraction size) and flow for any of the rivers. One interpretation for this lack of significance could be that particle size is controlled by supply rather than flow hydraulics. However, this would need to be tested with more field data. There is however a noticeable trend to a more uniform particle size distribution when flows exceed 15,000 L/sec, with particles in the Torrens River converging to a mean particle size around 40-50 mM, those in the Gawler to 20-50 mM, and those in the Onkaparinga to 40–60 mM.

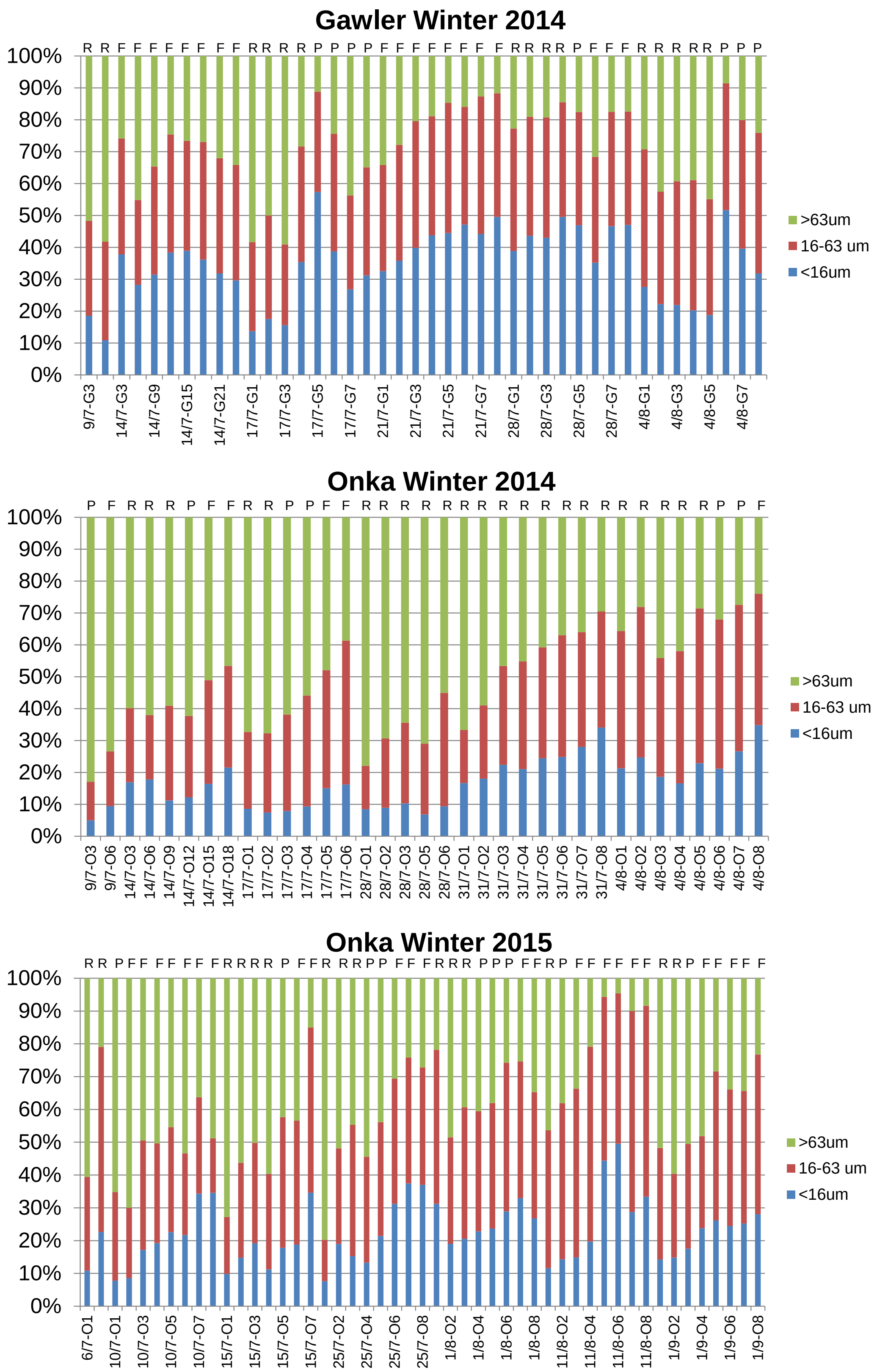


Figure 3-5 Distribution of particles in the Gawler (top) and Onkaparinga (middle, bottom) rivers according to size. The letters above the bars indicate when the samples were taken in the hydrograph: risil(R), peak (P) and falling (F) flows

3.2 Constituent generation – improved TSS, TP and TN event mean and dry weather concentration (EMC/DWC) values

3.2.1 Rationale for selected approach to constituent generation for ICUWM Model

Constituent generation is a fundamental step in the development of a hydrological model such as the ICUWM model delivered by this project. From assessment of the available data as described in Chapter 2, it was decided to use the EMC/DWC approach to constituent generation and not apply a Power Function model. This was because for nearly all sites, no clear relationships between constituent concentrations and event cumulative flow volumes were observed (Figure 3-6). TSS concentrations were expected to most likely be influenced by event flow magnitude however this was rarely observed. Typical examples of this lack of relationship between concentration and flow are shown in Figure 3-6 for downstream stations of some of the major water courses in the project area.

The EMC/DWC approach involved assessing the performance of three sets of values based on 1) a range of literature derived values, 2) a study on water quality in the Adelaide Mount Lofty Ranges, 3) Adelaide metropolitan flow gauging and composite sampling station data. Scenarios using these values are discussed in subsequent sections of this chapter and are compared to measured flows and loads reported in recent water data audit for the Adelaide region (Jones 2015). While every effort was made to ensure the quality of data collection and analyses with respect to measured flows and loads, it is noted that no dataset of this nature is exact and comparisons with modelled results do not necessarily mean either is entirely accurate. These comparisons serve as a quality check to ensure modelled results are within the range of what is realistically expected based on available data.

Figure 3-6 Total suspended solids (TSS) concentrations (mg/L) plotted against cumulative event flow (ML) for four gauging stations across the project area

3.2.2 Determination of EMC and DWC values

Prior to this project it would have been necessary to rely solely on literature-based EMC and DWC values. For example EMC/DWC values derived from studies by Fletcher (2004) and Fleming et al. (2010) as applied to Functional Units (FUs) in the SOURCE catchment model are given in Table 3-3. (Scenarios using each of these sets of values are hereafter referred to as 'Fletcher' and 'Fleming' scenarios.) However the Fletcher values have not been derived from relationships observed in South Australia and the Fleming values were derived from mainly rural catchments in the Adelaide Mt Lofty Ranges. For these reasons, both sets of values were considered to be likely sub-optimal when applied to Adelaide urban environments.

Table 3-3 Literature based EMC and DWC values for TSS, TN and TP applied at Functional Unit (FU) scale, from Fletcher (2004) and Fleming et al. (2010)

Functional Unit	Total area (Ha)	Fletcher	Fletcher	Fleming et al.	Fleming et al.
		TSS_EMC(mg/L)	TSS_DWC(mg/L)	TSS_EMC(mg/L)	TSS_DWC(mg/L)
AGRICULTURE	26,574	140	20	131	10
COMMERICIAL	9,252	140	16	61	14
FORESTRY	34,244	40	6	66	23
HORTICULTURE	30,244	140	20	308	21
INDUSTRY	6,670	140	16	40	12
LIVESTOCK	62,128	140	20	184	12
MINING	2,443	140	16	40	12
OPEN SPACE	7,703	40	6	43	10
ROAD	21,679	270	50	61	14
RURAL LIVING	24,860	90	14	131	10
URBAN	35,266	140	16	61	14
		TN_EMC(mg/L)	**TN_DWC(mg/L)**	**TN_EMC(mg/L)**	**TN_DWC(mg/L)**
AGRICULTURE	26,574	3	1.1	1.6	0.7
COMMERICIAL	9,252	2	1.3	1.8	1.5
FORESTRY	34,244	0.9	0.3	2.1	1
HORTICULTURE	30,244	3	1.1	5.3	3.4
INDUSTRY	6,670	2	1.3	1.3	1.3
LIVESTOCK	62,128	3	1.1	2.1	0.8
MINING	2,443	2	1.3	1.3	1.3
OPEN SPACE	7,703	0.9	0.3	1.8	0.6
ROAD	21,679	2.2	2	1.8	1.5
RURAL LIVING	24,860	2	0.9	1.6	0.7
URBAN	35,266	2	1.3	1.8	1.5
		TP_EMC(mg/L)	**TP_DWC(mg/L)**	**TP_EMC(mg/L)**	**TP_DWC(mg/L)**
AGRICULTURE	26,574	0.6	0.09	0.13	0.04
COMMERICIAL	9,252	0.25	0.14	0.1	0.08
FORESTRY	34,244	0.08	0.03	0.16	0.11
HORTICULTURE	30,244	0.6	0.09	0.93	0.34
INDUSTRY	6,670	0.25	0.14	0.12	0.07
LIVESTOCK	62,128	0.6	0.09	0.24	0.23
MINING	2,443	0.25	0.14	0.12	0.07
OPEN SPACE	7,703	0.08	0.03	0.18	0.05
ROAD	21,679	0.5	0.5	0.1	0.08
RURAL LIVING	24,860	0.22	0.06	0.13	0.04
URBAN	35,266	0.25	0.14	0.1	0.08

In this project, a similar method to Fleming et al. (2010) was used to calculate EMC and DWC values, with the advantage of being based on observed data from the project area. Data were limited to flow-weighted composite sampling data for this purpose and were applied at the sub-catchment scale where sufficient coverage existed, and at the catchment scale where data were limited. EMC/DWC values for each area were derived from downstream integrated composite sampling sites (Figure 2-1). Where data were not available for catchments (e.g. southernmost catchments (Figure 2-1) EMC/DWC values from Fleming et al. (2010) were applied.

EMCs were derived by calculating the cumulative flow for each composite sample and calculating the load as a product of this flow and the concentration measured in the composite sample for TSS, TN and TP. The sum of loads for each site gave a single figure representing the most accurate estimation of load recorded over the period of sampling. These cumulative flow volumes were then multiplied by a constant concentration (estimated EMC) resulting in a total load estimate that was subsequently optimized to match the calculated total load using the Microsoft Excel Solver function. Optimized EMC values were then the best EMC estimate across the sampling period. DWCs were derived using the EMC method described above however only the lowest decile of cumulative flow events were used for the calculations. Calculated EMC and DWC values relating to SOURCE model sub-catchment numbers are shown in Table 3-4; the scenario using this set of values is hereafter referred to as the 'Gonzalez' scenario.

Table 3-4 Gonzalez EMC (mg/L) and DWC (mg/L) values and number of events (in brackets) for TSS, TN and TP, calculated from observed data relating to SOURCE sub-catchments (identified by Catchment name and #)

Station No & name	SOURCE Catchment	SOURCE SC#	TSS EMC	TSS DWC	TN EMC	TN DWC	TP EMC	TP DWC
A5050510 Gawler River Virginia Pk	Gawler	74,21, 22, 23	35 (52)	59 (6)	1.9 (52)	3.6 (6)	0.22 (52)	0.41 (6)
A5051004 North Para River Turretfield	Gawler	14,15,17,16,18,19, 20,21	22 (36)	8 (4)	1.3 (37)	0.8 (4)	0.17 (37)	0.10 (4)
A5051005, A5051013 Smith Creek Womma Rd, Helps Drain Summer Rd	Smith & Adams Creek	0	44 (63)	22 (7)	1.7 (58)	2.4 (6)	0.17 (58)	0.23 (6)
A5041006 Little Para River d/s Pt Wakefield Rd	Little Para River	24	34 (14)	31 (2)	1.2 (14)	1.3 (2)	0.12 (14)	0.18 (2)
A5041005, PDS Dry Creek 250m u/s Salisbury Hwy, Parafield Drain Station	Dry and Cobbler Creek	1	35 (14)	36 (2)	0.8 (15)	1.6 (2)	0.05 (15)	0.21 (2)
A5040529 Torrens River Holbrooks Rd	Torrens River	55	51 (62)	29 (7)	1.5 (62)	1.5 (7)	0.10 (62)	0.10 (7)
A5040578 First Creek Botanic Gardens	Torrens River	60,37	73 (60)	30 (7)	1.4 (60)	1.5 (7)	0.16 (62)	0.11 (5)
A5041014 Torrens River Seaview Rd	Torrens River	73	56 (46)	37 (6)	1.4 (46)	1.3 (6)	0.09 (46)	0.09 (6)
A5041023 Torrens River d/s Second Creek	Torrens River	43,36,52,50,48,49, 46,47,33,51,34,54, 35	30 (66)	11 (8)	2.0 (66)	2.9 (8)	0.07 (66)	0.07 (8)
A5040523 Sixth Creek at Castambul	Torrens River	32	179 (69)	8 (7)	1.7 (69)	0.4 (7)	0.15 (68)	0.02 (7)
A5040583 Brownhill Creek Adelaide Airport	Patawalonga Basin	72,76,66,42,65	14 (57)	14 (7)	1.0 (57)	2.3 (6)	0.15 (57)	0.56 (7)
A5040901 Brownhill Creek Scotch College	Patawalonga Basin	38	18 (48)	11 (6)	0.7 (48)	0.4 (6)	0.05 (48)	0.05 (6)
A5040580 Brownhill Creek u/s Keswick Creek	Patawalonga Basin	64	120 (274)	45 (31)	1.3 (274)	1.4 (31)	0.18 (274)	0.19 (31)
A5040549, A5041042 Sturt River d/s/ Anzac Hwy, Drain 6 Oaklands Pk	Patawalonga Basin	44	49.2 (69)	31.5 (8)	1.2 (69)	2.2 (8)	0.10 (69)	0.1 (8)
A5040576 Sturt River d/s/ Sturt Rd	Patawalonga Basin	40,41	92.0 (635)	73.2 (72)	1.6 (635)	1.6 (72)	0.35 (635)	0.31 (72)

Station No & name	SOURCE Catchment	SOURCE SC#	TSS EMC	TSS DWC	TN EMC	TN DWC	TP EMC	TP DWC
A5040518 Sturt River u/s Minno Creek	Patawalonga Basin	39	47.0 (491)	8.1 (56)	2.3 (491)	1.5 (56)	0.87 (490)	2.01 (56)
A5041011, A5041012 Barker Inlet Wetland on HEP Drain, Barker Inlet Wetland on NAE Drain	Port Adelaide	53	27.1 (151)	19.3 (18)	2.2 (151)	2.8 (18)	0.28 (151)	0.40 (18)
A5041024, A5041025 Range Wetland Outlet, Magazine Wetland Outlet	Port Adelaide	56	22.8 (82)	22.6 (9)	2.7 (80)	6.3 (9)	0.58 (80)	1.31 (9)
A5041041 Port Road Drain u/s Old Port Road	Port Adelaide	58	17.0 (35)	4.9 (4)	0.9 (35)	1.6 (4)	0.11 (35)	0.30 (4)
A5041016 Kirkcaldy Wetland at Nash Street East Grange	Port Adelaide	57	48.3 (149)	103.5 (18)	1.2 (149)	2.9 (18)	0.17 (136)	0.23 (18)
A5031010 Field River u/s mouth	Field River	6	98.0 (60)	30.4 (7)	1.6 (60)	2.5 (7)	0.10 (60)	0.07 (7)
A5030547 Christie Creek Downstream of Galloway Road	Christie Creek	8	109 .72 (60)	14.95 (7)	1.31 (60)	1.12 (7)	0.12 (60)	0.04 (7)
A5030502 Scott Creek at Bottom	Onkaparinga River	27	48.9 (462)	13.3 (54)	4.1 (451)	0.8 (65)	0.35 (451)	0.06 (65)
A5031005 Onkaparinga u/s Estuary Old Noarlunga	Onkaparinga River	28,29,30,31	25.0 (45)	3.1 (5)	1.1 (45)	0.6 (5)	0.08 (45)	0.03 (5)
A5031009 Pedler Creek u/s Mouth	Pedler Creek	9,77	155.5 (42)	163.5 (5)	1.7 (42)	1.5 (5)	0.30 (42)	0.19 (5)

The main limitation with the EMC/DWC approach is that in general, the method overestimates loads from lower flows and underestimates loads from higher flows. Hence, this is why interpretation of results using this method should be restricted to long term (e.g. annual) estimates where the law of averages has a chance to smooth potential discrepancies.

3.3 Overcoming lack of inflow data for the ICUWM model

Within the ICUWM model, 3 inflow nodes for the Gawler, Torrens and Onkaparinga catchments were used. These nodes contained a time series of inflows representing releases and spills from reservoirs upstream of the project area catchments as well as replacing flows for upstream sub-catchments (as these could not be calibrated separately) within the project area. These inflow nodes corresponded to flow records at gauging stations for which flow data existed but water quality data were absent. Inflows were configured to replace flows (and consequently loads) from upstream sub-catchments.

- A5050503 – South Para River (Gawler River catchment), replacing flows from SC#23, 24
- A5040501 – Torrens River at Gorge Weir, replacing flows from SC#46, 47, 32
- A5030500 – Onkaparinga River at Clarendon Weir, replacing flows from SC# 27, 28, 29.

The IUWM model hydrological calibration using these inflow nodes is described in detail in Maheepala et al. (2014). The influence of these inflow nodes on total catchment flows was significant at times particularly during wet years. For example, in 2005, 51% of the total Gawler modelled flow was contributed by upstream inflow (Table 3-5).

Table 3-5 Annual contribution of inflow nodes proportional to total modelled flow volumes of each catchment over the period 2000 to 2013

Year	Gawler	Torrens	Onkaparinga
2000	22%	8%	55%
2001	25%	9%	35%
2002	11%	0%	0%
2003	19%	18%	45%
2004	50%	22%	56%
2005	51%	54%	48%
2006	9%	16%	0%
2007	8%	10%	2%
2008	8%	9%	0%
2009	49%	28%	56%
2010	23%	58%	69%
2011	48%	13%	0%
2012	42%	30%	23%
2013	14%	42%	26%
Average	**27%**	**23%**	**30%**

To represent the import of constituents via the inflow nodes the constituent configuration was set up for TSS, TN and TP. The important limitation to recognize is that the configuration is a single concentration value (analogous to an EMC value) that is static in time within the model. The effect of this is that any flow generated at the node is assigned the same concentration value so that load is simply a product of flow and concentration. As inflow volumes were considerable in some years (23-30% on average), the configuration values were highly sensitive model parameters.

There were no flow weighted composite sampling water quality data for South Para River from which to derive constituent loads and concentrations. The South Para River inflow node was consequently configured with EMC values derived for the downstream Gawler River gauging station at Virginia (A5050510) (Table 3-4) noting that between 62% and 87% of flow comes from the North Para River system (based on flow data from 2011-2013).

There were no flow weighted composite sampling data for the Torrens River at Gorge Weir station (A5040501). The inflow node constituent configuration for the Gorge Weir station (A5040501) was calculated based on a combination of EMC values calculated for the Sixth Creek station (A5040523) upstream of the confluence with the Torrens River, and the downstream station on Torrens River (A5041023) (Table 3-4). On average (2010-15), flows from Sixth Creek (A5040523) contributed 25% volume of flows at the downstream Torrens River station (A5041023). Using the flow proportions a 'combined' EMC was calculated according to

$$EMC_C = (0.25^{a} EMC_S) + (0.75^{a} EMC_T)$$

where EMC_C is the combined EMC, EMC_S is the EMC for Sixth Creek (A5040523) and EMC_T is the EMC for Torrens River (A5041023). This theoretically reduces the bias of using either station EMC for the inflow node with caveat that A5041023 is a few kilometers downstream of Gorge Weir (A5040501) so is likely to underestimate the proportional contribution of Sixth Creek.

There were no flow weighted composite sampling data for the Onkaparinga River at Clarendon Weir station (A5040501). This inflow node was configured using the EMC values calculated for the upstream Scott Creek station (A5030502) (Table 3-4). It is noted that the Scott Creek station is upstream of SOURCE sub-catchments SC#28 and SC#29.

3.4 Comparison of ICUWM modelled and recorded annual flows and loads

Annual flows and loads calculated from the ICUWM model runs were compared with recorded loads calculated based on gauged data reported by Water Data Services in a recent audit of stormwater data conducted for the EPA (Jones 2015). Discharges from the Gawler River, Torrens River and Onkaparinga River catchments were selected for comparison as they represented the three main runoff and river inputs to Gulf St. Vincent by volume within the project area. The years of 2005, 2006, 2009, 2010 and 2011 were chosen as these represented a range of below, above and median rainfall years for Adelaide (Figure 3-7).

Monthly flow volumes from 2005 to 2011 for the Gawler, Torrens and Onkaparinga Rivers revealed hydrological differences across the systems. The Torrens flows much more regularly than the Gawler and Onkaparinga especially during dry and median rainfall years. The Onkaparinga River flows are highly seasonal and it is not unusual for no flow to be recorded over several continuous months however when peak flows occur they can be very high. Gawler River flows are also seasonal and little to no flow can occur during dry years. Monthly peak flows in wet years and months are lower in magnitude than both the Torrens and Onkaparinga Rivers.

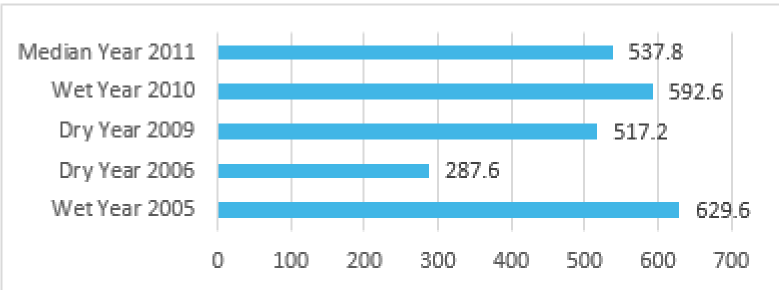
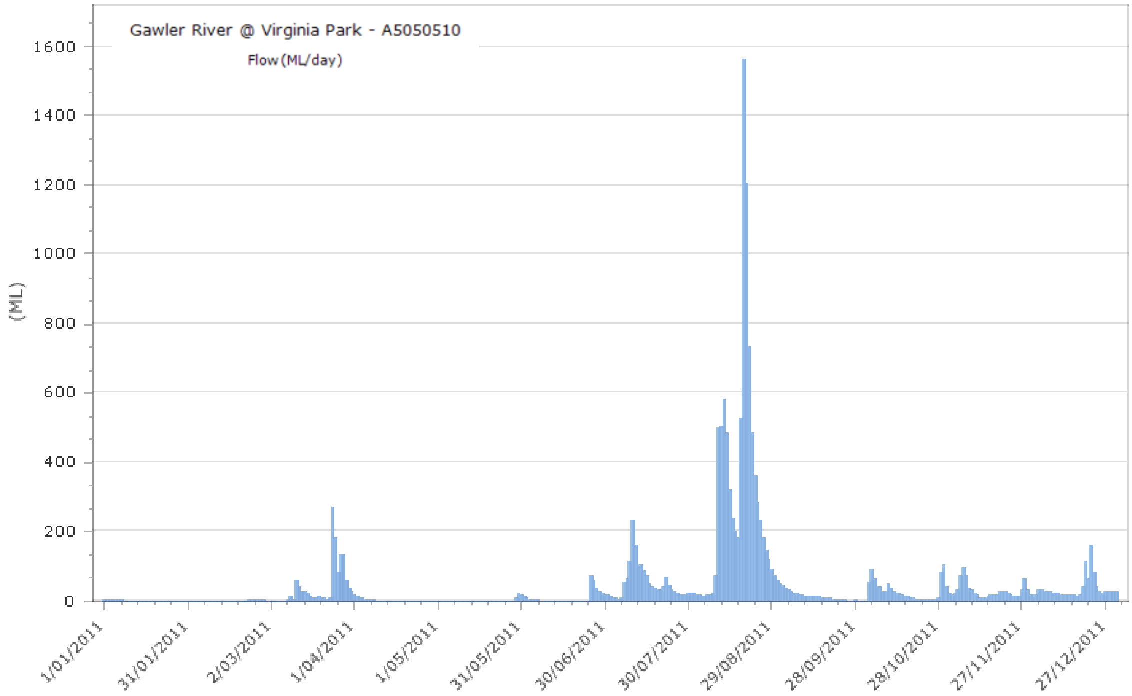


Figure 3-7 Annual rainfall totals gauged at Kent Town BOM weather station; median rainfall 537.8 mm (1977-2015) (Kent Town selected as being representative for the selected catchments)

Flow was not recorded at Gawler River (Virginia Park A5050510) for 2005, 2006, 2009 and unreliably in 2010 but for the median rainfall year of 2011 it is evident that the majority of flows were recorded over 3 consecutive months in late winter/early spring. There were extended periods of low or no flow in summer and autumn months and continuous low flow from late spring into summer (Figure 3-8 (top). The upstream gauging station on the North Para River (A5050504) contains historical flow records from May 1972 to July 2008. The wet year of 2005 saw several continuous months of high flows during winter and spring and little in the rest of the year. In the following dry year of 2006, very little flow was recorded during winter and no high flow peaks were recorded (Figure 3-8 (middle)).

For the Torrens system, wet years of 2005 and 2010 were punctuated by a few very wet months with very high monthly flow totals. Dry years of 2006 and 2009 did not record these large flow events in winter and overall monthly volumes were lower than other years. The median year of 2011 saw reasonably evenly distributed monthly flow volumes with a wet autumn continuing into winter and tailing off in spring (Figure 3-8 (bottom)).



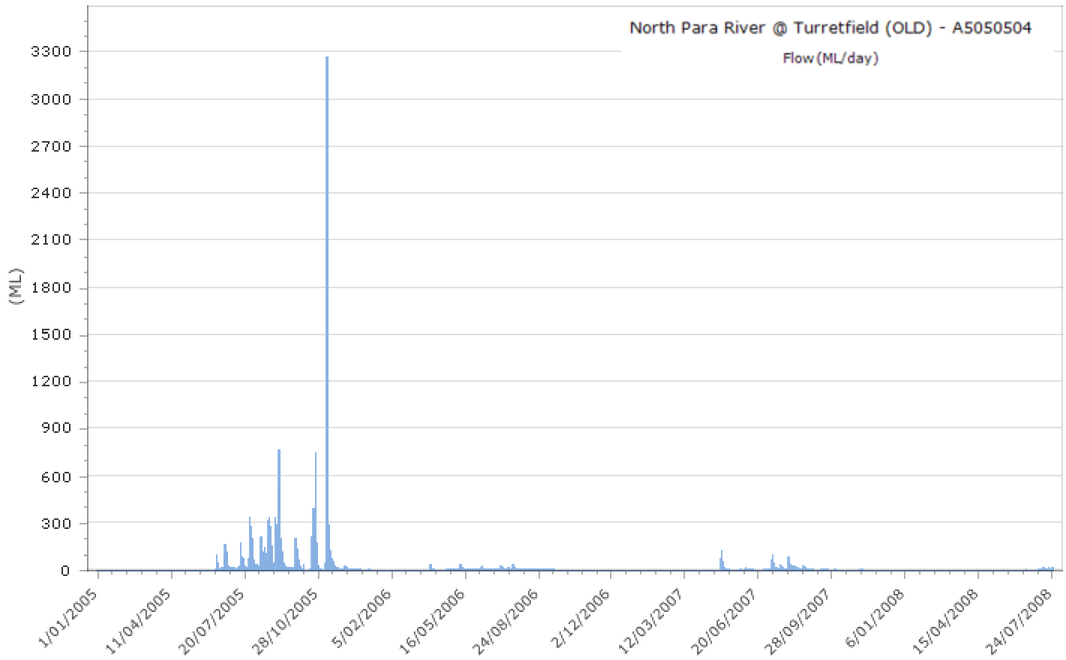
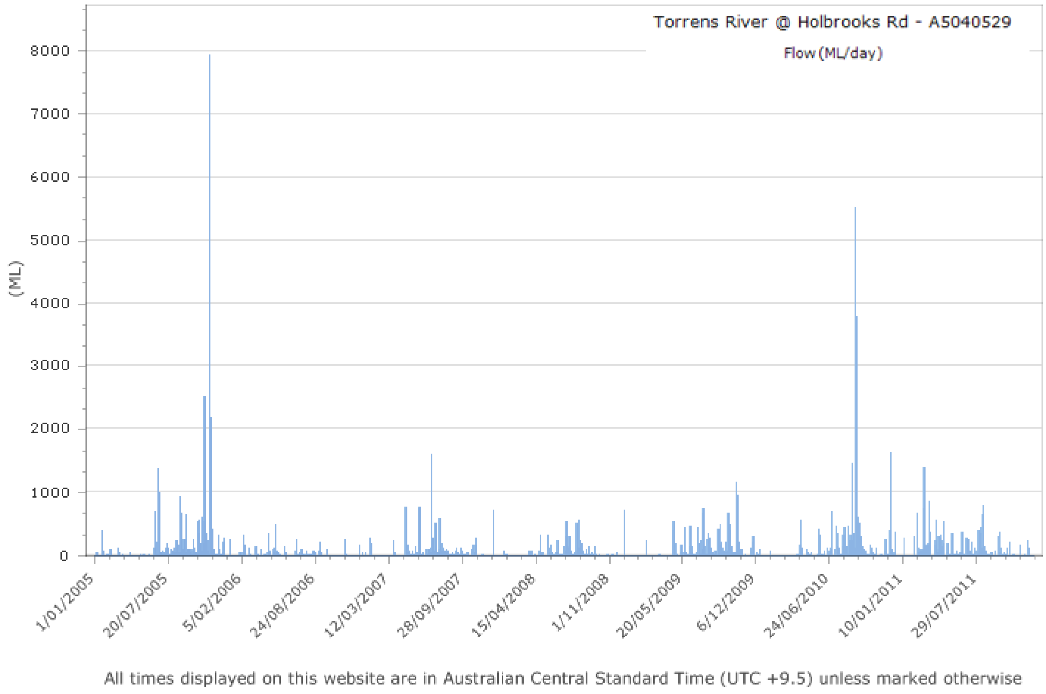


Figure 3 8 Monthly flow volumes (ML) for 2011 (median year) gauged at the Gawler River Virginia Park station (A5050510) (top), and for 2005-2008 gauged at the North Para River (A5050504) (middle) and Torrens River Holbrooks Road (A5040529) (next page) stations (AMLR NRMB 2016)



Prepared by Water Data Services on behalf of Adelaide and Mount Lofty Ranges Natural Resources Management Board.
Disclaimer: The data provided in this report has been collected under the framework of a Quality Management System (WDS - NATA AS / NZS ISO 9001/2008).
Water Data Services reserve the right to adjust this data based on new calibration data and/or new information that may become available.
Water Data Services and its employees cannot be held accountable in any way for the use of this data and/or any subsequent interpretation of the data.

Figure 3-8 Monthly flow volumes (ML) for 2011 (median year) gauged at the Gawler River Virginia Park station (A5050510) (top), and for 2005-2008 gauged at the North Para River (A5050504) (middle) and Torrens River Holbrooks Road (A5040529) (next page) stations (AMLR NRMB 2016)

The Onkaparinga River flows generally vary considerably with season and across years; flows are generally winter dominant (June to September) and large peaks are seen during wet months and little to no flow at other times of year. Flows at the A5031005 gauging station contain records from May 2006. Observations from 2007 to 2011 recorded no flow in the months of January through to April and very little in November and December. Barely any flow was recorded during the dry year of 2006 whereas during the wet year of 2010 large peaks were recorded in July and August (Figure 3-9).

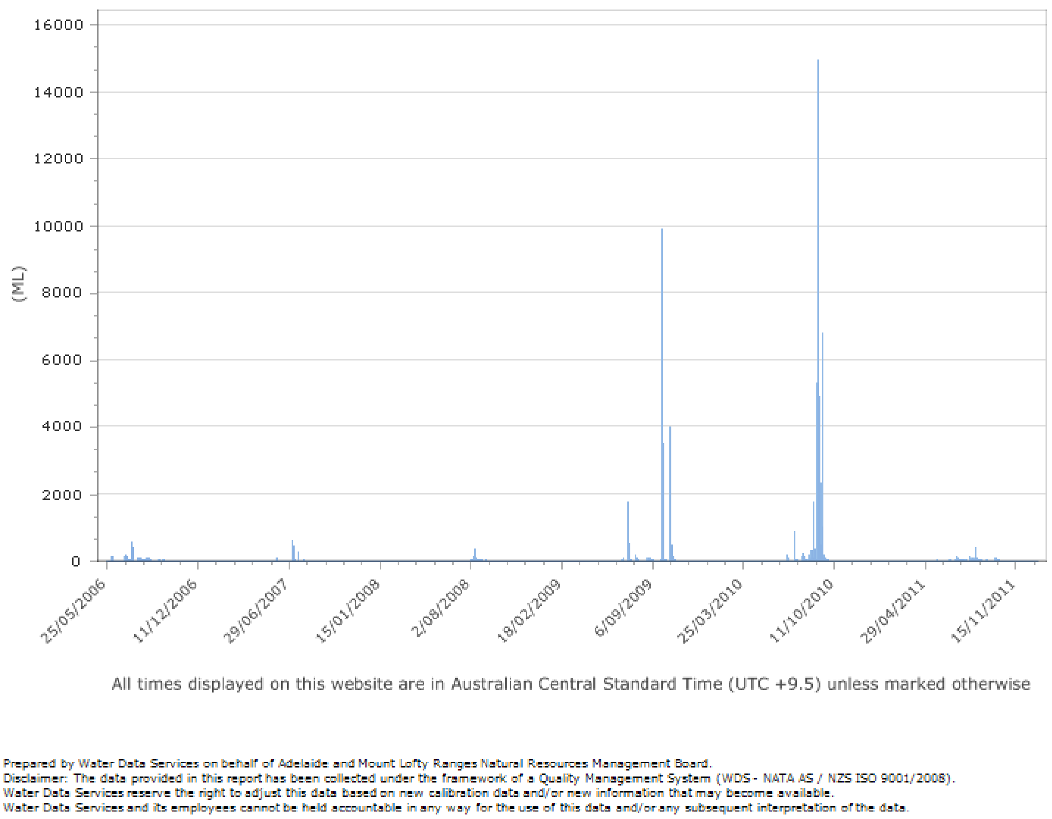


Figure 3-9 Monthly flow volumes (ML) for 2006-2011 gauged at the Onkaparinga River Old Noarlunga (A5031005) station (AMLR NRMB 2016)

Modelled and measured annual flow volumes were relatively close with a mean difference of 5% for Torrens, 7% for Onkaparinga and 12% for Gawler. However during the wet year of 2010, the modelled annual flow for the Gawler River was 21,063 ML or 33% less than that measured (Figure 3-10). The Gawler River catchment hydrological calibration was from 1/01/1972 to 31/12/1994; validation from 01/01/1996 to 31/12/2003 (record finished mid-2004). Several calibration methods were tried but the validation results were always very poor. Calculated runoff coefficients between the periods indicated a change between validation periods suggesting that the gauge record was unreliable for the latter period. Gauging station A5050510 was downstream of A5050505 and calibration was not performed at this gauge due to insufficient flow records. The hydrological parameters of A5050505 were assigned to the downstream catchment, however this site may be affected by upstream flows e.g. from dam spills or backwater effects. No data were available to indicate whether a large spill occurred in 2010. As the EMC/DWC model generates constituents using concentrations that are static in time, this large difference between measured and modelled flow volumes for the Gawler River translates into higher uncertainty of modelled loads.

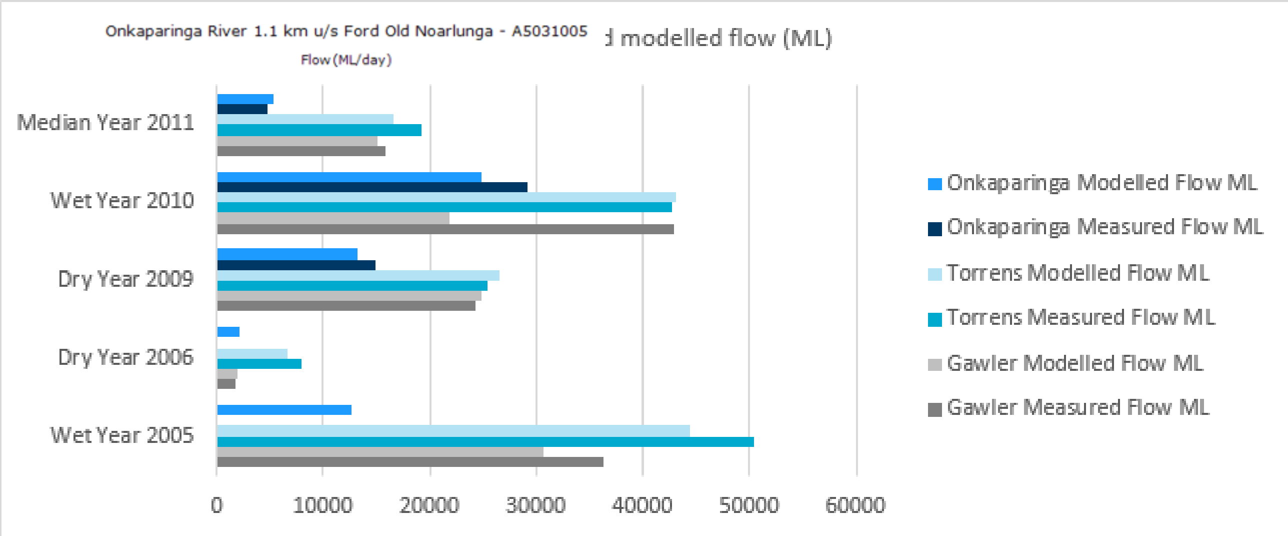


Figure 3-10 Measured and modelled annual flow volumes at outflow of Gawler, Torrens and Onkaparinga River catchments

3.4.1 Gawler River loads comparison

Measured TSS loads for the Gawler River were only available for 2009, 2010 and 2011 and for the dry and median rainfall years (2009, 2011). Modelled loads indicated that the Fleming scenario tended to overestimate TSS loads in dry and median years but were in agreeance with measured loads and results generated using the Fletcher scenario results in wet years. The Fletcher scenario reflected measured loads in wet and dry years but underestimated the median year compared to the other results. The Gonzalez scenario modelled substantially lower loads for wet years but reflected measured loads well in dry and median years (Figure 3-11a).

The Fletcher scenario underestimated TN loads across years compared to other results while use of Fleming and Gonzalez results well reflected measured loads for dry and median years. All model scenarios estimated substantially lower TN loads in the wet year compared to measured load (Figure 3-11b).

Similarly, the Fletcher scenario underestimated TP loads across years compared to other results, while measured loads for dry and median years were reflected well when Fleming and Gonzalez results were used. All three model scenarios estimated substantially lower TP loads in the wet year compared to the measured load (Figure 3-11c).

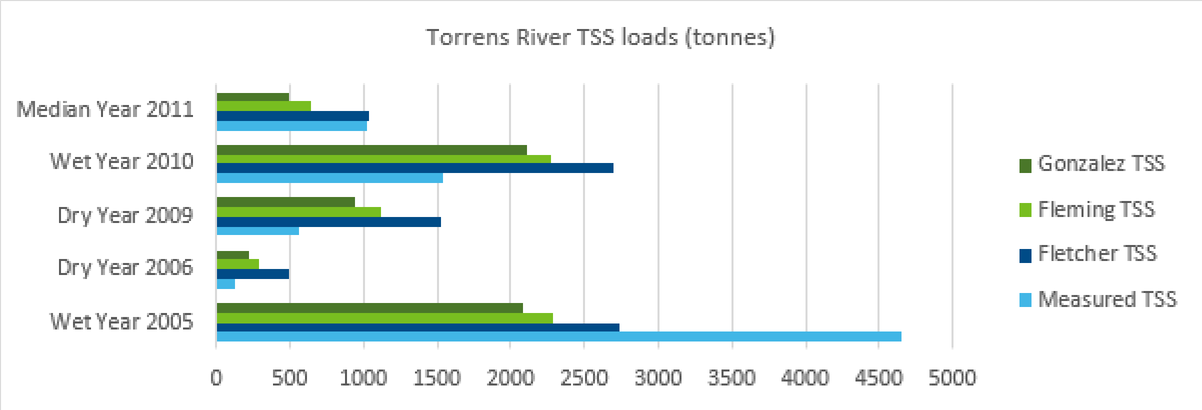
The general underestimation of TN and TP loads (cf. measured) for the wet year of 2010 is probably due to the much lower modelled flow (cf. measured) (Figure 3-10). This also suggests that modelled TSS loads may be overestimated in wet years by Fletcher and Fleming scenarios. The higher modelled loads in dry 2009 cf. median 2011 may be due to difference in proportional volumes of quick and slow flow (to which EMC and DWC values are applied respectively) occurring in in these years however this requires further analysis of model outputs.

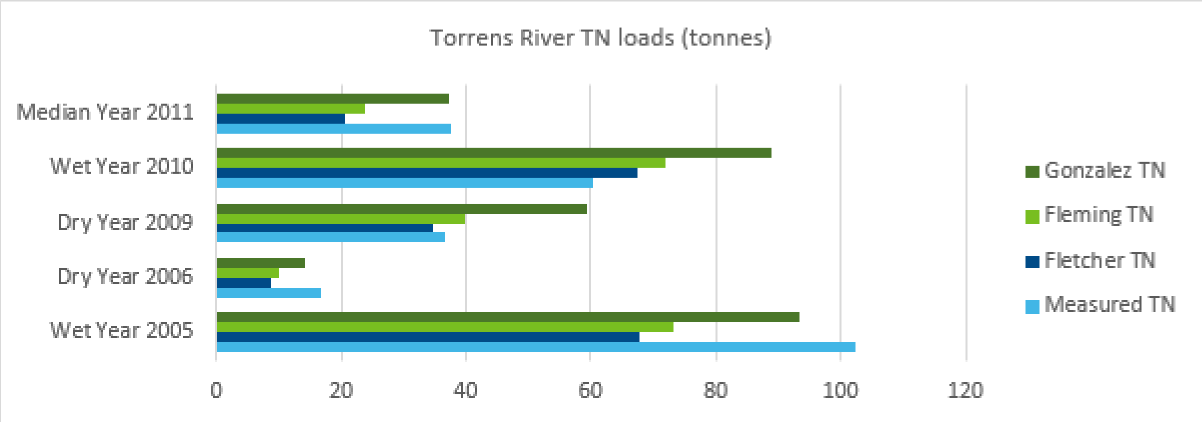
Figure 3-11 Measured and modelled (a) totals suspended solids, (b) total nitrogen and (c) total phosphorus loads for Gawler River

3.4.2 Torrens River loads comparison

With the exception of 2005, there was a tendency for all scenarios to model higher TSS loads in dry and wet years (cf. measured loads) (Figure 3-12a). Fleming and Gonzalez scenarios modelled lower loads in the median year while Fletcher results reflected measured load. The substantially lower modelled loads in 2005 cf. measured load were not driven by differences in modelled and measured flows (Figure 3-10).

Based on Torrens River at Holbrooks Road gauging station data, seasonality of flows in 2005 were different to 2010 in that the bulk of flows came in Oct and Nov after a relatively dry winter and particularly dry Sep whereas flows for 2010 occurred mostly in Aug, Sep and Oct. This indicates more catchment sediment build up may have occurred during the dry lead up period in 2005 and that rainfall intensity may have been higher during Oct and Nov. This could explain the higher loads measured in 2005 even though total annual flow (and rainfall) was comparable to 2010. One of the limitations of the EMC/DWC approach, due to the effect of averaging, is that inter-annual and seasonal rainfall intensity/flow dynamic variability, or the effect of sediment build up during extended dry periods and subsequent wash off, cannot be sufficiently represented.





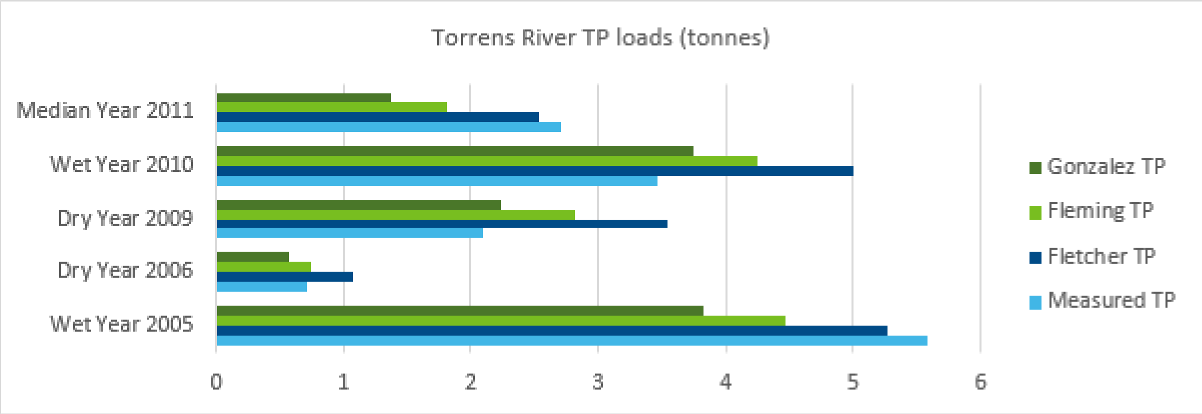


Figure 3-12 Measured and modelled (a) total suspended sediment, (b) total nitrogen and (c) total phosphorus loads for Torrens River

All three model scenarios generally reflected TN and TP loads for all years (Figure 3-12b, Figure 3-12c). The Gonzalez scenario resulted in highest modelled TN across years with little difference between Fletcher and Fleming scenarios. The Fletcher scenario resulted in highest modelled TP across years while the Gonzalez scenario was lowest and Fleming results somewhere in between.

Large differences between modelled and measured TN and TP loads were not seen for 2005 as they were for TSS. As noted for the Gawler modelled loads, higher modelled loads in the dry 2009 year cf. the median 2011 year (and wet 2010 year in the case of Fletcher and Gonzalez scenario TN and TP loads) may be due to difference in proportional volumes of quick and slow flow (to which EMC and DWC values are applied respectively) occurring in these years however this requires further analysis of model outputs.

3.4.3 Onkaparinga River loads comparison

No flow records were available for the A5031005 gauging station prior to the site opening in mid-2006 and water quality data were not sampled for a full year prior to 2009. For the years with available flow records, modelled flows were within 7-8% of measured volumes (Figure 3-10). Only 2010 and 2011 contained measured annual loads for TSS, TN and TP so comparison with modelled loads was limited. In the wet year of 2010, modelled TSS loads from the Fletcher and Fleming scenarios were substantially lower than measured and Gonzalez scenario loads (Figure 3-13a). Gonzalez scenario load were still only about half of the measured load for 2010. The modelled loads for the median year of 2011 were about double the measured loads for the Fletcher and Fleming scenarios and about half for the Gonzalez scenario.

A similar pattern was seen for modelled TN and TP loads in the wet year of 2010 where modelled loads were substantially lower than measured load and this difference was greatest for the Fletcher and Fleming results (Figure 3-13b, Figure 3-13c). Modelled and measured TN loads for 2011 closely agreed while Fletcher and Fleming TP loads were higher than the Gonzalez and measured loads.

Storm event flow (or ‘quick flow’ to which EMC values are applied in SOURCE) is more likely to occur in wet years so the lower modelled loads for TSS, TN and TP in the wet year of 2010 compared to measured values suggests that EMC values may be underestimated, particularly in the Fletcher and Fleming models and to a lesser degree in the Gonzalez model. The overall differences between measured and modelled results for 2010 suggests the inflow node (which accounts for 69% of the total catchment flow in 2010) configuration for TSS, TN and TP may be underestimated. During dry and median years where more base flow (or slow flow to which DWC values are applied in SOURCE) could be expected, results suggest DWC values may be overestimated for the Fletcher and Fleming scenarios and about right or slightly underestimated for the Gonzalez scenario.

As noted for the Gawler and Torrens modelled loads, higher modelled loads in the dry 2009 year cf. the median 2011 year were observed and may be due to difference in proportional volumes of quick and slow flow (to which EMC and DWC values are applied respectively) occurring in in these years however this requires further analysis of model outputs.

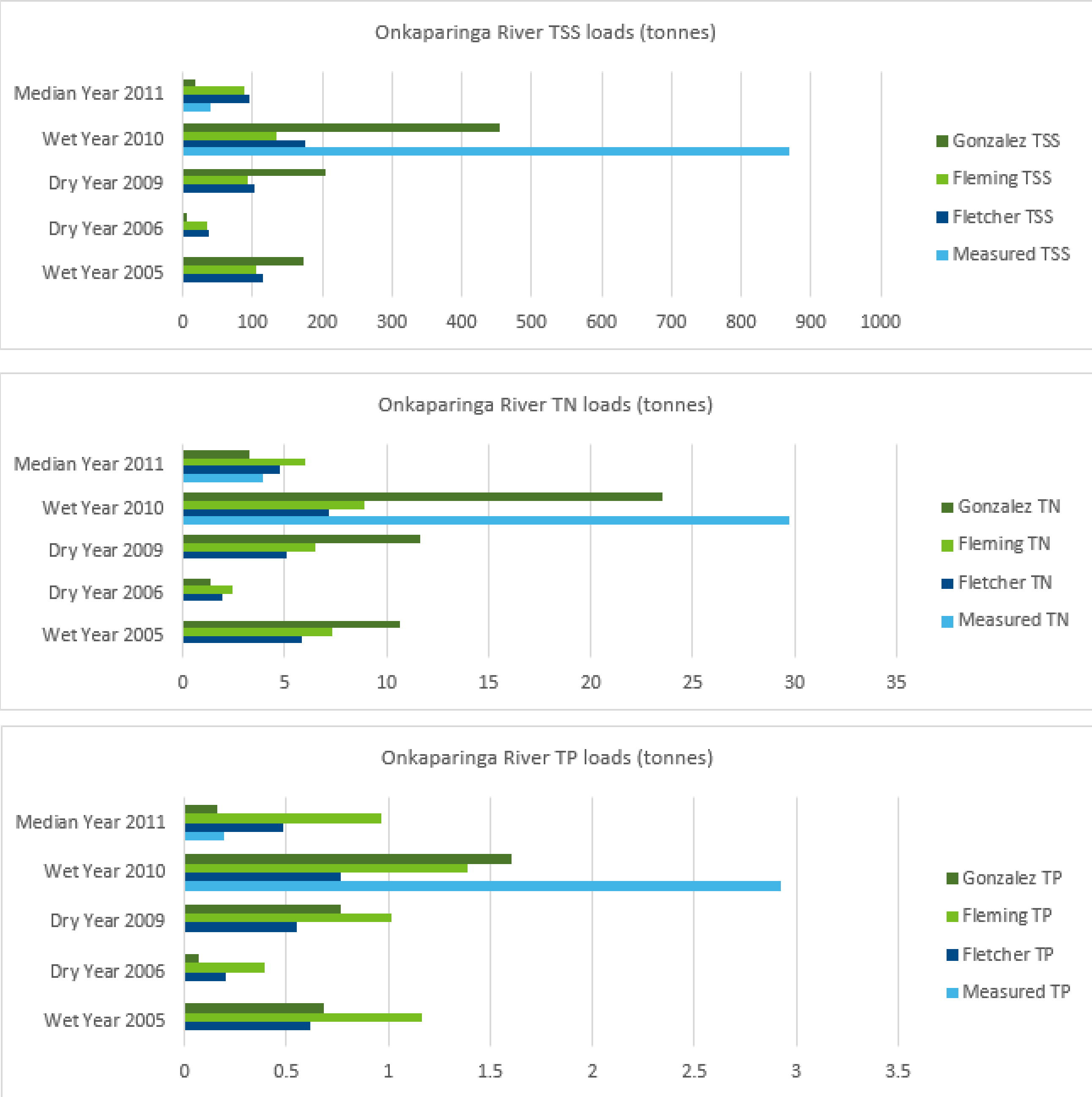


Figure 3-13 Measured and modelled (a) total suspended sediment, (b) total nitrogen and (c) total phosphorus loads for Onkaparinga River

3.5 Use of ICUWM model to identify high TSS yielding sub-catchments

In addition to generating input data for the AREMp, the ICUWM model was used to model mean annual and mean annual areal TSS, TP and TN loads over the 2003–2013 time period at the sub-catchment scale. The mean annual areal values represent constituent discharge from a hectare of the sub-catchment being considered. The mean annual TSS distribution and mean annual areal distribution for TSS are shown in Figure 3-14 and Figure 3-15 respectively. Mean annual areal distribution is useful when comparing discharges and constituents from different catchments because it allows comparing flows and constituents discharging from a unit area of a catchment.

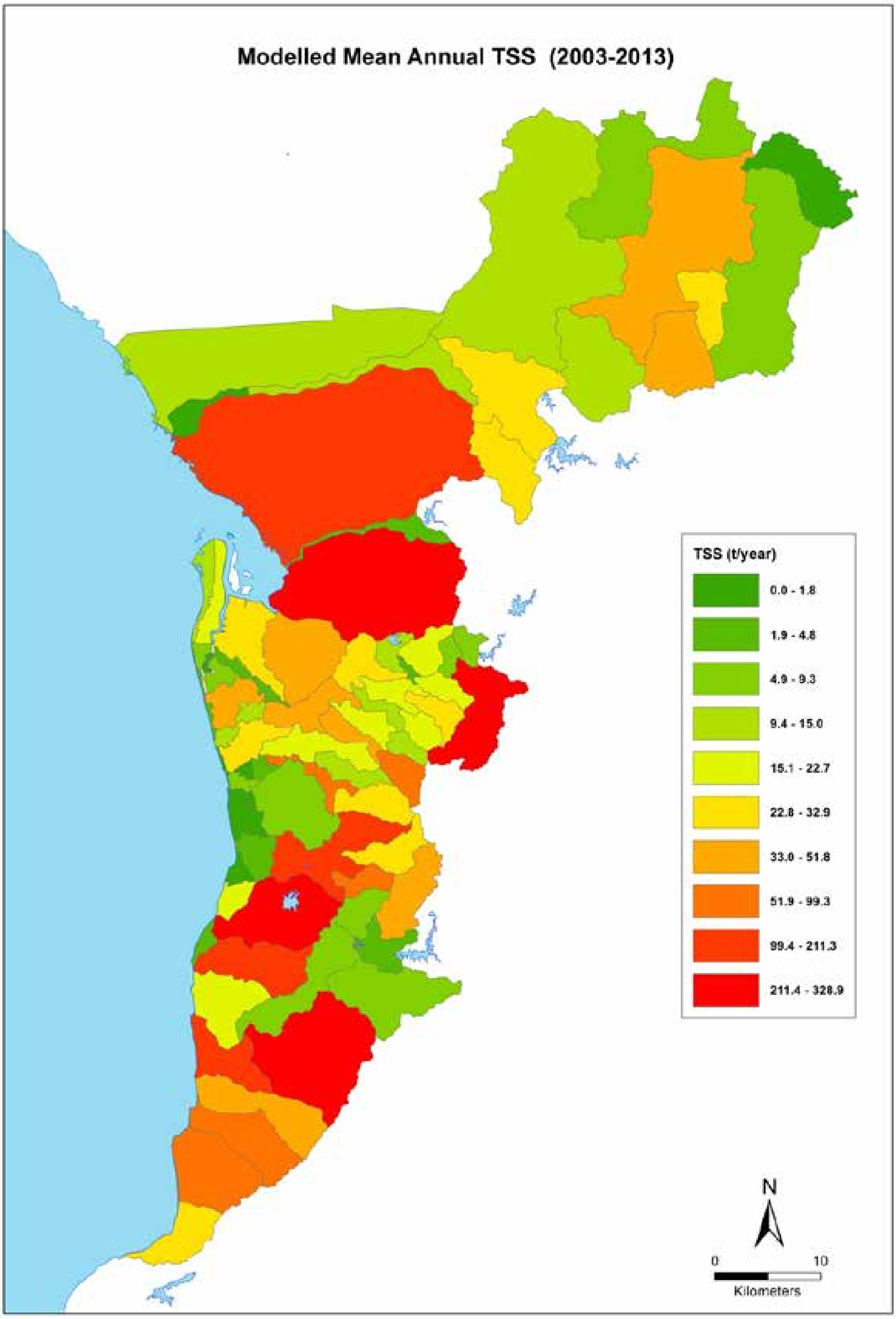


Figure 3-14 Modelled mean annual TSS (tonnes/year) distribution

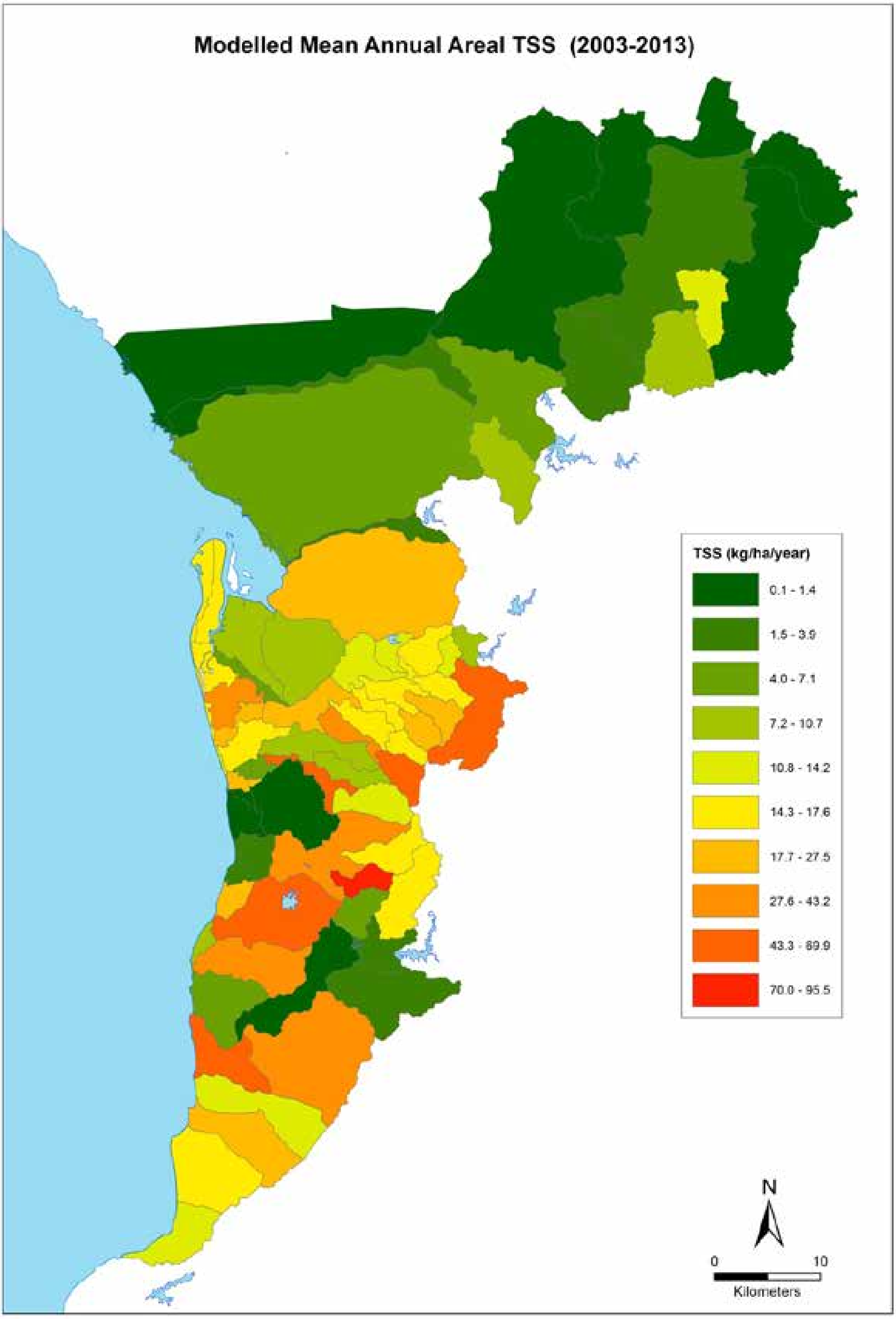


Figure 3-15 Modelled mean annual areal TSS (kg/ha/year) discharging from sub-catchments

Results indicated that the annual areal TSS loads discharging from sub-catchments associated with Pedler Creek, Christie Creek, Field River, Patawalonga Basin, Torrens River, and Dry and Cobbler Creeks range from 18 kg/ha/year to 96 kg/ha/year. Sub-catchments associated with Pedler Creek, Christie Creek, Field River and Patawalonga Basin are in the higher end of this range (i.e. 28–96 kg/ha/yr) whereas Torrens and Dry and Cobbler Creek are in the lower end of this range (11–43 kg/ha/yr).

If seagrass is sensitive to sediments associated with stormwater discharges, catchments to be targeted for reducing sediments include Pedler Creek, Christie Creek, Field River, Patawalonga Basin, Torrens River and Dry & Cobbler Creek.

Of the potential stormwater harvesting schemes considered in this project (based on Wallbridge and Gilbert (2009), ref Table 2-9), the majority (~80%) are located in sub-catchments that are likely to contribute sediment loads at the higher end of the range (18–96 kg/ha/year). The other 20% are located in the Gawler catchment which discharges sediment at the lower end of the range. Based on the results of this project, intervention should be focussed on the sub-catchments associated with Pedler Creek, Christie Creek, Field River, Patawalonga Basin, Torrens River and Dry & Cobbler Creeks, rather than the sub-catchments associated with Gawler River.

3.6 Thresholds of coastal impact and Hotspots

Two approaches were used to provide guidance regarding the water quality targets stormwater interventions should seek to deliver in the coastal waters. The first comprised modelling of historic land-based discharges and their subsequent correlation to changes in the extent of seagrass loss described in §3.6.1, and the second used the AREMp model to identify hotspots of impact where water quality currently fails to meet the requirements of healthy seagrass as a result of land-based discharges, described in §3.6.2.

3.6.1 Thresholds from historical data

The ASLs for 1940, representing the situation preceding major losses, indicate that maximum loads experienced along the central coast were less than 1 tonne nitrogen/km^2 and 20 tonnes suspended solids/km^2 (Figure 3-16 top). The extent of bare sand nearshore was not clearly correlated with load contours, and the location of the blue line of seagrass at the time was likely to have been determined by other factors such as coast geomorphology and wave exposure. Some meadow fragmentation was observed offshore of Glenelg north, but since these seagrass maps were generated from aerial imagery acquired in 1949 (i.e. after the Glenelg outfall started operation in 1943), any offshore loss might have been a consequence of this new input (excluded from the 1940 ASLs).

The ASLs for 1975, representing the period of significant seagrass loss along the central Adelaide coast, indicate much higher nitrogen load contours than in the 1940s, reaching up to 5 tonnes/km^2 near the mouth of the Torrens River, the Patawalonga lake system, and the Glenelg outfall (Figure 3-16 bottom). SS inputs remained below 20 tonnes/km^2 throughout the area, comparable to the 1940s. The areas of loss were mostly constrained within the 1.5 tonnes/km^2 nitrogen contour. The frequency histogram obtained from plotting the number of pixels where loss occurred versus the nitrogen load experienced at each pixel, corroborate the idea of a threshold for loss around 1.5–1.6 tonnes/km^2 (Figure 3-17). No clear correlation with suspended solids load contours was observed.

Bare sand extent and seagrass loss layers were sourced from the Department of Environment, Water and Natural Resources (Hart 1997).

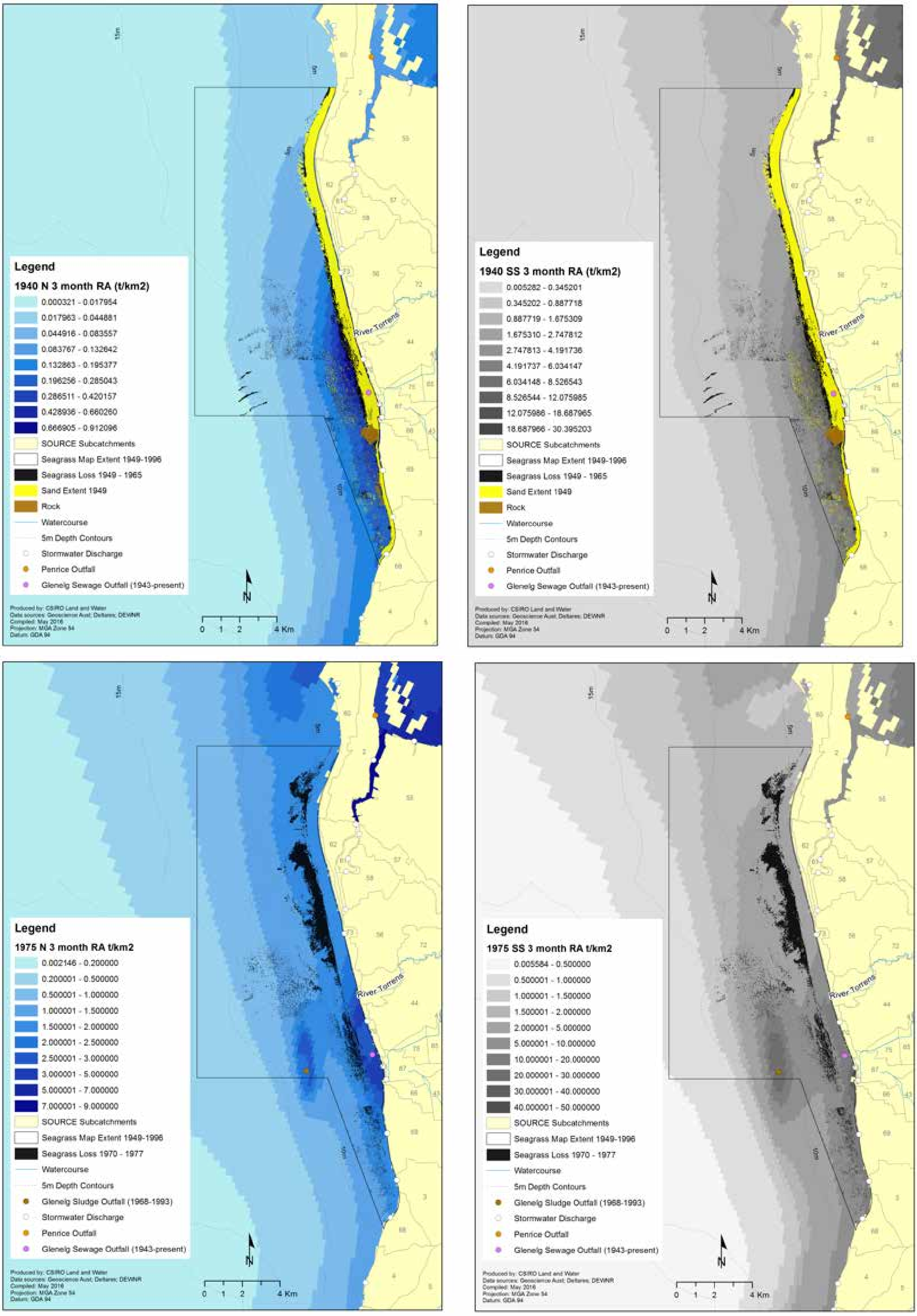


Figure 3-16 Nitrogen (left) and suspended solids (right) ASLs in the 1940s (top) and 1975 (bottom) in tonnes/km^2. Values shown are the 3-month running average (3mRA), chosen as indicative of pressure to the more sensitive species. Different scales have been used for 1940s and 1975 to enable their interpretation

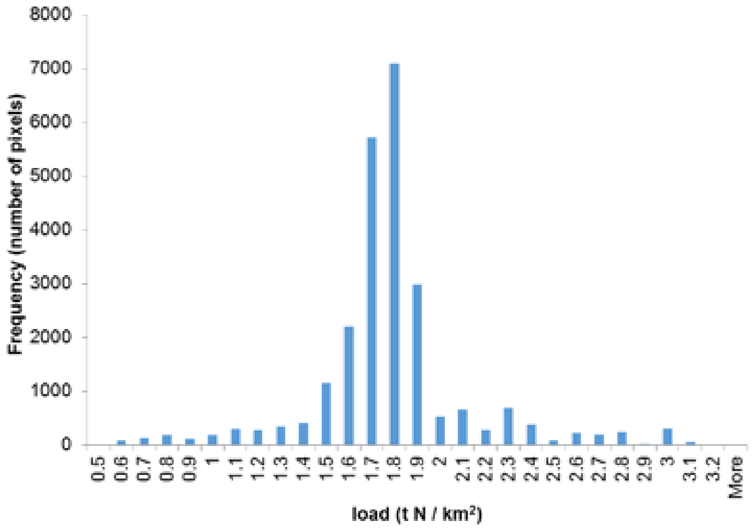


Figure 3-17 Number of 20m pixels where seagrass loss occurred in 1975 versus nitrogen load modelled for each pixel

The evolution of loads over time is crucial for interpretation of historical loss. The suspended solids load to the central coast increased by almost 4 times between 1940 and the mid-1960s, while the nitrogen load increased over 10 times (Figure 3-18). This increase in loads however did not lead to widespread seagrass loss until the period 1970–1977.The total suspended solids load decreased between 1963 and 1975 as a result of the completion of Kangaroo Creek reservoir on the Torrens River in 1968, which acted to significantly decrease river discharge from daily peaks of 8000 ML to less than 1000 ML (Figure 3-19, Wilkinson et al. 2005b). It is therefore unlikely that losses observed in 1970-1977 are a consequence of direct stormwater inputs of suspended solids.

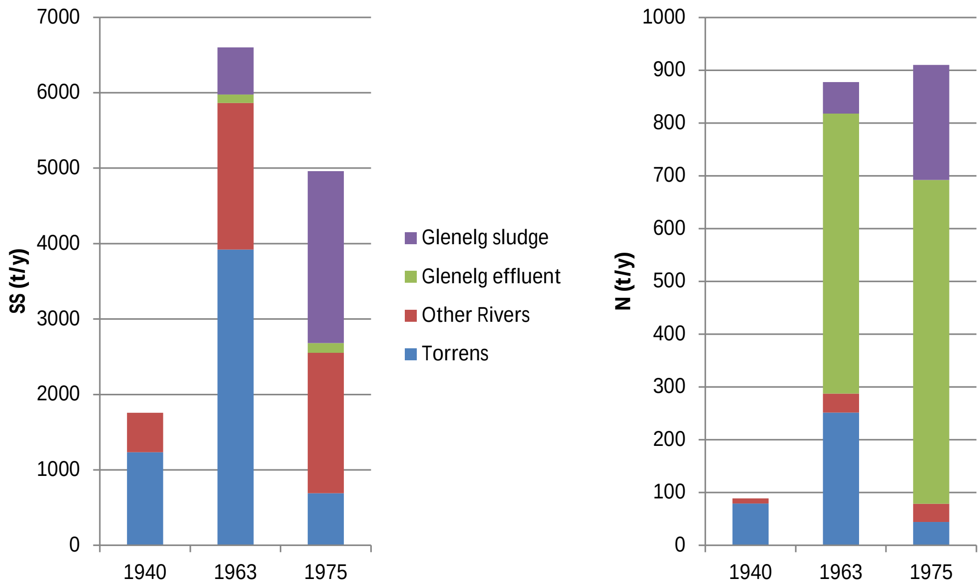


Figure 3-18 Comparison of total loads to the Adelaide central coast per source in 1940, 1963 and 1975 (raw data compiled from Wilkinson et al. (2005b)). 1940 was chosen as representative of a near pristine scenario, 1963 for peak stormwater inputs, and 1975 for peak seagrass loss

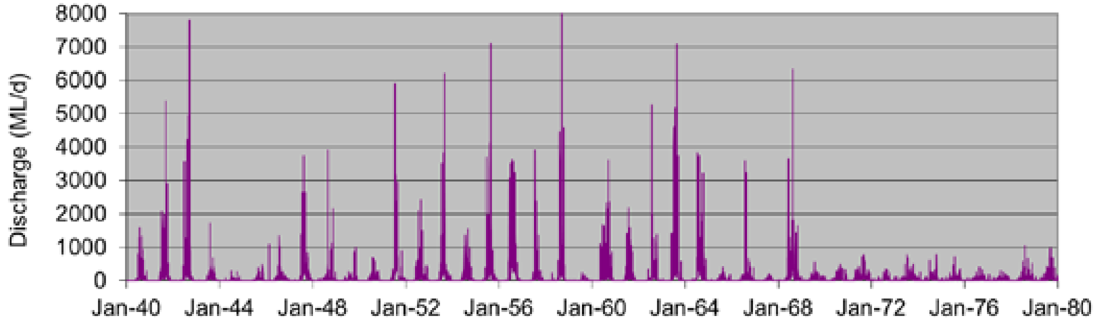


Figure 3-19 Torrens River discharge between 1940 and 1980

In contrast to suspended solids, the nitrogen load continued to increase after the 1960s as a result of higher discharges from the Glenelg WWTP, the most marked change being a steep increase in sludge discharge from about 5 ML/month in the 1960s to >20 ML/month after the mid-1970s (Figure 3-20). The threshold for seagrass loss appears thus tempered by the nature of the discharge, with nitrogen delivered in organic form within sludge likely having a larger detrimental effect on seagrasses than the discharge of dissolved nutrients. The threshold of 1.6 tonnes/km^2 observed here should thus be considered a worst case scenario, and the actual threshold for seagrass loss could be higher if nitrogen discharges are mainly delivered in dissolved form.

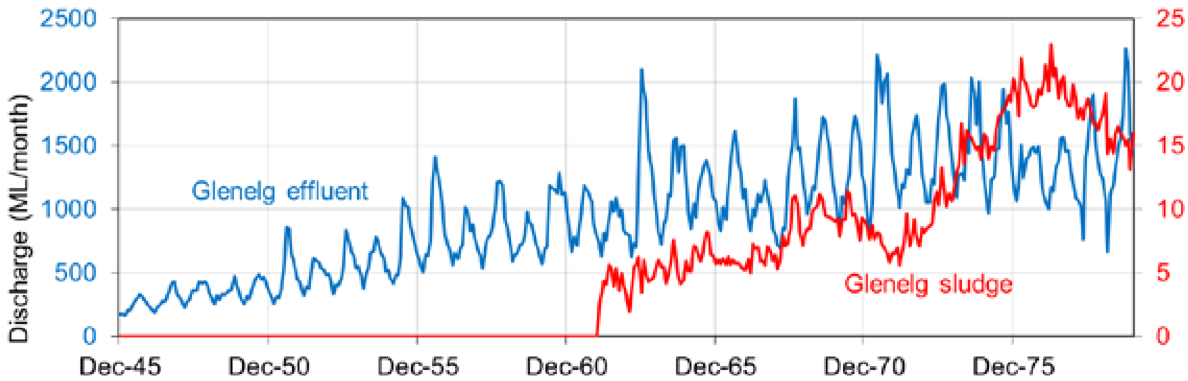


Figure 3-20 Glenelg WWTP discharge between 1945 and 1980

The limitations of the AREMp (e.g. spatial resolution) also affects this number, which should be treated with caution. This scoping work however demonstrated the power of running hindcast simulations to derive thresholds for loss. Given the close proximity of wastewater and stormwater sources along the Adelaide coast, the driver for loss could change between periods and the analysis of other periods would likely shed further light on the role of stormwater inputs on seagrass loss.

3.6.2 Hotspots for the current situation

The simulations to determine hotspots of impact along the coast for the current situation were based on the year 2011. This year was chosen as representative of an average year, with the annual rainfall of 538 mm falling in the median range observed for the decade of 1995-2004 (Jones 2015). The total loads of suspended solids in 2011 amounted to:

- 3200 tonnes from rivers
- 2000 tonnes from WWTPs
- 10,300 tonnes from Penrice.

The total loads of nitrogen were:

- 125 tonnes from rivers
- 605 tonnes from WWTPs
- 1114 tonnes from Penrice.

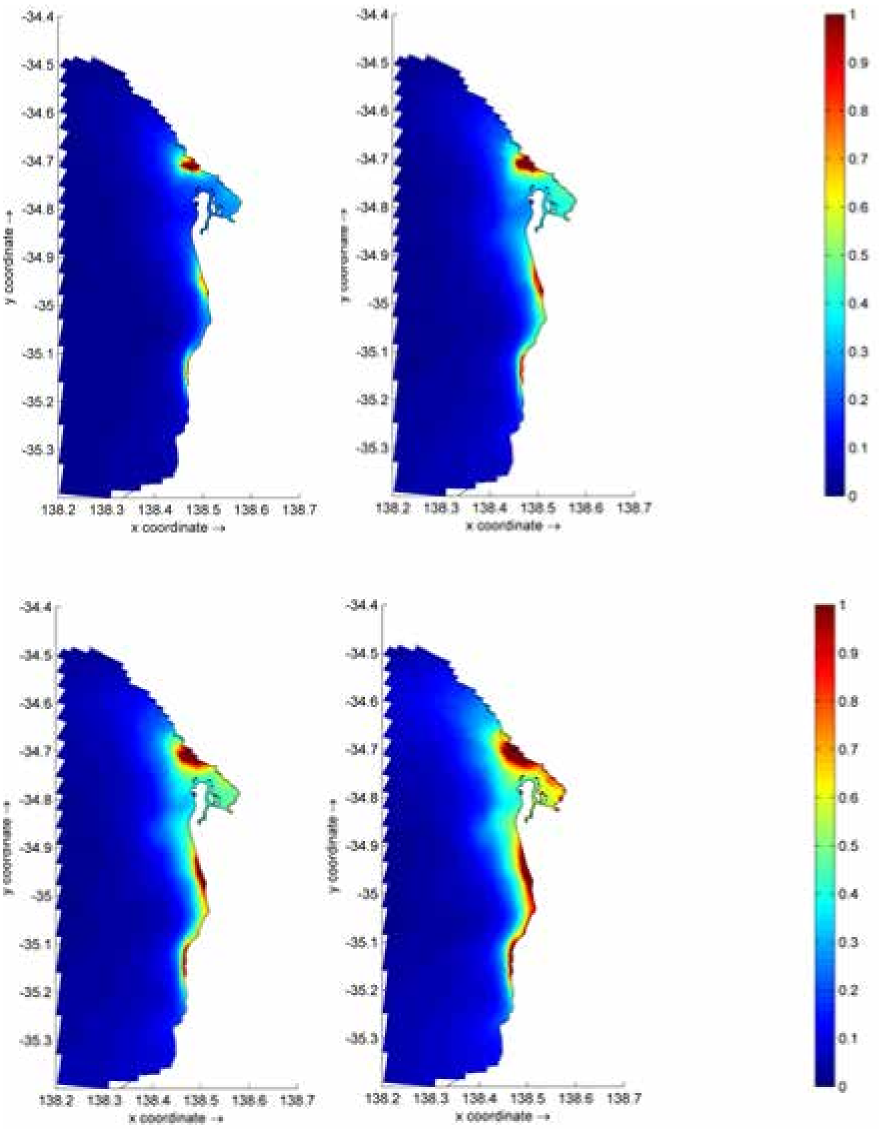


Figure 3-21 Nitrogen ASLs (tonnes/km²) from all rivers and WWTPs, excluding the Penrice discharge; mean values (top left), six-month running average (6mRA) values (top right), three-month running average (3mRA) values (bottom left) and one-month running values (1mRA) values (bottom right)

The ASLs for nitrogen and suspended solids are summarised in Figure 3-21 and Figure 3-22, where discharges from Penrice were excluded as this source has ceased discharging to the Adelaide coast since July 2013 (ASLs including this discharge are shown in Appendix D). The results are provided as four figures per parameter:

- annual mean ASL
- maximum value of a 6 month rolling average (6mRA) of the ASL
- maximum value of a 3 month rolling average (3mRA) of the ASL
- maximum value of a 1 month rolling average (1mRA) of the ASL.

The 3mRA and 6mRA ASLs are considered of particular importance given these time periods represent the upper limits of tolerance for low light levels of seagrass species *Amphibolis spp* and *Posidonia spp* respectively. In addition, Figure 3-23 shows four similar graphs for the Overall Impact Indicators (OIIs) using a discrete colour scale (below 1 to above 1). These indicate four primary impact hotspots: Christies/Onkaparinga, Torrens/Glenelg/Patawalonga, Barker Inlet, and Bolivar/Gawler. The Christies/Onkaparinga and Bolivar/Gawler hotspots are driven by nitrogen loads, while the Barker Inlet hotspot is driven by SS loads. The Torrens/Glenelg/Patawalonga hotspot is driven by both nitrogen and SS.

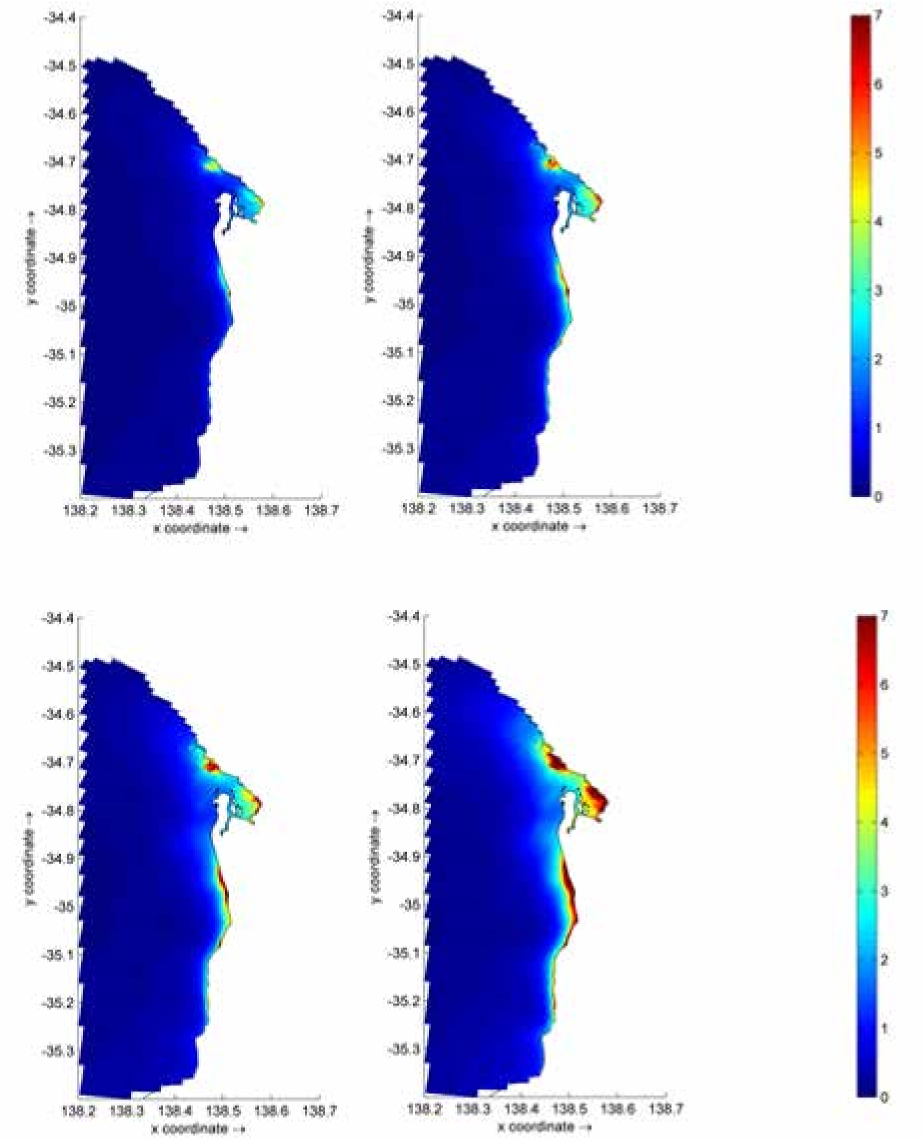


Figure 3-22 Suspended solids ASLs (tonnes/km^2) from all rivers and WWTPs, excluding the Penrice discharge; mean values (top left), six-month running average (6mRA) values (top right), three-month running average (3mRA) values (bottom left) and one-month running average (1mRA) values (bottom right)

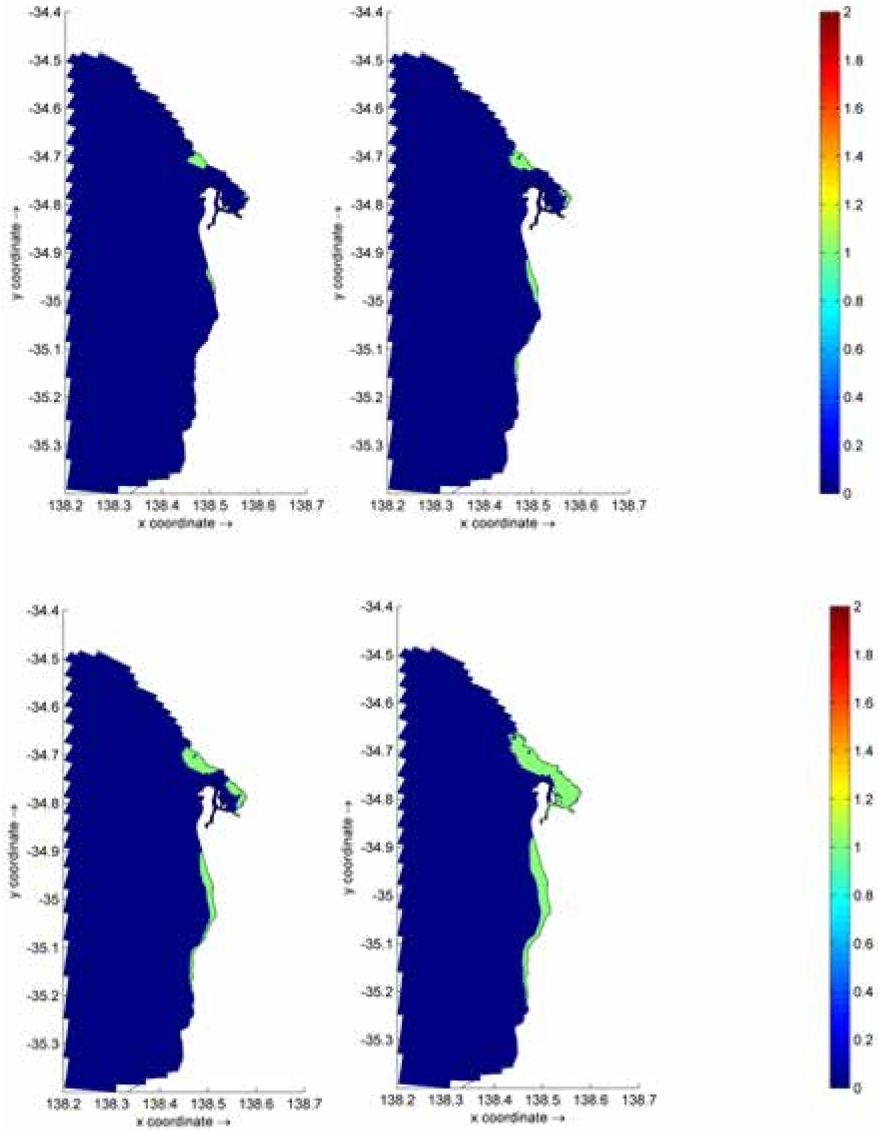


Figure 3-23 Overall Impact Indicators from all rivers and WWTPs; mean values (top left), six-month running average (6mRA) values (top right), three-month running average (3mRA) values (bottom left) and one-month running averages (1mRA) values (bottom right)

The time variability of nitrogen and SS concentrations at selected locations (Bolivar/Gawler, Torrens/Glenelg/Patawalonga, and Christies/Onkaparinga) was used to infer the time variability of ASLs as daily averages, 1 month, 3 months and 6 months running averages (Figure 3-24 to Figure 3-27). Although concentrations cannot be directly compared to ASLs, the time variability of the concentrations is presumed proportional to the time variability of ASLs, and the former is used to estimate the latter.

Temporal variability of ASLs is the result of temporal variability of loads and temporal variability of residual currents as affected by wind conditions and tides. As a result, temporal variability is stronger for rivers than for WWTPs. This also implies that the temporal variability of SS ASLs is generally stronger than that of nitrogen ASLs, since the former are dominated by rivers and the latter by WWTPs.

All hotspots showed marked seasonal variability, with concentrations peaking in late winter, generally in August (Figure 3-24 to Figure 3-27).

The Barker Inlet hotspot had a bimodal evolution of concentrations, peaking around March and again around August (Figure 3-25). Bolivar/Gawler also had a peak for SS in March (Figure 3-24). This analysis emphasises the critical role of winter loads in defining hotspots where loads are higher than recommended by the ACWS.

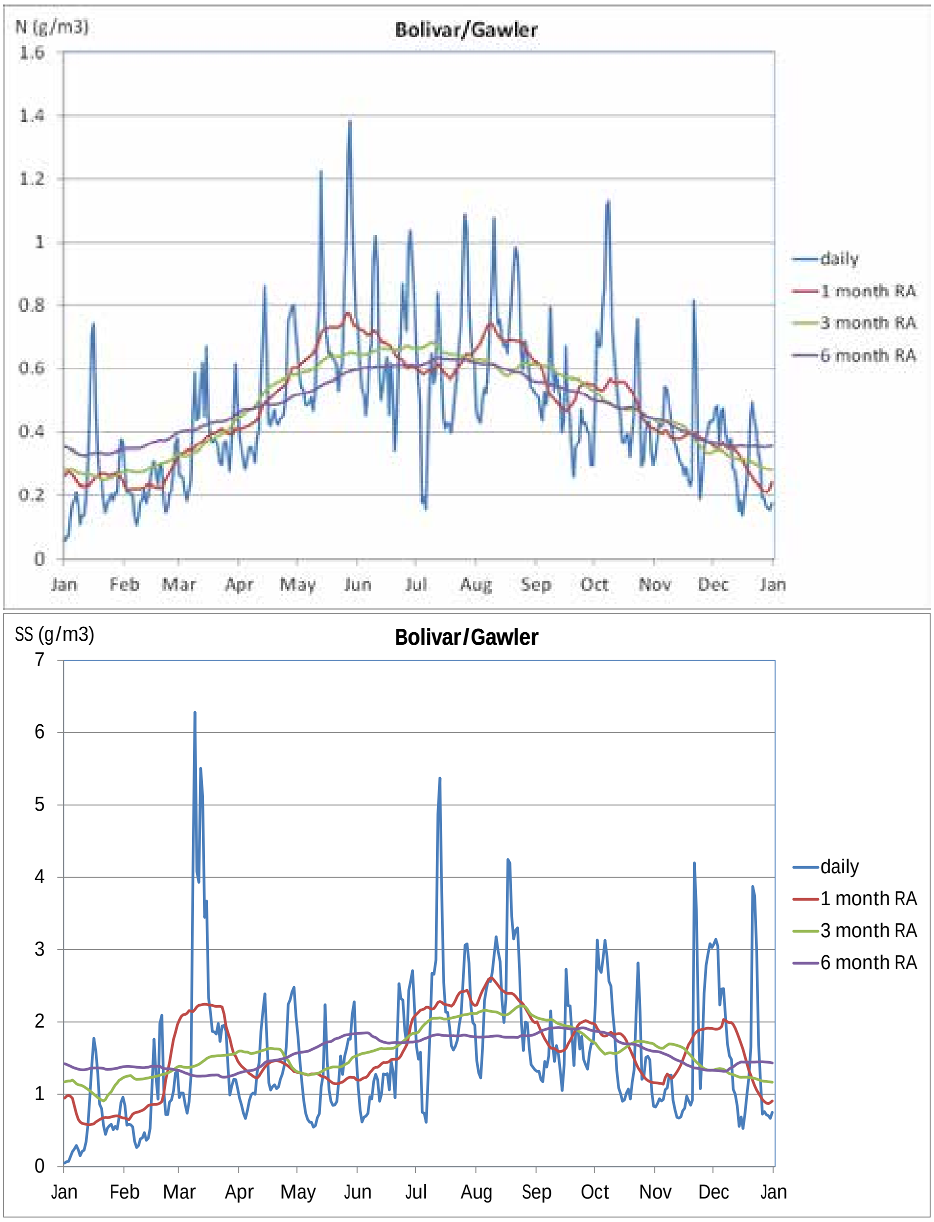
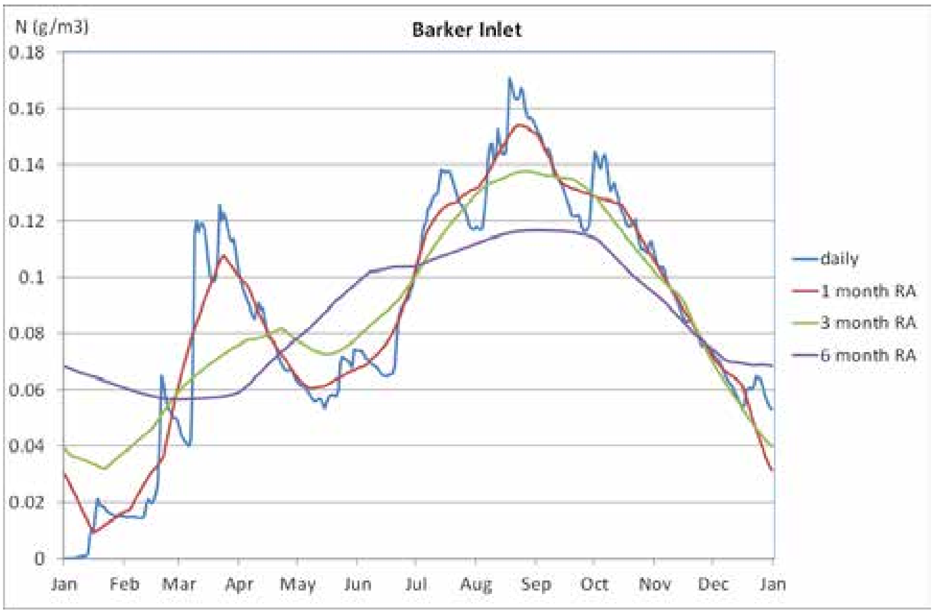


Figure 3-24 Time series output for N (g/m^3) (top) and SS (g/m^3) (bottom) at the Bolivar/Gawler station



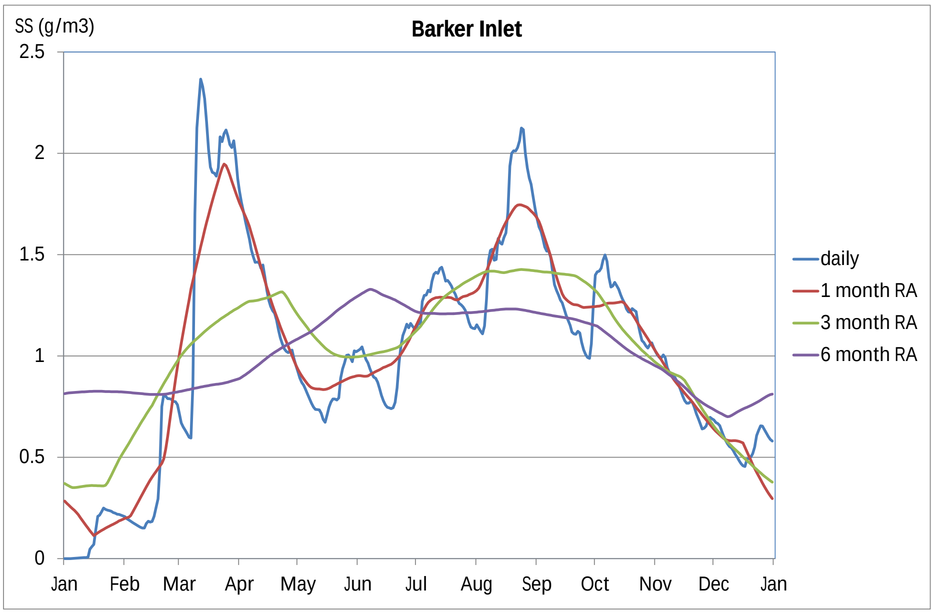
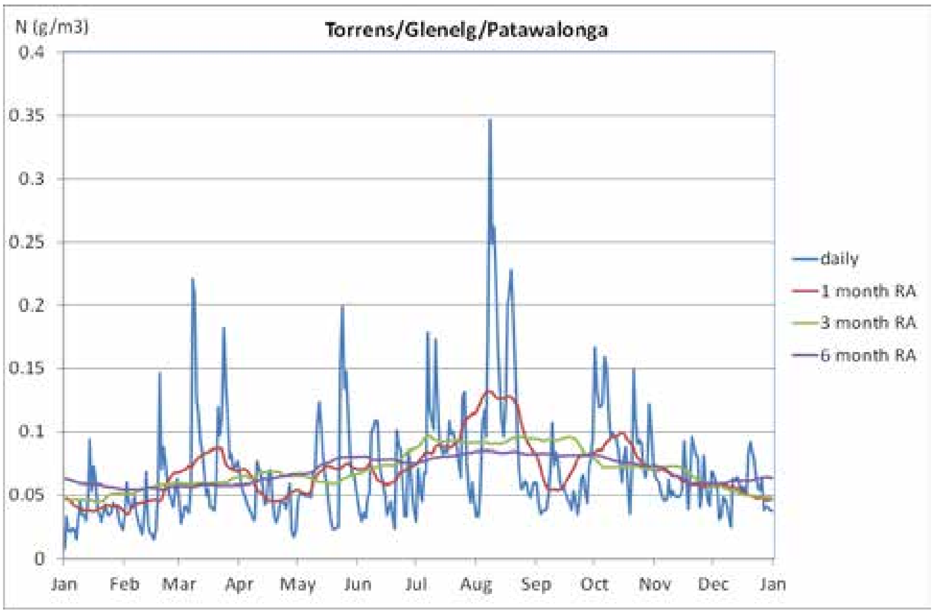


Figure 3-25 Time series output for N (g/m^3) (top) and SS (g/m^3) (bottom)at Barker Inlet station



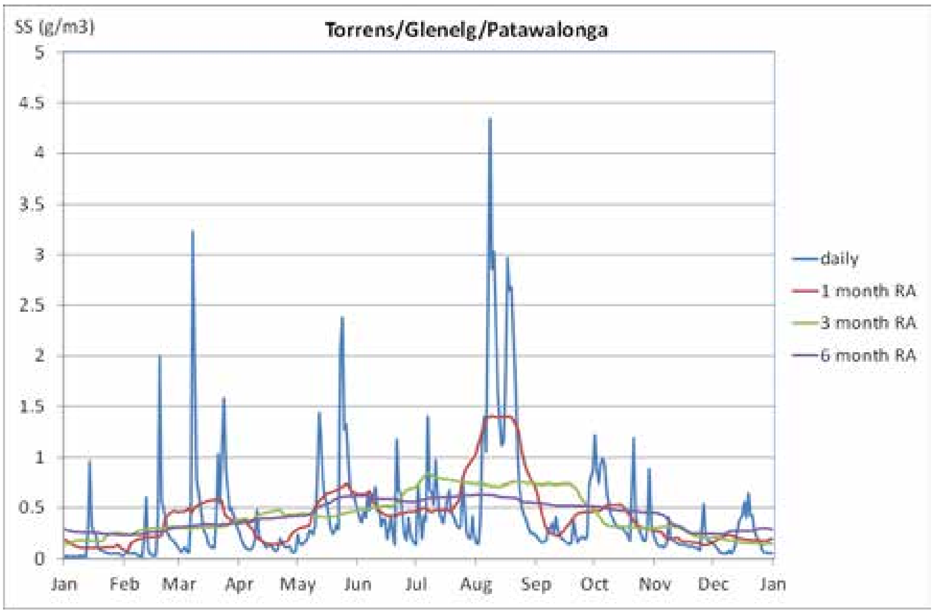


Figure 3-26 Time series output for N (g/m^3) (top) and SS (g/m^3) (bottom) at Torrens/Glenelg/Patawalonga station

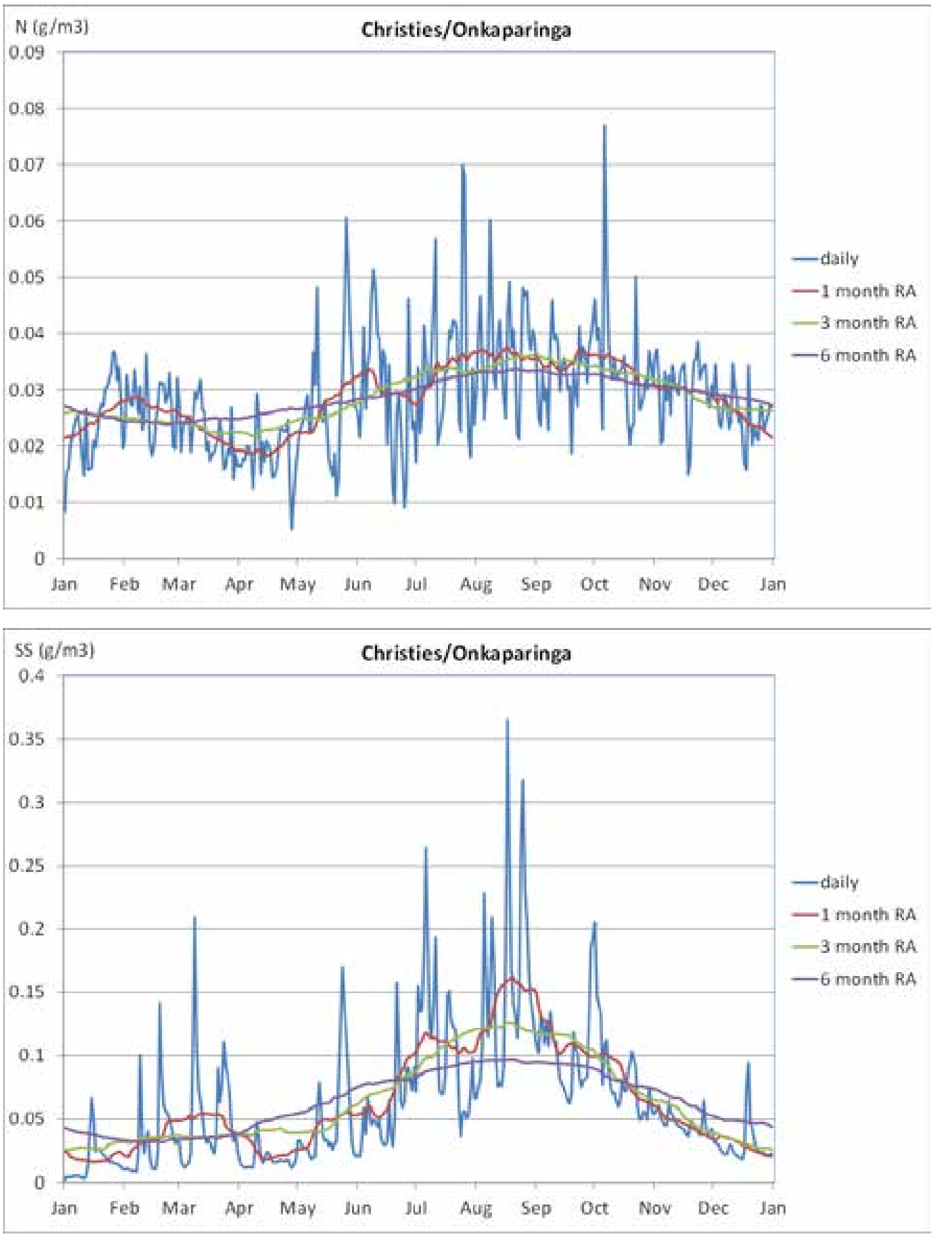


Figure 3-27 Time series output for N (g/m^3) (top) and SS (g/m^3) (bottom) at Christies/Onkaparinga station

3.7 Scenario modelling

3.7.1 Catchment model scenario results

The four scenarios run and analysed for the Torrens catchment to demonstrate methods and results for flow, TSS, TN and TP are presented in Figure 3-28. As expected, the filter model[4] did not reduce flow volumes and the effect of stormwater harvesting was the most pronounced on flows and loads. Modelling results showed stormwater harvesting at 50% of design capacity has a greater effect on total discharged loads than the filter model applied at all ‘urban’ FUs (‘urban’ FUs accounted for 6048 ha or 3% of the total Torrens catchment area).

[4] From Table 2-9, the filter model (scenario D) comprises 100% stormwater harvesting + 50% reduction in urban TP, TN, TSS

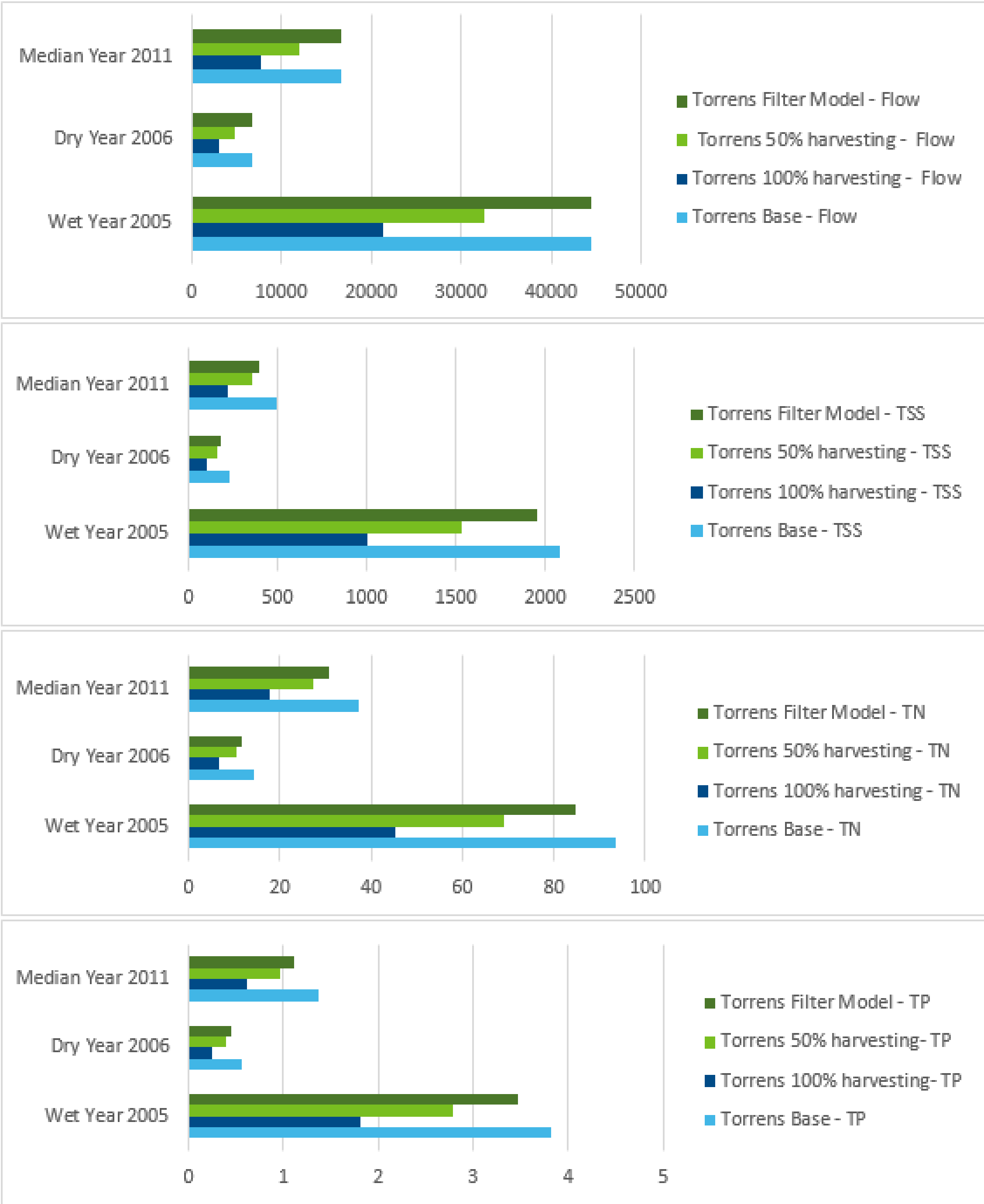


Figure 3-28 Modelled (a) flow, (b) TSS, (c) TN and (d) TP for Torrens River catchment

3.7.2 Coastal model scenario results – high level

The impact of stormwater on the coastal ecosystem was investigated through the outputs of light and habitat suitability for seagrasses in the AREMp. Simulating the average year of 2011 (rainfall 538 mm) with and without the discharge of rivers indicates that the direct input of rivers has only a marginal influence on the coastal light climate (Figure 3-31 top). As light is the main driver of habitat suitability in the region (Zijl et al. 2014), the overall impact of direct river inputs on seagrass suitability is also only marginal (Figure 3-32). These results should be taken with caution however, as the AREMp is not able to accurately represent the light climate nearshore due to factors such as limited granularity in river inputs (daily inputs as opposed to hourly inputs) and lack of coastal spatial resolution (200-300m). These shortcomings are being addressed as the full AREM is developed, with preliminary results of model calibration suggesting a significant improvement in predictive ability.

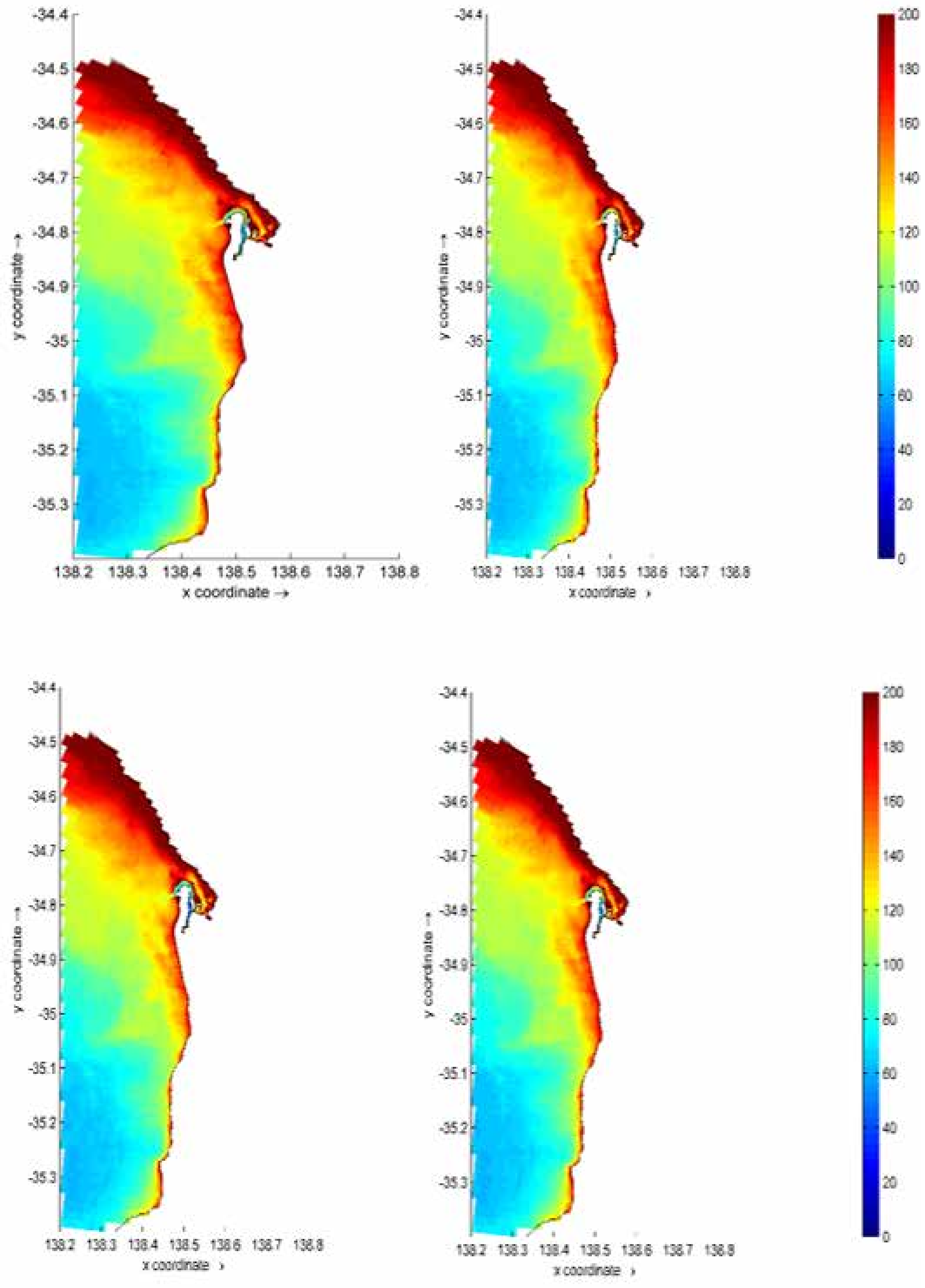


Figure 3-29 Average light intensity (W/m^2) reaching the seafloor in 2011 (top left), and in 2011 without the input of rivers (top right), in 2005 (bottom left), and 2006 (bottom right)

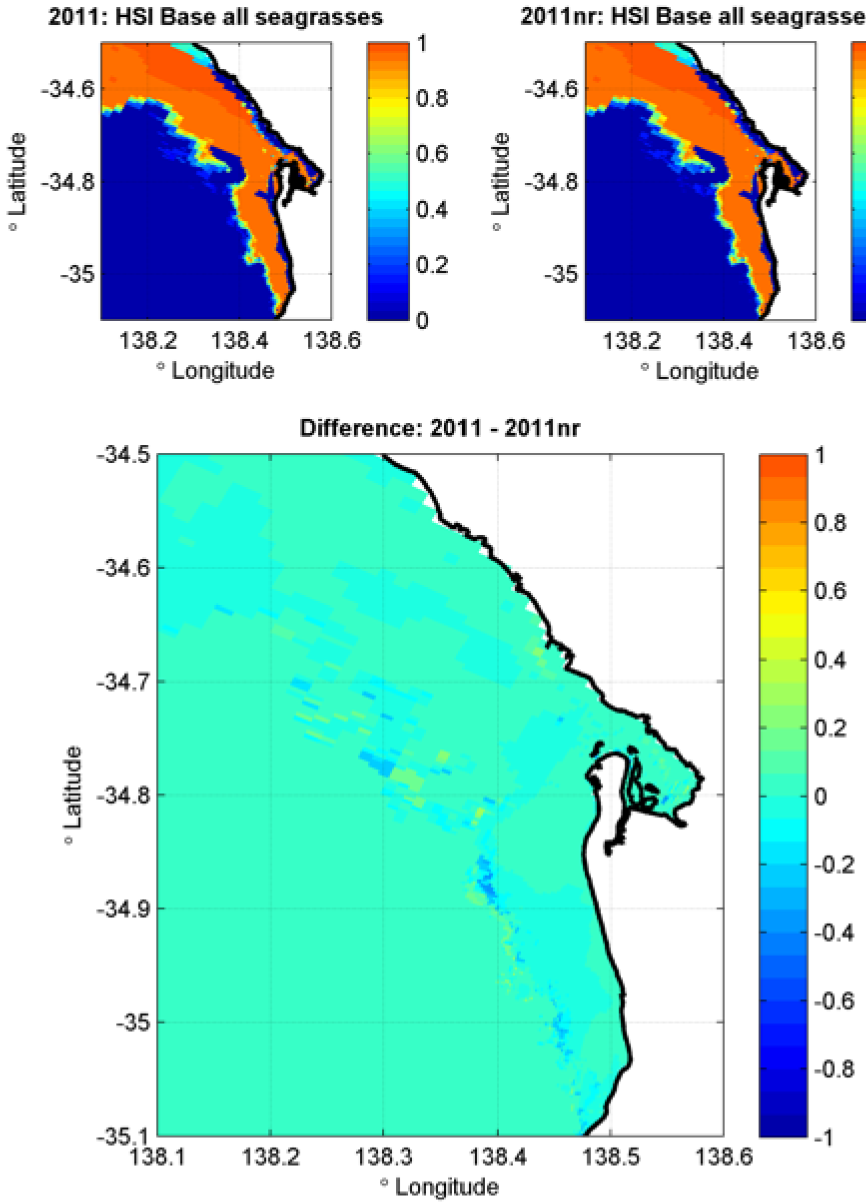


Figure 3-30 Habitat Suitability Index (HSI) for seagrasses in 2011 (top left), and in 2011 without the input of rivers (2011nr, top right), and the difference between years, with a positive value indicating higher suitability, and a negative value lower suitability, in 2005

Given this general finding, it is not surprising that ACDC modelling indicated no difference in the impact on nearshore sea grass with and without direct stormwater discharges Scenarios A and Scenario E, and also due to Scenarios A-2005, A-2011 and A-2006.

Zijl et al. (2014) have shown that light climate is significantly affected by resuspension of coastal sediments as opposed to direct river inputs. However the pool of sediments available for resuspension is obviously dependent on the input from stormwater including rivers. While the residence time of sediments cannot be determined by the AREMp, based on the estimated pool of fines in the top layer of the marine sediments and the estimated annual inputs, sediments are expected to remain trapped in the coastal zone for a significant period (likely in the order of decades). This means that historical loads of sediment will be available for resuspension, with the cumulative amount in the system at any one time varying according to annual inputs and discharges of the readily resuspendable fraction.

The contrast of a wet (2005, rainfall 630 mm) and a dry year (2006, 288 mm) (Figure 3-31), or a wet and an average year (2011, 538 mm, Figure 3-32), produces similar outputs. The wet year shows a small loss of habitat suitability at the deep seagrass edge, but some habitat gain in the shallow northern areas. This gain is related to lower SS inputs from Penrice in 2005 (988 t) than in 2006 (1,877 t) or 2011 (10,300 t), translating to lower SS load contours to the north (Figure 3-33).

In this northern region, the inputs from Penrice and WWTPs are as significant to SS load contours as the input from rivers, while further south along the coast, rivers are dominant (Figure 3-34).

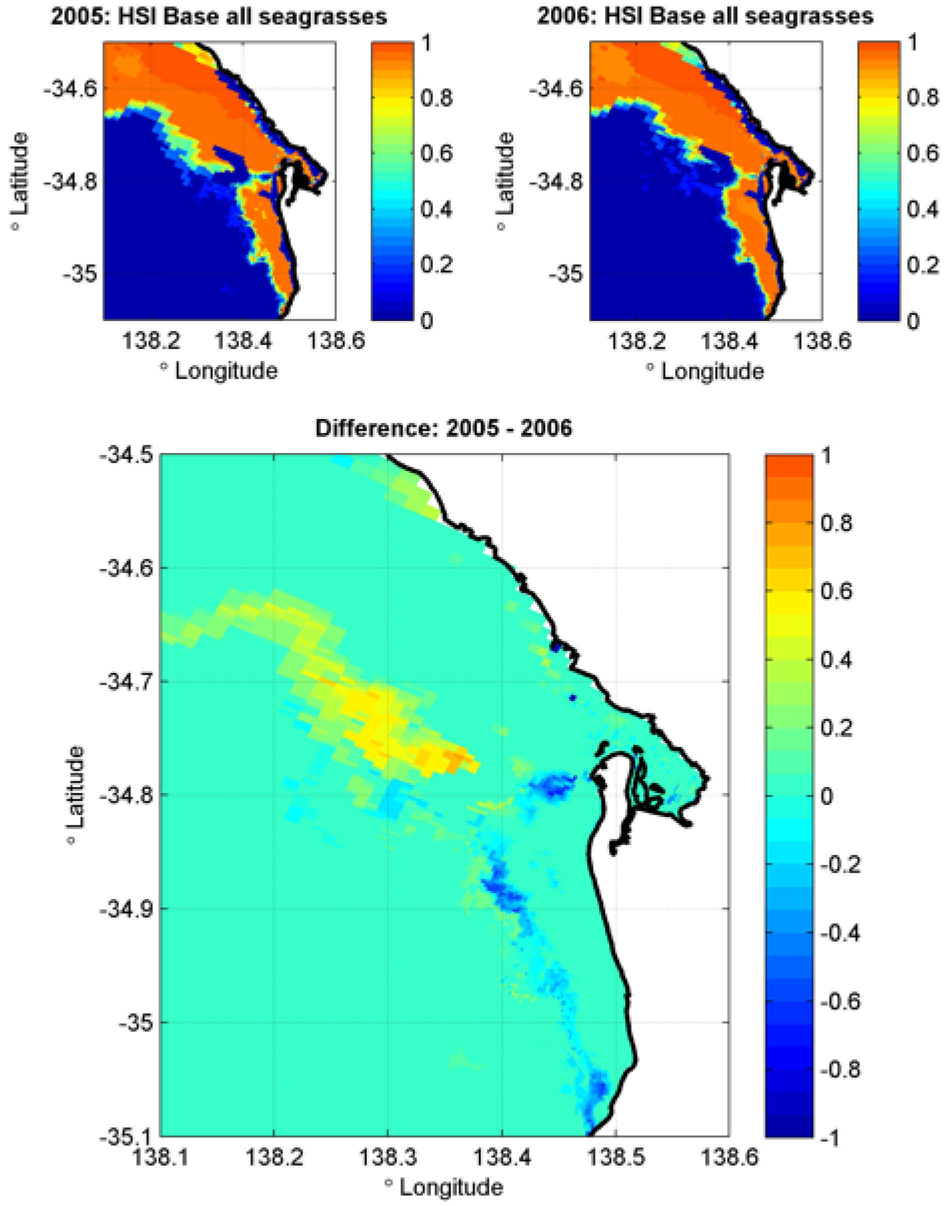


Figure 3-31 Habitat Suitability Index (HSI) for seagrasses in 2005 (wet year, top left), and in 2006 (dry year, top right), and the difference between years, with a positive value indicating higher suitability, and a negative value lower suitability, in 2005

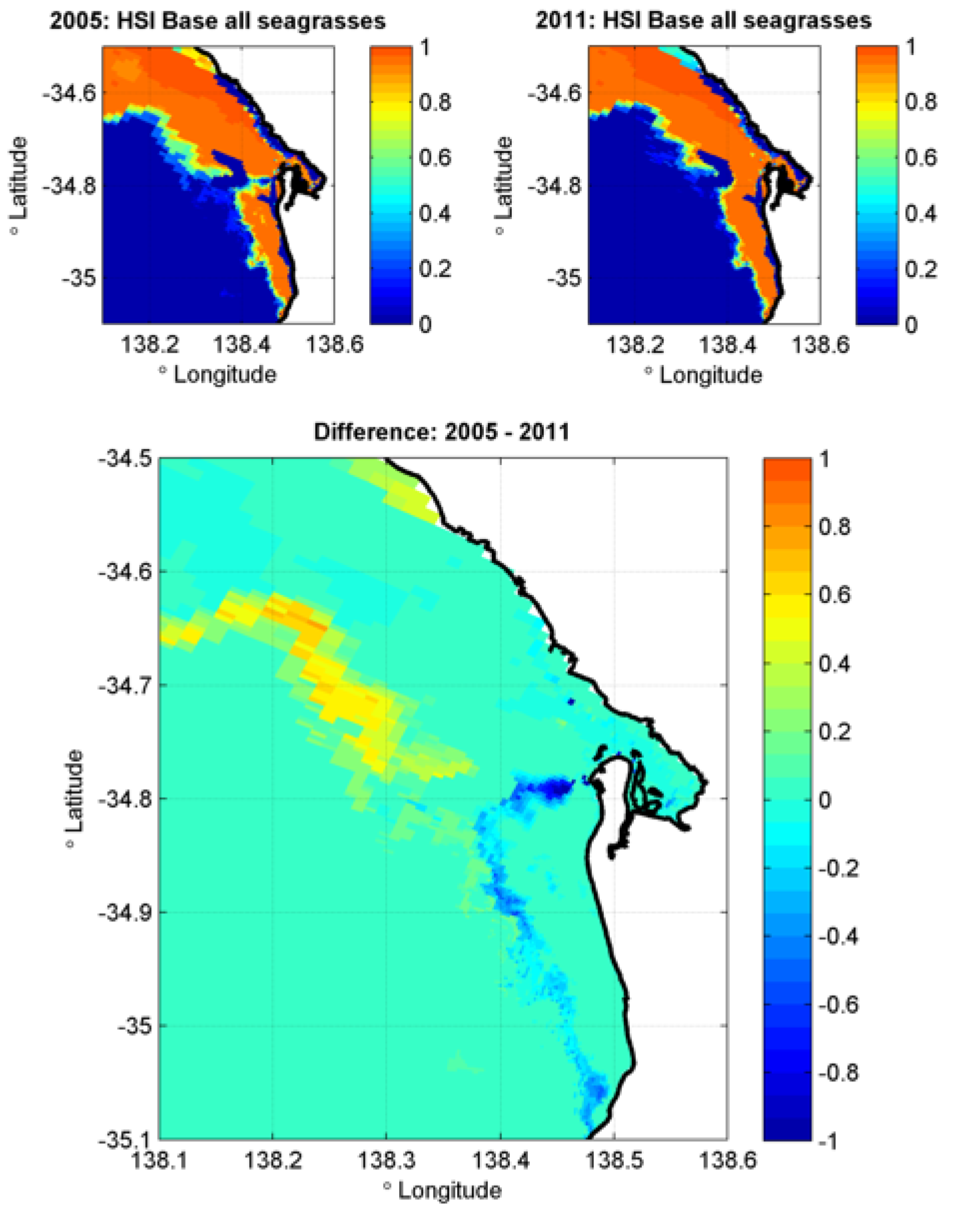


Figure 3-32 Habitat Suitability Index (HSI) for seagrasses in 2005 (wet year, top left), and in 2011 (average year, top right), and the difference between years, with a positive value indicating higher suitability, and a negative value lower suitability, in 2005

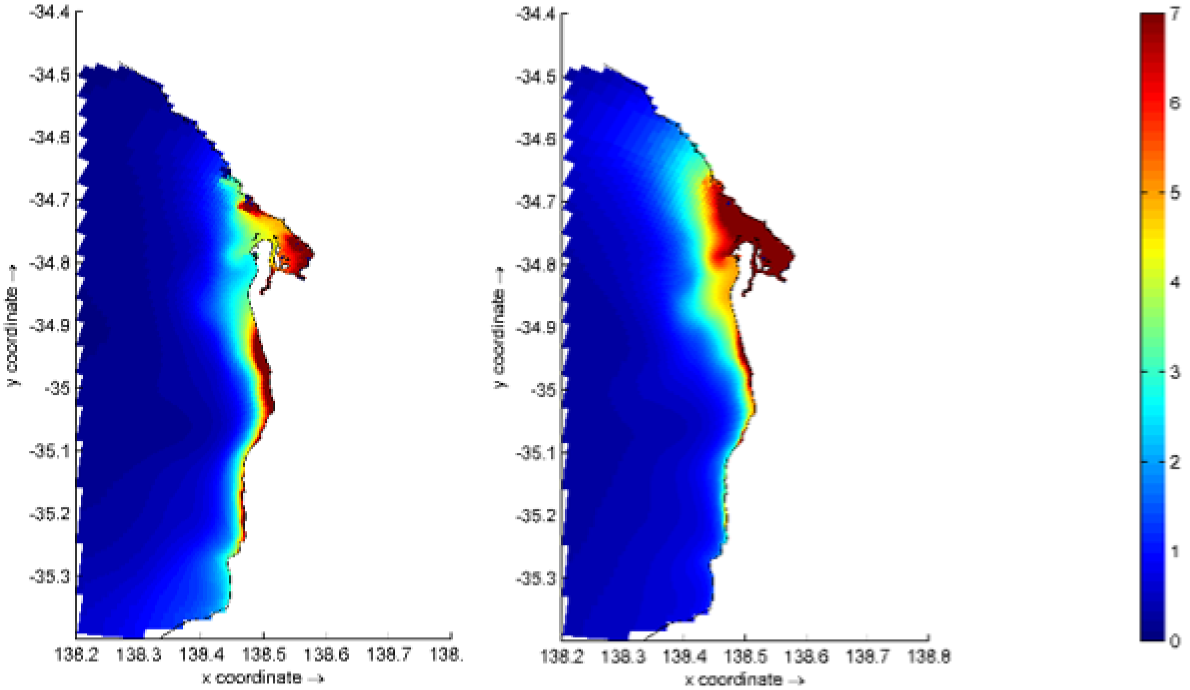


Figure 3-33 Suspended solids ASLs in 2005 (left) and 2011 (right) in t/km^2, including the Penrice discharge. Values shown are the 3-month running average (3mRA), chosen as indicative of pressure to the more sensitive seagrass species such as *Amphibolis*

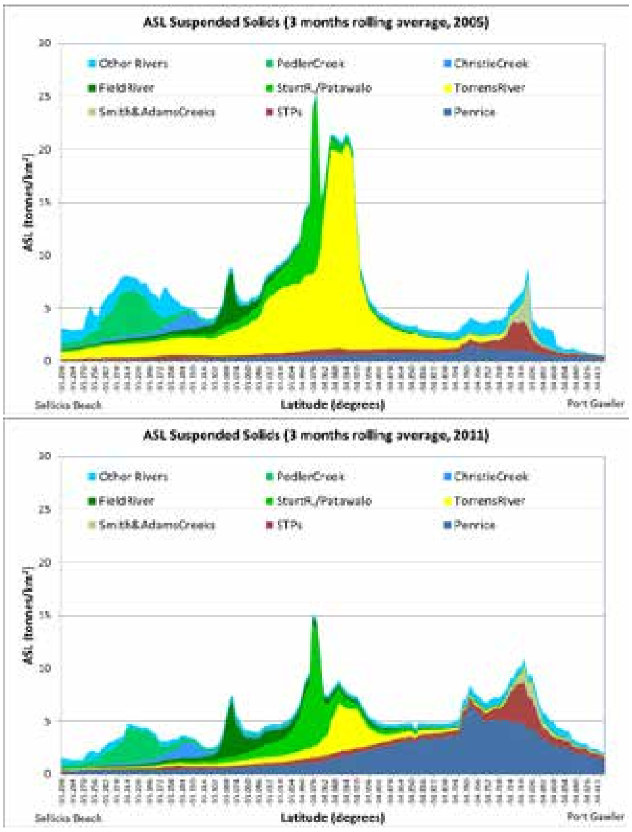


Figure 3-34 Contribution of each source to suspended solids ASLs in 2005 (left) and 2011 (right) in a line extending along the coast from Sellicks Beach in the south to Port Gawler in the north. Plots for nitrogen are included in Appendix D . Values shown are the 3-month running average (3mRA), chosen as indicative of pressure to the more sensitive seagrass species such as *Amphibolis* (3.6.6)

3.7.3 Coastal model scenario results – local

This project also attempted to differentiate the input from rivers and stormwater drains according to the zones defined as part of the Adelaide Coastal Waters Study (Fox et al. 2007).

Zone 1 comprises Gawler River, Smith Creek and all inputs physically discharging within the Barker Inlet/Port River system. Zone 2 comprises discharges to the central zone from Largs Bay to Marino. Zone 3 comprises discharges to the southern zone from Marino to Sellicks Creek. The classification was complicated by the fact that many 'natural' rivers and creeks have now been lined and ultimately discharge as a stormwater drain. Recognizing this shortcoming, the classification used is documented in Table 3-6. The loads from Helps drain and Smith and Adams Creeks were split 50:50.

Table 3-6 Classification of catchment model discharge points between stormwater drain and rivers/creeks according to zones defined as part of the Adelaide Coastal Waters Study: 1 (northern), 2 (central) and 3 (southern)

Catchment #	Name	Type	Zone
3	Lefevre Peninsula East	drain	1
61	Lefevre Peninsula West	drain	1
1	Helps	drain	1
57,63,58,59,62	Kirkcaldy-Westlakes	drain	1
26	Salt & Templers Creeks	river	1
56	Magazine Creek	river	1
53	Barker Inlet	river	1
2	Dry & Cobbler Creeks	river	1
25	Little Para River	river	1
1	Smith & Adams Creeks	river	1
75	Gawler River	river	1
74	Tennyson	drain	2
71	Patawalonga Basin 2	drain	2
72	Patawalonga Basin 3	drain	2
70	Glenelg	drain	2
4	Holdfast Bay	drain	2
69	Seacliff	drain	2
73	Torrens River	river	2
67,68	Sturt R. / Patawalonga system	river	2
5	Hallett Cove	river	3
6	Field River	river	3
7	Curlew Point	river	3
8	Christie Creek	river	3
78	Onkaparinga River	river	3
77	Pedler Creek	river	3
10	Wirra Creek	river	3
11	Willunga Creek	river	3
12	Silver Sands	river	3
13	Black Hill	river	3

In the northern Zone 1, the contribution of drains to the sediment load amounts to 28% of the total in 2011 (Figure 3-35, driven primarily by inputs from Helps drain (approximately half of the total load from drains). Despite the many small stormwater drains discharging into the central Zone 2, less than 1% of the sediment load in this zone is discharged by drains, being largely driven by inputs from the Torrens River. In Zone 3, all discharges are delivered through rivers and creek systems. These results highlight the importance of managing sediment loads from man-made drains into the northern zone. These inputs are comparatively much smaller in the central and southern zones, but the presence of several drains in the central zone would warrant the investigation of the impact of these discharges at the very nearshore if a high spatial resolution can be achieved in modelling (i.e. 50m or less, as opposed to the 200-300m resolution in AREMp).

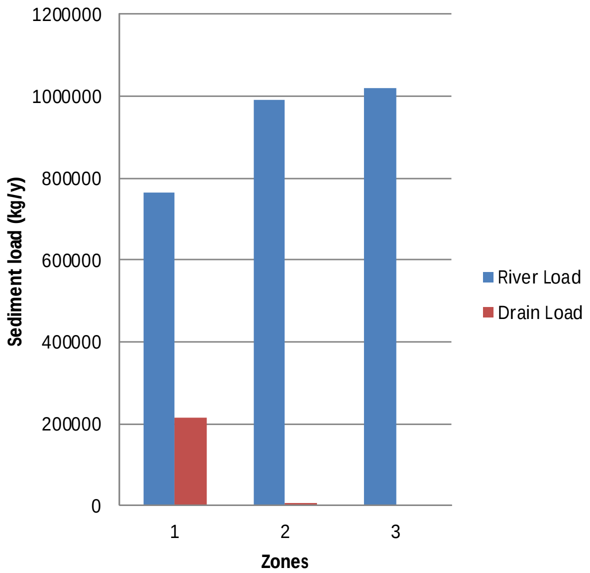


Figure 3-35 Distribution of sediment loads between rivers and stormwater drains in 2011 for each zone defined as part of the Adelaide Coastal Waters Study: 1 (northern), 2 (central) and 3 (southern)

3.7.4 Coastal model scenario results – temporal

This project further attempted to quantify the importance of individual events to the total sediment load discharged using 2011 as an example. For the Gawler River, one event in August delivered 55% of the total annual load (Figure 3-36). Four other events occurring in July, October, November and December, delivered each between 4 and 5% of the annual load. The total load discharged by these five main events represented 73% of the total annual load. The load delivered solely in winter, between July and August, accounted for 65% of the total.

The importance of individual events is even more marked for the Onkaparinga River (Figure 3-36), where the first increase in flow delivering more than 100 ML/d discharged 17% of the annual SS load, and an event in August discharged 67%. The total load delivered between July and August accounted for 88% of the annual suspended solids load.

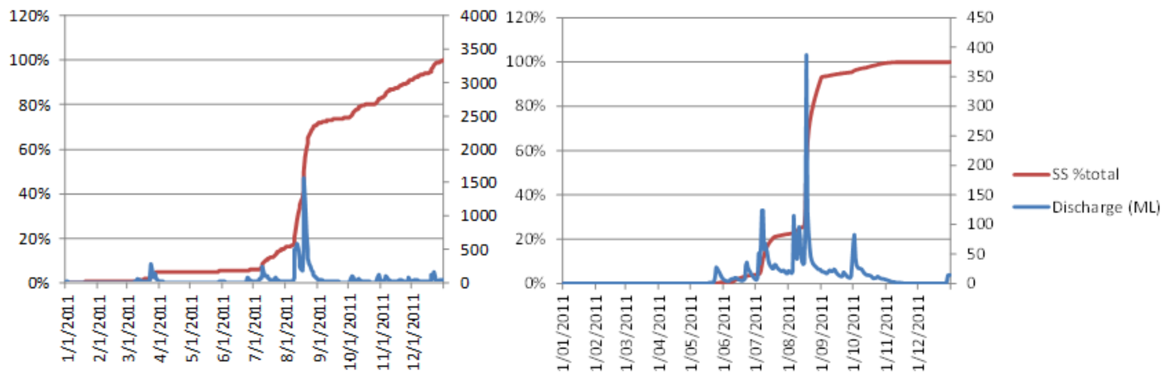


Figure 3-36 Running total for the fraction of the annual suspended solids load delivered per day during 2011 for the Gawler (left), and Onkaparinga (right) rivers. Daily discharge is also shown

The situation is similar for the Torrens River (Figure 3-37). Although a complete dataset is not available for 2011, data from 2012 and 2013 illustrate the importance of individual events. In 2012, one event in June carried 18% of the annual SS load, and one extended event in August another 53%. The total for the five largest events accounted for 81% of the annual load. In 2013, one event in June carried 18% of the annual SS load, and separate events in July and August 20 and 28%, respectively. The total for the five largest events was 74%. The winter load was less important than for the Gawler or Onkaparinga Rivers, accounting for 59% of the total in both 2012 and 2013.

Overall, for the three main rivers discharging SS to the Adelaide coast, the five largest events carried over 70% of the total annual load, and the winter discharge accounted for between 60 and 90% of the total. Interventions that target winter loads or main flow events would thus intercept the majority of solids discharged to sea.

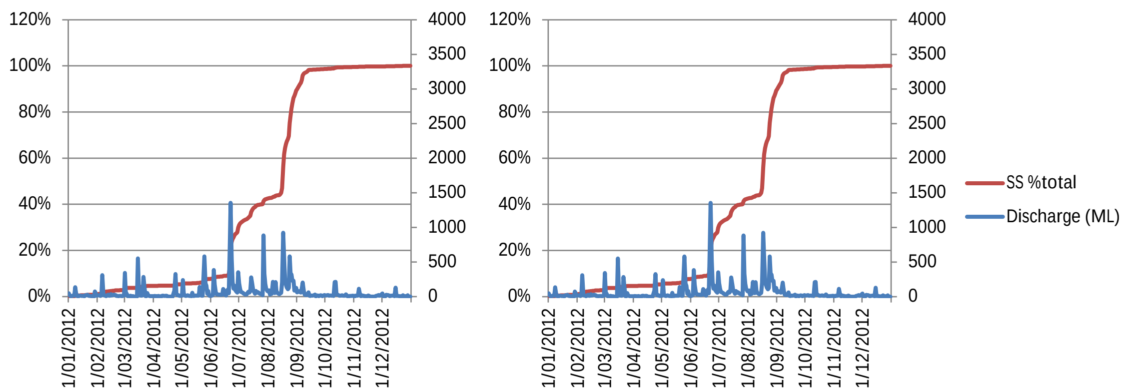


Figure 3-37 Running total for the fraction of the annual suspended solids load delivered per day during 2012 (left) and 2013 (right) for the Torrens River. Daily discharge is also shown

4 Discussion

In essence the goal of this research project was to develop new tools and knowledge that could assist in targeting stormwater interventions in time, space and scale, to support seagrass health and recovery in the Adelaide coastal waters.

This was to be achieved primarily by establishing a common version Source model for water quantity and quality across metropolitan Adelaide (ICUWM); and coupling this with an improved version of SA Water's pilot biogeochemical model of the coastal waters (AREMp) to develop a 'proof of concept' computational modelling capability that could be used to identify the magnitude and general locations of stormwater intervention measures with potential to reduce the sediment discharges from urban catchments that contribute to 'hotspots' of impact in the coastal waters.

Results from this research project described in Chapter 3 reveal that it has been possible to develop the proposed modelling capability. The first part of this chapter, §4.1, explores the extent to which the modelling capability developed is 'fit for purpose' and whether there are improvements that could be made.

The second part of this chapter, §4.2, explores how the results of this project can be interpreted so as to inform policies, plans and performance standards regarding stormwater interventions. The approach taken is sequential, working back from the water quality needed to sustain healthy seagrass in Adelaide's Coastal Waters, to how stormwater discharges impact this according to characteristics such as their timing (season), duration, sediment load and particle size distribution, through the geographic origin of the discharges at the catchment and sub-catchment scale, and finally to the extent to which different types of interventions could reduce coastal impacts.

By taking this approach, the results of this project are presented in a manner that can also be readily used as inputs to any of the ACWQIP conceptual models produced for the EPA by Cheshire (2015).

4.1 Suitability of models for identifying stormwater interventions

4.1.1 Use of EMC/DWC values tailored for Adelaide catchments

In the absence of the locally-derived Gonzalez EMC/DWC values, this project would have used typical MUSIC model parameters (Fletcher 2004) for constituent generation in the ICUWM model. MUSIC parameters are available by land uses whereas the Gonzalez EMC/DWC values are available by sub-catchments and have been derived using observed local data. On this basis the Gonzalez EMC/DWC values are considered more applicable to the project area and provide a better outcome than other concentration values, which may not be reflective of local conditions.

Using the EMC/DWC approach however, it is not possible to vary constituent concentration with flow in the Source (or MUSIC) model. Use of event mean concentrations ameliorates this limitation to some extent as theoretically it means that under/over estimation should cancel out over a longer period. This approach is considered acceptable for long-term planning (but not for short-term operational planning).

With regard to this project, use of an EMC/DWC approach would only be less suitable if seagrass health is sensitive to short spells of high discharges over a few months, as it does not provide sufficient temporal granularity to capture sub-seasonal fluxes in concentration. Given that the dominant species of seagrass along the Adelaide coast *Amphibolis* spp and *Posidonia* spp can tolerate low light conditions for 3 months and 6 months respectively, and the benefits of adopting an existing approach augmented by local data, the generation and use of Gonzalez EMC/DWC values is considered generally suitable for this project. As this project has subsequently found that the timing of sediment discharges has a large impact on the light calculated by the coastal model, for the purposes of coupling catchment and coastal models an approach that also enabled granularity at least to a seasonal level would be useful.

On a detailed level the Gonzalez (2015) values show good fit with measured data for dry/average years but represent wet years less well. This is almost certainly the result of having a small dataset (albeit local and sub-catchment specific) for deriving the parameters.

A better approach would be to derive water quality constituent generation values by using flow/constituent data that are representative of climatic conditions such as wet, dry, average, low base flow and high base flows. This would require further work using a wider set of data collected from a range of sub-catchments that can be characterised by their land uses. This is a large research task, but ultimately preferable to using MUSIC's typical parameters.

4.1.2 Suitability of the ICUWM (Source) model for the task

In selecting a model for simulating catchment based discharges and constituents, key aspects to consider include:

- spatial scale
  - the spatial extent required to be covered by this project was metro Adelaide to complement the coverage of the coastal model i.e. the coastline of metro Adelaide at high resolution, and beyond at lower resolution
- temporal scale
  - sub-daily temporal scale was required to achieve optimal outcomes because the coastal model operates on a sub-daily basis.

Three options were available for this project: (1) Source, (2) MUSIC or (3) undertake a review to identify a suitable, alternative commercially available model. Option #3 was out of scope for this project. MUSIC meets sub-daily temporal scale requirements, and theoretically, could be applied at metro scale (although common practice is to apply MUSIC at local catchment rather than regional scale), but has limited ability to represent water management options. MUSIC is however good for examining individual WSUD installations (e.g. wetlands, bio retention, rain gardens). Given that Source can be applied at multiple spatial and temporal scales, and has the necessary functionality to represent a wide range of land and water management options, the decision was made to adopt Source as the modelling platform.

Source provides more power than MUSIC to identify optimal locations and the magnitude of management action in terms of reduction in constituents and flow, by considering coastal hot-spots. There is a subsequent role for MUSIC however as it can be applied at local scale to define details of the management action (e.g. size of wetland, nature of treatment and specific location).

It is worth investigating Source's ability/limitations with regard to applying the model at sub-daily scale, or to linking with companion models such as MUSIC, to provide this functionality. Such an investigation was outside the scope of this project.

4.1.3 Suitability of the AREMp for the task

The pilot coastal water quality model, AREMp, aims to explain and predict light availability for seagrass as a function of coastal discharges (WWTPs, industry and stormwater) and its application in this project produced a first set of promising results. The model is driven by hydrodynamic, waves and water quality simulations and supported by an unprecedented inventory of quantitative habitat suitability thresholds for nine local seagrass species. The wave module suggests near-shore zones of high wave stress and significant sediment resuspension. By performing simulations with and without stormwater, this project highlighted the importance of sediment resuspension (as opposed to short-term SS inputs) in controlling the underwater light climate in areas where seagrasses have disappeared or become fragmented, something not yet widely acknowledged.

AREMp however has not yet been calibrated and validated to successfully reproduce the low light levels observed nearshore. The current limitations of AREMp include low spatial resolution close to shore (200m), insufficient temporal resolution in the definition of river outflows, low resolution substrate maps that do not take into account the role of seagrass cover in stabilizing sediment resuspension, wave model uncoupled to water quality model with implication for sediment resuspension, no coverage of autochthonous CDOM sources, and epiphyte growth parameters based on macroalgae leading to an underestimation of epiphyte cover.

4.1.4 Suitability of coupled ACDC model

This project proved that it is possible to couple the ICUWM and AREMp models and there is merit in doing so.

To simulate potential impacts of land-based discharges on seagrass, either outputs from a Catchment model or observed flow and constituents data are required. For the project area the availability of observed flow data is limited. Therefore, a catchment model is needed to produce land-based discharges. Furthermore, use of a catchment model helps examine various combinations of land and water management and climate scenarios, which cannot be done solely with the use of observed data. This is because the way to vary the observed data to simulate changes to discharges under different combinations of land and water management and climate scenarios is not known.

Given there are two models, coupling of them is essential. The best case is for both models to operate on the same temporal scale. If there are difficulties in setting up both models for the same temporal scale (e.g. due to lack of quality data for model calibration or inadequate representation of physical processes in the model at the required temporal scale), a method is needed for coupling. We used a 'wet', 'average' and 'dry' year approach. This is because the coastal model has the limitation of long run times, making it prohibitive to run simulations for periods longer than 12 months, at this stage (the ICUWM model has a simulation period of 30 years).

4.2 Targeting stormwater interventions using project results

While some limitations to the models have been identified comprising mainly data availability for the ICUWM model, nearshore light climate for the AREMp, and coarseness of timesteps for ACDC, these limitations are not considered sufficient to hinder Proof of Concept demonstration that ACDC and its component models can potentially be used to inform targeting of stormwater interventions as described below. They do however mean that results should be treated with caution and at this stage only used as indicative.

4.2.1 Nitrogen and suspended sediment load targets for Adelaide's coastal waters

Understanding the nutrient and sediment thresholds for seagrass decline and seagrass recovery is fundamental to identifying the load/concentration targets that interventions collectively (not necessarily just for stormwater) need to achieve to support healthy seagrass.

While it did not use the term Area Specific Load (ASL), the Adelaide Coastal Waters Study (ACWS) did in fact suggest an ASL threshold for TN of 1 tonne/km^2 applied uniformly across all parts of the coastal waters. It did not propose an ASL for TSS, however based on the ACWS recommendation that a 50% reduction was required from 2003 load levels, this project calculated an overall ASL for TSS of 7 tonnes/km^2.

Modelling undertaken in this project indicates that most of the coast is below the ACWS thresholds except for a few hotspots, explaining recolonisation in several areas.

For nitrogen, other lines of evidence pursued in this project comprising evaluation of historic load data at times of significant seagrass loss, suggest that a target of 1.5 tonnes TN/km2 might be appropriate. This should be seen as conservative however, given the unknown role played by nitrogen speciation (i.e. the effect of organic N is possibly larger than inorganic N).

For TSS the overall target ASL remains unclear due to the unknown residence time of particles in the system. Additional modelling studies currently being undertaken by SA Water are anticipated to provide some indication of the residence time. In the meantime this project has worked with the ASL calculated from the ACWS recommendation which is considered likely to be conservative.

While one of the key findings of the ACWS and the basis of this project is the role of suspended sediment in causing water quality unsuited to healthy seagrass due to light attenuation, it is important to also note the potential role of stormwater derived CDOM. Despite the effort in this project to calibrate CDOM inputs with new UV-absorption measurements in rivers and WWTPs (Appendix C), AREMp currently underestimates light attenuation by CDOM in the coastal zone (as measured by SA Water during model calibration). Solutions to this problem might include the inclusion of CDOM in the catchment model and a better understanding of the contribution of other sources of CDOM to the coastal light climate (e.g. seagrasses, mangroves, sediment resuspension).

4.2.2 Interactions between stormwater and wastewater discharges

Having identified the relevant targets, the next question to answer is

Is it necessary to reduce both stormwater (SS) and wastewater (predominantly nitrogen) inputs to promote seagrass health i.e. are stormwater interventions necessary/can they make a difference?

One approach used in this project was to identify locations (hotspots) where the ACWS-derived ASLs for TN and TSS are exceeded. Results of this approach start to provide spatial resolution regarding current areas of impact from land-based discharges of sediment and nitrogen to the coastal waters and the discharge source. Specifically the use of Overall Impact Indicators (§3.6.2) revealed that two of the four main impact hotspots – Christies/Onkaparinga and Bolivar/Gawler – are driven by nitrogen loads; the Barker Inlet hotspot is driven by suspended sediment loads; and the Torrens/Glenelg/Patawalonga hotspot is driven by both nitrogen and sediment. This suggests that reductions in both stormwater and wastewater inputs are only needed to ameliorate one hotspot, and implementation of stormwater interventions to reduce suspended sediment in discharges from Barker Inlet, and along the coast from Glenelg to Grange would possibly be more effective than in other areas.

4.2.3 Impact mechanism for sediment discharges - resuspension

A second approach used in this project comprised investigating the impact of stormwater on seagrass health using the outputs of light and habitat suitability for seagrass from the AREMp model. These results suggest that while direct stormwater discharges have only a marginal influence on the coastal light climate and by inference seagrass health, the resuspension of coastal sediment originally derived from stormwater, has a strong and direct influence. This finding is supported by more detailed analysis of data collected as part of the ACWS during the set-up of AREM, which reveals periods of low light levels in the coastal waters occur at times isolated from stormwater discharge events and characterised by windy conditions. Such wind driven resuspension events occur throughout the year at a rate of 2 to 3 per month.

With regard to stormwater interventions, this means that benefits in terms of seagrass habitat suitability are unlikely to be seen in a short time frame (i.e. a few years) but are speculated to have a protracted positive effect over decades given the likely slow flushing regime for the coastal waters. Understanding the mechanisms and timeframe of flushing of sediment from the system, is an important next step with potential to be explored using SA Water's fully developed AREM.

4.2.4 Factors affecting sediment resuspension and stormwater's contribution

The extent of resuspension of sediment in Adelaide's coastal waters has been shown to depend largely on two key factors – wind activity as described above, and particle size. Physics dictate that the particle size most susceptible to resuspension is the sub 63µm fraction, with these particles remaining in suspension for longer, and having a higher specific light attenuation coefficient than larger particles.

Historically the main source of sediment in the coastal waters has been stormwater, although the contribution of wastewater has steadily increased, now corresponding to about half of all suspended solids discharged to the coast. However stormwater discharges remain the prime target for interventions given fine particles <63µm on average comprise 54–71% of the total load from stormwater as measured at Torrens, Gawler and Onkaparinga Rivers (Table 3-1) compared to 30–65% of the load from wastewater discharges depending on the outfall (Wilkinson et al. 2003).

The size of flow events is a key factor affecting the load of TSS discharged to the coastal waters. This project has shown that the bulk of fine sediments is delivered to the coast during large stormwater flow events rather than multiple small (<7mm rainfall) events.

4.2.5 Potential guidance for targeting stormwater interventions in space

Recognising that sediment sourced from stormwater is a major contributor to the sediment pool available for resuspension in the coastal waters, an important first step for targeting interventions geographically is to identify those catchments making the greatest contribution of the particle size of concern. Results of this project showed that among the three catchments with the highest annual TSS discharge loads – Torrens, Gawler and Onkaparinga (Jones 2015), the Torrens and Gawler deliver the highest percentage of sub 63µm particles (71%) during winter flows. When coupled with consideration of coastal hotspots that revealed TSS load to be a driver of the Glenelg/Patawalonga/Torrens hotspot but not the Bolivar/Gawler hotspot, the Torrens catchment emerges as a potential priority for stormwater interventions.

For the other smaller contributing catchments, particle size data are not available and hence assessment can only be made based on their contributions to total TSS loads (which are much smaller than the three major catchments) and their contribution to hotspots. On this basis, other potential catchments to be targeted for reducing sediments include the Patawalonga basin, Dry Creek and Cobbler Creek.

Furthermore, in the northern coastal zone the contribution of drains to the sediment load is significant amounting to 28% of the total in 2011 (driven primarily by inputs from Helps drain), highlighting the importance of managing sediment loads from man-made drains into this zone. By contrast less than 1% of the sediment load in the central zone is discharged by drains. Given the occurrence of multiple small drains in this zone however, investigation of the localised impact of these discharges at the very nearshore may be warranted if a high spatial resolution can be achieved in modelling (i.e. 50m or less, as opposed to the 200-300m resolution in the AREMp).

Within the discharging catchments, more specific guidance on locations where stormwater interventions could be most effective is provided by model outputs showing sub-catchments discharging the highest loads of TSS. It is interesting to note that of the stormwater harvesting schemes considered in this project, approximately 80% are located in sub-catchments identified as likely to contribute sediment loads at the higher end of the range (18–96 kg/ha/year).

Further resolution to the level of TSS contribution by land use would be ideal however this is beyond the capability of the ICUWM model because the revised values for EMC/DWC were derived and applied at sub-catchment, not land use scale. While it is technically feasible to apply EMC/DWC values at the land use scale, derivation of appropriate values for land use classes based on local data requires monitoring data from small catchment areas with homogenous land use and for the range of possible land uses to be adequately represented. The available data from existing monitoring stations used in this project were generally located at the outlet of sub-catchment areas with mixed land use hence preventing determination of clear water quality parameters for specific land use classes.

4.2.6 Potential guidance for targeting stormwater interventions in time

This project has revealed a number of temporal aspects of relevance to targeting stormwater interventions.

The reviews by Erftermeijer (2014, 2015) indicate that the light threshold for seagrass species varies between 2 and 20% of surface irradiance, over three months for the more sensitive species and six months for the more resilient species. These sensitivities have been built into the project methodology and used to identify the 3- and 6-month rolling average hotspots as most relevant.

Project results show winter to be the season of greatest impact on coastal water quality from stormwater discharges. The bulk of fine sediments are delivered to the coast in winter, and all hotspots show marked seasonal variability, with suspended sediment concentrations generally peaking in August. The underlying cause of this temporal feature is shown to be individual large stormwater discharge events that happen predominantly in winter. For example in 2012 for the Torrens River one event in June carried 18% of the annual SS load, another extended event in August carried 53%, and overall the largest five events accounted for 81% of the annual load.

Another key temporal aspect relates to differences in particle size distribution of sediments across the hydrograph of stormwater discharges. While there are some differences between the three major catchments, in general the <63µm fraction is most concentrated in flows associated with the falling hydrograph.

Finally as highlighted earlier, a key gap in knowledge relates to both the local and whole-of-system flushing times for sediments which is important for quantification of the pool of sediment available for resuspension at any one time. Addressing this knowledge gap was however beyond the capability of this project.

4.2.7 Towards a metro Adelaide performance specification for stormwater interventions that best contribute to healthy seagrass habitat

Only a small element of this project was directly focused on assessing the type of stormwater intervention that would be most effective in alleviating coastal hotspots as a' proof of concept'. This comprised scenario testing of stormwater harvesting versus interventions characterised by a 'filter' approach e.g. wetlands or swales, which revealed that for the Torrens catchment stormwater harvesting at 50% of design capacity delivers a greater reduction in total discharged loads than the filter model applied at all 'urban' functional units. While these outputs principally demonstrate that the ICUWM model can be used successfully to explore such questions, they also support other lines of evidence suggesting that to be effective (in reducing coastal impacts arising from sediment discharges), interventions must be capable of tackling large flow events.

Findings that only emerged during (from SA Water's AREM project) and as a result of this project regarding the critical role of resuspension of sediment in causing coastal habitat to be unsuitable for seagrass growth, meant that some components of this project could not be tailored to targeting stormwater interventions on this basis. For example catchment modelling did not differentiate particle sizes in sediment loads whereas particle size was included in the AREMp speciation of inputs.

Nevertheless it is possible to suggest generic guidance regarding the type and location of stormwater interventions that might best facilitate coastal water quality suitable for healthy seagrass from a suspended sediment perspective.

Characteristics of such interventions would potentially include:

- capacity to operate effectively during large flow events especially in winter
- ability to trap <63µm particles
- ability to permanently remove sediment from the system (by active maintenance if necessary)

- preferential location in the Torrens catchment, followed by Patawalonga, Dry Creek, Cobbler Creek and Helps drain catchments
- preferential location in sub-catchments that yield high volumes of <63µm sediment

design focus on capturing sediment during peak and falling hydrograph.

5 Conclusions and recommendations

5.1 Conclusions

This study has found that sufficient and suitable data exist to underpin development of computational models of the metropolitan Adelaide catchment (ICUWM model) and the adjoining coastal waters model (the AREMp) focused on suspended sediment and nitrogen.

The modelling outputs are considered very useful, although the modelling accuracy could be improved with more and better data especially relating to sediment and nutrient inputs at the upstream catchment boundary. Insufficient data are available for consideration of coloured dissolved organic matter (CDOM) however, which is a significant data gap given CDOM is recognised as a contributing factor to light conditions unsuitable for healthy seagrass growth.

The ICUWM model and the AREMp developed in this study and the ACDC model formed by their coupling, have been demonstrated as new tools with potential to be used to inform the design of stormwater interventions aimed at achieving coastal water quality suitable for healthy seagrass.

Coupling works well, but to achieve better representation of the light climate of the coastal waters both the ICUWM model and the AREMp need to be further developed with particular emphasis on:

- finer spatial resolution to delineate impacts nearshore and wave-flow coupling in the AREMp
- better temporal representation to deliver hourly inputs of SS, and further development to include CDOM in the ICUWM.

Using the models developed, this study investigated two lines of evidence to assess the impact of stormwater on coastal seagrasses. Determination of Area Specific Loads (ASLs) enabled the identification of the areas along the coast where the load limit recommended by the ACWS is exceeded based on load inputs and hydrodynamics, and where potentially seagrasses are at greater risk of loss. Use of habitat suitability maps went a step further, and took into account not only load inputs and hydrodynamics, but also resuspension and thresholds of impact for several water quality parameters including light as affected by direct and indirect shading. The ASL approach suggests that load limits are only exceeded in localized areas nearshore. Results of the habitat approach are tentative given the AREMp limitations, but also suggest low suitability nearshore, albeit as a function of wave dynamics (physical forcing) rather than light. These two lines of evidence should be viewed as complementary in the assessment of spatial impact of loads and their effect on habitat suitability.

The analysis of underpinning data and operation of the models yielded new knowledge of particular relevance to both the targeting of stormwater interventions and conceptual models of the Adelaide catchments and coastal waters as follows:

- A nitrogen load limit in the same order of magnitude to that recommended by the ACWS is supported by the historical reconstruction of input loads and seagrass extent, but the nature of the nitrogen discharge (i.e. particulate vs dissolved) appears to have a decisive role on the actual threshold.
- A suspended sediment area specific load limit can be calculated based on the ACWS recommended load reductions and used to inform impact hotspot identification.

- Four hotspots of coastal impacts from land based discharges were identified: Christies/Onkaparinga, Torrens/Glenelg/Patawalonga, Barker Inlet, and Bolivar/Gawler, with suspended sediment from stormwater only a significant factor for the hotspots at Torrens/Glenelg/Patawalonga and Barker Inlet.
- While the direct effect of stormwater on the coast is suggested to be negligible using the AREMp with its current configuration, other lines of evidence indicate resuspension of sediment derived from stormwater to have a decisive role in limiting light available to seagrasses.
- Sediment flushing from the coastal waters is expected to be slow and hence sediment originating from stormwater may have a long legacy effect on seagrass health through sediment resuspension.
- The size fraction most susceptible to resuspension is sub 63 µm.
- Based on the limited sampling to date, the bulk of suspended sediment is delivered to the coast in a few large events, mostly in winter, and from the Torrens and Gawler catchments.
- More fines (<63 µm) are delivered in the falling limb of event hydrographs during winter flows, with the contribution of <63 µm particles typically increasing from 50-60% of the total to >70%.
- In this project no simple relationships were found between particle size distribution (mean particle size, percentage of each fraction size) and flow for any of the rivers.
- Interventions such as stormwater harvesting that remove all of the suspended sediment contained in the harvested flow, are likely to be more effective in reducing TSS loads discharged than 'filter based' interventions in predominantly urban catchments, especially in wet years.

While this project was not scoped to fully answer all of the questions relating to targeting stormwater interventions, based on the findings it is possible to suggest preliminary and generic guidance regarding the type and location of stormwater interventions that might best facilitate coastal water quality suitable for healthy seagrass from a suspended sediment perspective. Characteristics of such interventions would include:

- capacity to operate effectively during large flow events especially in winter
- ability to trap <63um particles especially during falling hydrograph flows during winter
- ability to permanently remove sediment from the system (by active maintenance if necessary)
- preferential location in sub-catchments that yield high volumes of <63µm sediment – high TSS yielding sub-catchments have been identified but further work is required to establish if these are also the dominant source of <63µm sediment
- preferential location in the Torrens catchment as this is the highest contributor of suspended sediment and its discharge drives a coastal impact hotspot.

5.2 Recommendations for further work

While this project has achieved its objectives there are several areas where greater value could be derived through more modelling, additional data collection and/or analysis. These comprise:

- improved understanding of seagrass distribution at times and locations of differing discharge loads
- estimation of the residence time of sediment within the coastal system
- increased knowledge of CDOM loads and sources
- increased knowledge of fine sediment loads and sources

- upgrading of the ICUWM to deliver finer granularity in timestep and inflow node data.

In addition, while the results of this project suggest that stormwater-borne sediment loads may not drive hotspots of impact in the southern parts of the coastal waters from a seagrass perspective, no assessment has been made of the impact of sediment loads on reefs. This could be achieved by:

- augmentation of the AREMp to include habitat suitability for reefal communities.

5.2.1 Improved correlation between seagrass distribution and pollutant loads

The data available for the historical reconstruction of load thresholds suffers from the poor temporal resolution of seagrass mapping in Adelaide, which has been done generally on 5-year intervals. The development of techniques that allow for cost-efficient collection of yearly datasets would be useful to pinpoint exact loads and temporal lags in loss and recovery. The earlier datasets (pre-2007) also do not allow for an estimate of the impact of the northern WWTP of Bolivar, the largest in Adelaide, and future effort would benefit from focus in the area to investigate thresholds for recovery now that the Penrice discharges to the Port River have ceased.

5.2.2 Estimation of sediment residence time in the coastal system

For TSS the overall target area-specific load remains unclear due to the unknown residence time of particles in the system. Additional modelling studies currently being undertaken by SA Water are anticipated to provide some indication of the residence time. Once this estimate is known, the overall area-specific load target can be refined and hotspot identification reviewed.

5.2.3 Increased knowledge of CDOM loads and sources

Despite the effort in this project to calibrate CDOM inputs with new UV-absorption measurements in rivers and WWTPs (Appendix C), the AREMp currently underestimates light attenuation by CDOM in the coastal zone (as measured by SA Water during model calibration). Solutions to this problem might encompass the inclusion of CDOM in the ICUWM catchment model and a better understanding of the contribution of other sources of CDOM to the coastal light climate (e.g. seagrasses, mangroves, sediment resuspension). The review of data undertaken at the start of this study identified a paucity of CDOM data and hence there is a need to collect more CDOM data if this parameter is to be included in the ICUWM model.

5.2.4 Increased knowledge of fine sediment loads and sources

Given this study’s finding regarding the importance of the relationship between fine sediment resuspension and seagrass habitat suitability, it is critical to develop a better understanding of the source and loads of these sediments at finer spatial and temporal resolutions.

It is very difficult to derive clear water quality signatures for urban land use classes, so a focus on sub-catchment level monitoring including particle size distribution is recommended. Most important for coastal water quality and seagrass habitat is determining the contribution of different parts of a catchment to the total SS load delivered to the coast and those areas that deliver the highest amount of fines. Temporal trends such as seasonal and inter-annual variability will also be important to understand.

Better temporal and spatial resolution of monitoring is also required for CDOM (if shown to be important) in main catchments and both TSS and CDOM in catchments where little/no data is currently collected and there is/will be significant development in the future (i.e. southern catchments).

5.2.5 Augmentation of the ICUWM model

The catchment model inputs to the AREMp are simulated on a daily basis, and therefore underestimate turbidity peaks (and probably overestimate base or slow flows). The simulation of these peaks could be improved with a finer temporal resolution for peak flows and constituent concentrations. SA Water has collected additional data to relate flow to SS concentrations. The data collected was based on grab samples and targeted peak flow events in the larger rivers, to supplement regularly collected data by the AMLR NRMB which are biweekly and use flow-proportional sampling. The correlations derived from these data could be applied in the ICUWM model to provide hourly inputs if calibrated against load inputs calculated from flow-proportional sampling. From a more general perspective, it is considered worth investigating the Source platform's ability/limitations with regard to its application at sub-daily scale, which should include testing of the relevance of flow and constituent generation and transportation processes at a sub-daily scale, too. This is a big task but would improve Source's application to water management in urban areas.

Within the ICUWM model, 3 inflow nodes for the Gawler, Torrens and Onkaparinga catchments were used. These nodes contained a time series of inflows representing releases and spills from reservoirs upstream of the project area catchments as well as replacing flows for upstream sub-catchments (as these could not be calibrated separately) within the project area. Sufficient and suitable flow weighted composite sampling water quality data were not available to enable inflow constituent concentrations or loads to be accurately characterised. Given inflow volumes are significant in some years (23–30% on average) it is important to not only ensure that the water quality values used for configuring the constituent generation for the inflow nodes are appropriate, but that the load and particle size distribution of sediments entering the system is well understood.

This project has focused on seagrass health as a measure of suitable water quality for the Adelaide coastal waters, however in the nearshore environment water quality suitable for primary contact recreation from a pathogen perspective is also a requirement, especially during the warmer months of the year. Use of the EMC/DWC based models to evaluate microbial pollution from stormwater discharges is reported from the literature and hence the ICUWM may also have potential to be augmented for this purpose.

5.2.6 Augmentation of the AREMp

This study utilised a suite of indicators for seagrass habitat suitability derived from the literature relating to local and interstate populations. A similar exercise could be undertaken for the macro-algal reef communities that are prevalent in the southern parts of the project area using literature values and findings of many years of monitoring of reef health. The impacts on this habitat type of either sediment originating from stormwater or sediment introduced or mobilised by other activities such as dredging or disposal of dredged material, could then be assessed.

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Appendix A Details of water quality datasets used in the project

A.1 Catchment data

ApxTable A-1 Links to publicly available water quality monitoring datasets

Database/dataset	Hyperlink
AMLR NRMB monitoring databases	<http://www.naturalresources.sa.gov.au/adelaidemtloftyranges/about-us/our-regions-progress/monitoring-and-evaluation/water>
AMLR Water Information portal	<http://amlr.waterdata.com.au/>
EPA Data portal	<http://www.epa.sa.gov.au/environmental_info/water_quality/water_quality_monitoring_data>.
DEWNR WaterConnect	https://www.waterconnect.sa.gov.au

It should be noted that none of the catchment datasets that were reviewed cover the Salt and Templers Creek catchments in the north, or Ingleburne Creek, Willunga Creek, Silver Sands or Black Hill catchments in the south. As these are all relatively small catchments (each comprise 0.9-6.3% of total ACWS model catchment area, and together comprise 11.8%) and together contribute only 4.7% of total modelled flows, we considered that their exclusion would not affect the findings of this project.

A.1.1 Adelaide & Mt Lofty Ranges Natural Resources Management Board (AMLR NRMB)

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
Water Information	AMLR NRMB	Public	33	1994-present	Integrated composite	Y	Y	Y*	Y
Water Information	AMLR NRMB	Public	5	1972-present	Flow gauge	Y	N	N	N
Water information	AMLR NRMB	Agreement	2	2008-2013	Integrated composite	Y	Y	Y	Y

(Extract from Table 2-1)

The AMLR Natural Resources Management Board (AMLR NRMB) holds a large amount of data, including composite sampling data for the parameters of interest (TSS, TN and TP) and linked flow records for 33 sites within the project area.

The most relevant of AMLR NRMB's monitoring databases is the 'Surface Water' database which is a long-term surface water monitoring program looking at water quality and quantity. This database contains flow, TSS, TN and TP data (among other parameters) collected at many sites, of which 33 are within the project area (see Figure 2-1 for their locations) with records dating from 1994 to the present. This is a useful dataset for the current project as it contains many long term sets of composite sampling data analyzed for the parameters of interest. An additional five flow gauging stations without water quality data were included. Two further sites, the A5041011 Barker Inlet Wetland on HEP Drain and the A5041012 Barker Inlet Wetland on NAE Drain, were acquired. These 40 (33+5+2) datasets were deemed to be suitable for constituent (TSS, TN, TP) modelling at a daily time step.

The W5040002 Old Port Rd Drain Outfall site was also acquired but deemed unsuitable as it contained only two water quality grab samples in September and August of 2009 and did not have associated flow data. Three other AMLR NRMB databases – Stormwater, Waterwatch and Environmental Flows – were investigated, but were found to be unsuitable for this project. The Environmental Flows and Stormwater databases had no water quality data and Waterwatch did not have the water quality parameters of interest to the project, and were also grab samples.

ApxTable A-2 lists the number of samples (where concentrations were >0 mg/L) and period of record for the 33 sites grouped by their location within the modelled sub-catchments. Sub-catchments are identified by their sub-catchment number (SC#). Matching daily flow data were sourced for all records.

ApxTable A-2 Number of samples(n) and period of record for composite water quality monitoring from the AMLR NRMB Surface Water database and 2 additional sites in the Port Adelaide #1 sub-catchment

SC#	SOURCE sub-catchment	TSS n	TP n	TN n	Period of record
1	Central Sturt River	95	95	95	May-10 to Jan-15
14	Christie Creek	52	52	44	Oct-10 to Nov-14
2	Dry & Cobbler Creeks	40	40	40	Apr-11 to Oct-11
25	Lower Little Para River	22	22	22	Aug-10 to Jan-15
30	Field River	54	54	51	Jul-10 to Dec-14
32	Gawler River	73	73	70	May-10 to Dec-14
38	Lower Brownhill Creek #1	69	69	57	May-10 to Nov-14
40	Lower First Creek	1357	1358	1343	Sep-94 to Nov-14
44	Lower Onkaparinga River	93	93	80	May-10 to Dec-14
53	Lower Pedler Creek	603	591	560	May-04 to Feb-15
54	Lower Third Creek	286	283	283	Jan-97 to Dec-08
55	Port Adelaide #1	163	163	146	Aug-09 to Dec-14
56	Port Adelaide #2	118	118	100	May-10 to Dec-14
57	Port Adelaide #3	196	193	181	Oct-04 to Nov-14
58	Port Adelaide #4	41	41	40	May-11 to Dec-14
6	Sixth Creek	377	377	376	Feb-01 to Dec-14
60	Smith & Adams Creeks	78	78	68	May-10 to Nov-14
66	Sturt River	377	378	373	Jan-97 to Dec-14
73	Torrens River #5	53	53	49	May-11 to Nov-14
75	Torrens River #6	61	61	53	Sep-10 to Aug-14
77	Turretfield	51	51	48	Aug-10 to Jan-15 – TSS and TP Aug-10 to Dec-14 - TN
8	Upper Brownhill Creek	74	74	68	May-10 to Nov-14

N.B. n where concentration was >0 mg/L; TN was calculated as the sum of TKN and NOx.

A.1.2 Environment Protection Authority (EPA)

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
Goyder MLR WQ Modelling project	EPA	Public	8	1971-2007	Grab	N	N	Y	Y
Goyder MLR WQ Modelling project	EPA	Agreement	27	2008-2011	Integrated Composite	Y	N	Y	Y
Goyder MLR WQ Modelling project	EPA	Agreement	21	1973-2008	Integrated Composite	Y	N	N	Y
Goyder MLR WQ Modelling project	EPA	Agreement	1	2011-2015	Grab	Y	Y	N	N

(Extract from Table 2-1)

EPA data were compiled for an earlier Goyder Institute project (the Mount Lofty Ranges (MLR) Water Quality Modelling project, Kuhnert et al. 2015) and made available to the project.

Grab sample data (TP and TN, not flow or TSS) for eight sites were collected every six months or so between 1971 and 2007 (see Figure 2-1 for site locations). Grab sampling, as opposed to integrated composite sampling, represents a single point in a hydrograph. It is therefore possible that some samples were taken during times of no flow (i.e. standing water). Flow could potentially be linked, by date, to other datasets (e.g. AMLR NRMB gauging data) or to daily modelled flow from the existing SOURCE Catchment hydrological model.

TN and TP data from 2008 to 2011 from the 27 EPA sites located within the project area were provided. A further 21 sites (also monitored by the EPA) from 1973 to 2008 included only TP. These data used an integrated composite sampling methodology taking samples across the hydrograph at a daily temporal resolution.

The EPA provided a set of monitoring data relating to a site immediately downstream of a quarry in the Greater Adelaide metropolitan region. These data included daily flow gauging, turbidity measurements and occasional event-based grab sampling for TSS. A strongly positive linear relationship between daily turbidity measurements and grab sampled TSS observations was found (R^2>0.9). This relationship was used to derive daily estimates of TSS with flow for the 2011–2015 time series.

A.1.3 SA Water

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
AREM Project	SA Water	Agreement	3	2010-2014	Discrete composite	Y	Y	Y*	Y
Goyder MLR WQ Modelling project	SA Water	Agreement	1	1996-2013	Integrated composite	Y	Y	Y	Y

(Extract from Table 2-1)

Flow and water quality parameters including EC, TSS, temperature, turbidity, pH, TP, TKN, NOx, and total Cu, Pb and Zn were collected at three gauging stations along the Onkaparinga, Gawler and Torrens Rivers from 2010 to 2014 by SA Water (see ApxFigure A-1 for site locations). These data were collected using a discrete interval auto-sampler to capture samples over the hydrograph of a flow event with the intention of investigating the dynamics of concentrations with flow. These relationships, together with composite data, could allow reconstruction of loads at higher temporal resolution.

Data compiled by Kuhnert et al. (2015) from a site at the bottom of Scott Creek (A5030502) and sampled using an integrated composite sampling method from 1996 to 2013 at a daily temporal resolution were also made available to the project. Water quality parameters included TSS, TN and TP, and flow. These data were assessed as potentially suitable for use in constituent generation modelling in the project.

A.1.4 DEWNR WaterConnect

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
WaterConnect	DEWNR	Public	59	1968-present	Flow gauge	Y	N	N	N

(Extract from Table 2-1)

The Department of Environment, Water and Natural Resources (DEWNR) host a database of surface water monitoring sites across the project area. The 'Surface Water Data' database is an interactive map relating to surface water resources across South Australia with a focus on the historical perspective. Within the AMLR NRMB region, the database consists of mainly flow gauging data and rainfall. Water quality data, where available, are limited to temperature, EC, and turbidity. A total of 59 stations within this database are located within the project area with periods of records extending back to the 1960s for some stations. Data from these stations were assessed as potentially suitable for our purposes.

A.1.5 CSIRO

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
MAR Research Projects	CSIRO	Agreement	1	2006	Composite	Y	Y	Y	Y
MAR Research Projects	CSIRO	Agreement	1	2010-2012	Integrated composite	Y	Y	Y	Y
MAR Research Projects	CSIRO	Agreement	9	2010-2012	Grab	Y/N	Y	Y	Y

(Extract from Table 2-1)

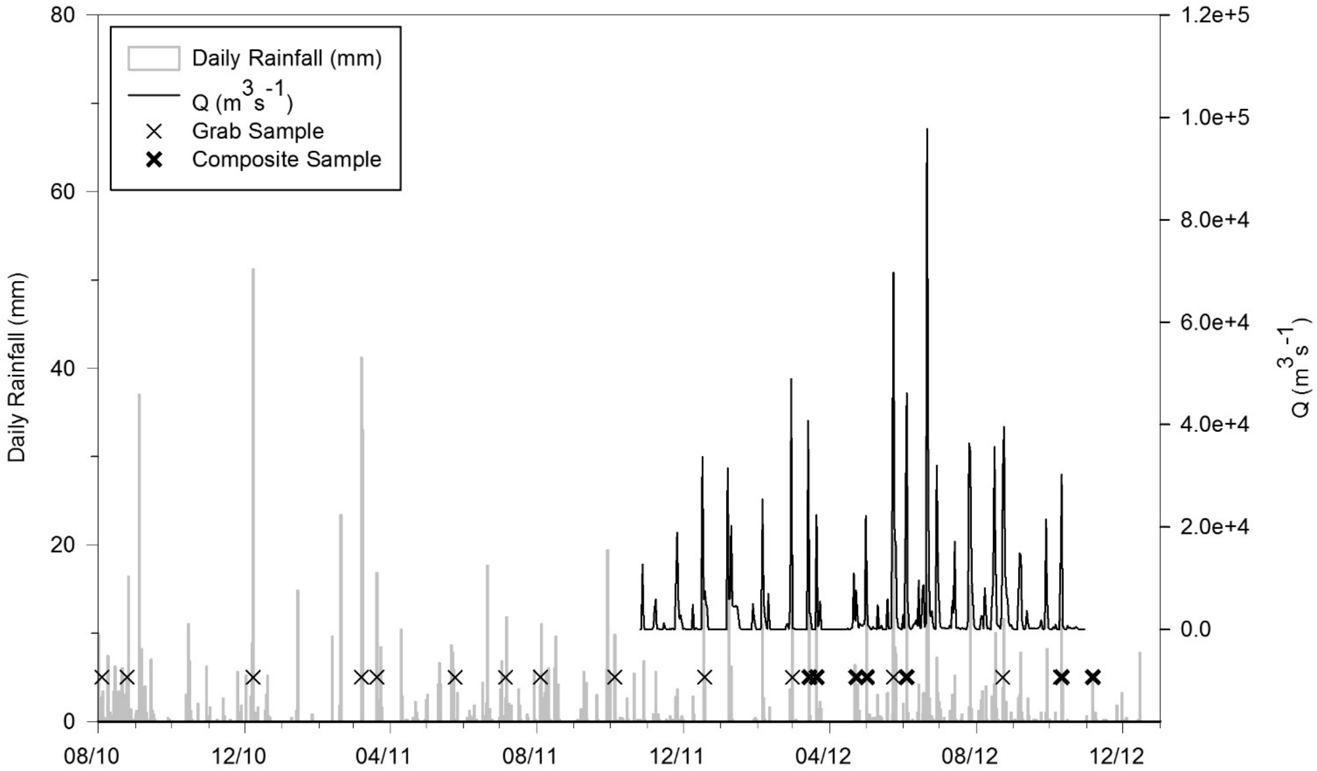
CSIRO has collected water quality samples within the Salisbury area of northern Adelaide as part of research projects on managed aquifer recharge (MAR).

Event-based composite sampling was conducted at Parafield in 2006. While this study captured seven events and flow-averaged concentrations of total nutrients and suspended solids (among other parameters) (see Table 3 in Page et al. 2008), these data were collected at the outlet of a 50 ML instream basin (Page et al. 2008).

Event-based integrated composite sampling and grab sampling were conducted in 2010–12 at the Parafield Drain during flow (ApxFigure A-1).

Grab samples were also collected at eight other sites within the Parafield catchment during flow (Page et al. 2013), resulting in a total of 70–85 samples for total nutrients and suspended solids (among other parameters) (Appendix 5, Page et al. 2013). Flow was recorded at the Parafield Drain throughout the composite sampling program.

CSIRO flow weighted composite sampling data from the outlet of the detention basin at Parafield were excluded from modelling constituent concentrations. Water quality could be affected by physically, biologically and chemically driven processes during surface storage. Sampling data from the 9 catchment sites were limited to integrated composite data collected only at the Parafield Drain site where flow was also recorded. These integrated composite data were the only datasets deemed suitable for use in this project.



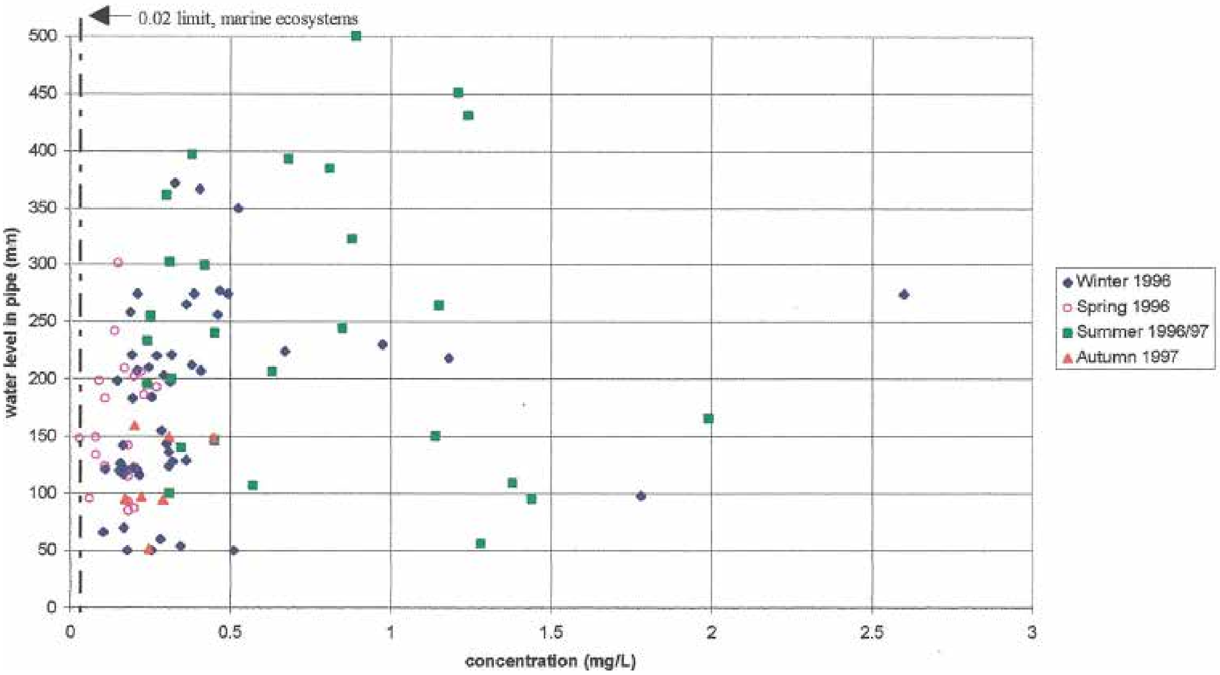
ApxFigure A-1 Grab and composite event-based sampling at the Parafield Drain monitoring site from 2010-2012

A.1.6 University of SA Drain 18

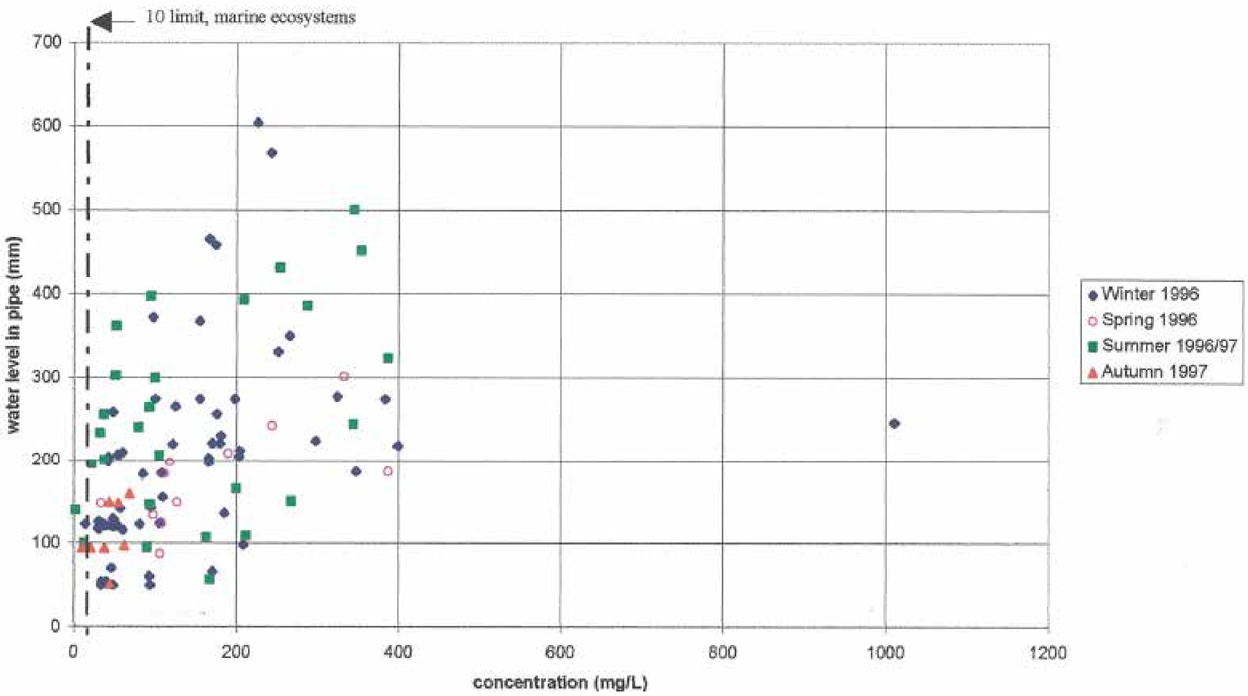
Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
Drain 18	Uni SA	Not available	1	1994-1997	Auto	N	Y	Y	Y

(Extract from Table 2-1)

Between winter 1994 and autumn 1997, the University of South Australia conducted stormwater quality monitoring of a stable medium density urban residential catchment in the suburb of Glengowrie. Sampling was undertaken using an auto-sampler to collect water samples analyzed for nutrients, suspended solids and heavy metal concentrations. The only accessible report for this work details sampling undertaken at the Frederick Street site from winter 1996 to autumn 1997 (Scott 1997). A total of 162 samples were collected over this time from 43 rainfall events and were analyzed for TP, TSS, TKN, turbidity, TDS, and heavy metals (chromium, copper, zinc and lead) . Raw data from this or previous monitoring were unavailable. Scatter plots of TP and TSS concentrations against water level in the pipe appear in the report and are reproduced in this report for reference (ApxFigure A-2, ApxFigure A-3). No clear relationship between level and concentration is evident. TP concentrations appeared to be high in winter and summer samples with no obvious seasonal patterns in the TSS concentrations. Conversion of pipe level to volumetric flow rates would require the rating curve for the pipe at the Frederick Street site. This information was judged unsuitable for use in the project.



ApxFigure A-2 TP concentrations against pipe water level for sampling at Frederick Street site (from Scott 1997)



ApxFigure A-3 TSS concentrations against pipe water level for sampling at Frederick Street site (from Scott 1997)

A.1.7 Local Government Authorities

Project/Program	Custodian	Access	#Sites	Period of record	Sample type	Related flow	TSS	TN	TP
Council projects	City of Salisbury	Agreement	3	2003-2008	Integrated Composite	Y/N	Y	Y	Y
Council projects	City of Playford	Agreement	3	2007-2012	Integrated Composite	Y	Y	Y*	Y

(Extract from Table 2-1)

WDS monitored and maintained composite sampling stations for two local councils within the project area: The City of Salisbury and The City of Playford. The City of Salisbury composite sampling water quality data included TSS, TP and TN for three sites collected from 2003–2008 in the Smith and Adams Creek catchment (see Figure 2-1 for site locations). A data use agreement was obtained from the City of Salisbury for the data to be provided by WDS.

The City of Playford (also through WDS) provided monitoring data for another three sites in the Smith and Adams Creek catchment (see Figure 2-1 for site locations) monitored from 2007–2012. These data relate to integrated composite sampling data for TSS, TP and TN (approximated as the sum of TKN and NOx) and also record daily flow.

A.2 Coastal data

A.2.1 Seagrass distribution

Seagrass extent change maps (seagrass and bare substrate) were produced from digital (or digitised) aerial imagery, sourced from DEWNR. These data covered a series of change mapping study periods from 1949 to 2013. The 2007 to 2013 change map extents ranged from Middle Beach in the north to Hallett Cove in the south (Hart 2013). Mapping extents for earlier periods (1949–1996) were from Largs Bay in the north to Marino in the South (EPA 1998). These data enable results from the current coastal model to be compared with seagrass mapping of similar years/periods.

A.2.2 Freshwater inputs

Freshwater inputs to the sea were derived from the modelling work done by Jeremy Wilkinson as part of the ACWS (Wilkinson et al. 2003, Wilkinson 2005, Wilkinson et al. 2005a, Wilkinson et al. 2005b). Input files were provided by the University of Western Australia Oceans Institute, and were those used to model the impact of inputs to the Adelaide coast during the ACWS (Pattiaratchi et al. 2007). These included daily flows for the following rivers and stormwater drains: Gawler River, Smith Creek, Helps drain, Little Para River, Dry Creek, Port Catchment, Torrens River, Patawalonga system, southern drains (Pier Street, Broadway, Marine Street, Harrow Road, Wattle Avenue, Edward Street, Young Street, Marino), Field River, Christie Creek, Onkaparinga River, Pedler Creek, Maslin Creek, Willunga Creek, Aldinga Creek and Sellicks Creek.

A.2.3 Wastewater streams

With regard to treated municipal wastewater, only monthly flows were available for WWTPs. The first piped discharge from the Glenelg WWTP occurred in 1943, and the WWTPs of Bolivar and Christies Beach were commissioned in 1967 and 1971 respectively (Wilkinson et al. 2003). The Port Adelaide WWTP started discharge into the Port River in 1935 for a population of 37,320, but was disregarded as there are no data to estimate loads, and discharge was not directly to the coast. As noted later in the report, historic seagrass reconstructions were prepared for two years - 1940 and 1975. Wastewater inputs for 1940 were considered negligible and wastewater discharges for 1975 comprised the WWTPs of Bolivar, Port Adelaide, Glenelg (including sludge discharge) and Christies Beach.

With regard to industrial wastewater, no data were available to quantify the discharge from the Penrice soda ash factory to the Port River in 1940, and an annual load estimate was used for 1975 (Appendix C).

A.2.4 Water quality data

The water quality data used to calculate daily loads from land-based sources were sourced from the ACWS historical reconstructed loads (Table 5.6 in Wilkinson et al. 2005b). The water quality data for stormwater is restricted to samples collected since 1972 (Table 6 in Wilkinson et al. 2005a). Daily loads for 1940 were calculated using the same water quality as that compiled for 1975. This approach was taken as, while the catchment was less developed in 1940 than 1975, there would have been little or no sediment control. This represents a worst case scenario.

The 1940 and 1975 load inputs were translated into area-specific loads using the coastal model AREMp (see §2.2.2 for a description of this model). The load contours were overlayed either to the extent of bare sand (1940) or areas of seagrass loss (1975).

A.3 New data

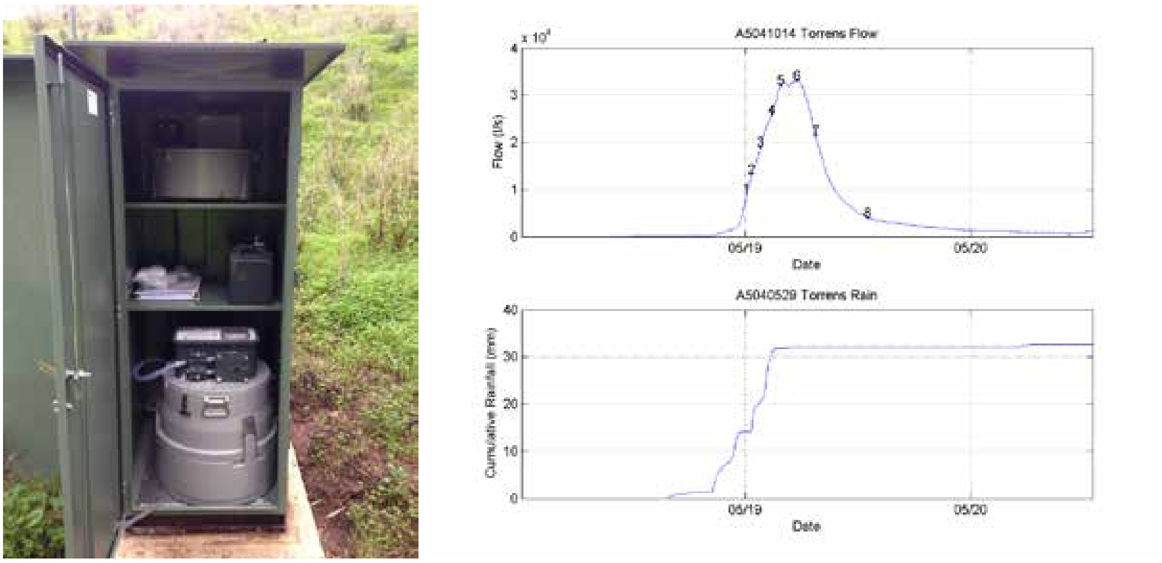
A.3.1 Stormwater particle size and organic carbon content

New data were collected to verify the distribution of particle sizes in stormwater inputs. The AMLR NRMB operates a large water quality monitoring network in Adelaide's rivers, comprising flow-proportional composite water quality sampling, and flow data. Autosamplers were installed by SA Water at three major AMLR end-of-catchment sites (ApxTable A-3), at the same locations where routine composite samples are collected. These sites were chosen because they contributed more than half of all sediment inputs to the Adelaide coast between 2009 and 2014, with 36% delivered by the Torrens River, 13% by the Gawler River and 7% by the Onkaparinga River (Jones 2015).

ApxTable A-3 Grab sampler locations for measuring stormwater particle sizes

Site Code	Site Name
A5041014	Torrens River @ Seaview Road Bridge
A5050510	Gawler River @ Virginia Park
A5031005	Onkaparinga River 1.1 km u/s Ford Old Noarlunga

Stormwater samples were collected for all events above 7mm rainfall at several points across the hydrograph and analysed for their particle size fractions (IM1 (< 16mm), IM2 (16-63 mm) and IM3 (>63 mm) and their total or dissolved organic carbon content. Particulate organic carbon was determined as the difference between total and dissolved organic carbon content.



ApxFigure A-4 Automatic sampler installation (left), and example hydrograph with sample collection times shown (right)

The Gawler, Torrens and Onkaparinga sampling sites are hydrometric monitoring stations funded by the AMLR Board. These stations have a flow rated weir. The intake of the samplers is located just below the crest of the weir, allowing sampling away from the river bed but still able to capture small events.

During significant rain events (>7mm rain) grab (instantaneous) samples were taken at various times during the hydrograph (ApxFigure A-4(right)). Samples were collected between July 2014 and September 2015. As a highly urbanised river with a high proportion of impervious surfaces, the Torrens River discharged after rain events throughout the year. This allowed for samples to be collected in the winter of 2014 (n=26), summer and autumn (n=37), and winter of 2015 (n=28). The Gawler River has a large agricultural catchment and was only sampled in the winter of 2014 (n = 42 samples over 5 events). This river did not have any significant flows during 2015 because the catchment never became saturated (BOM dryness index for Mt Crawford). The Onkaparinga River has a mixed catchment, with more urbanised areas than the Gawler River. Samples for this river were collected in the winter of 2014 (n=35) and 2015 (n=49).

Samples were retrieved as soon as practical (usually next day) after the full set samples were collected. Bottles were transported in eskies on ice and then stored in a refrigerator until the time of analysis. Samples were analysed in a LISST laser diffraction particle size analyser (Sequoia Scientific, USA). Particle size fractions are reported as IM1 (< 16mm), IM2 (16-63 mm) and IM3 (>63 mm). Mean particle size was calculated with the Folk and Ward graphical method in the software package GRADISTAT v.8 (Blott and Pye 2001).

Samples for the analysis of organic carbon were either analysed unfiltered for the determination of total organic carbon (TOC) or filtered for the determination of dissolved organic carbon (DOC; 0.45 mm syringe filter). Samples were acidified to release inorganic carbon, then digested with sodium persulphate at 98–100 °C and the resulting carbon dioxide measured in a OI Analytical Organic

Carbon Analyser 1030 (APHA-AWWA-WEF 1995). Particulate organic carbon (POC) was determined as the difference between TOC and DOC.

Appendix B Alternate approaches to constituent generation modelling

B.1 Power Function

The Power Function (PF) is a constituent generation model that comes installed within the SOURCE program that relates constituent load or concentration to flow rate. The power function is commonly applied at a catchment scale but can also be applied at sub-catchment scale. We considered applying the Power Function at the sub-catchment scale by pairing water quality and flow data points across all sites within each sub-catchment and deriving rating curves based on Equation 2 (after Kelley and O'Brien 2012):

$$fC_i = aQ^b + c$$

Where fC_i is concentration at flow rate i, Q is flow rate, a is the slope coefficient (on semi-log axes), and b is curvature.

This approach is dependent on being able to observe a sufficiently clear relationship between concentration and flow. The PF approach on daily time step for urban catchments may not be applicable given the often sub-daily nature of urban hydrology and limited sampling across individual flow events.

It was proposed to explore the use of the PF approach at the catchment scale in the first instance. Catchments for which no data exist could be modelled according to the PF from the most similar (in terms of land use, area and hydrology) catchment. However, time did not permit this exploration during the life of the project.

Some of the long term water quality and flow datasets (e.g. AMLR NRMB Surface Water Database) may also support consideration of the contribution of flooding and high flow events versus smaller more frequent flows to constituent concentrations and loads to the coast. It was proposed to explore data to examine water quality trends according to a range of average recurrence intervals (ARIs) for flow rates. These data could also be explored using outputs from EMC/DWC SOURCE model runs. Outputs from the SOURCE model could be interrogated to determine the modelled contribution of infrequent higher flow events to overall long term loads to the coast.

B.2 Dynamic sediment budget river network (SedNet) model

Sediment flux modelling is typically applied to estimate long term sediment loads simulating spatial patterns of erosion and deposition across large catchment areas. The Sediment budget river Network (SedNet) model is an example using data on terrain, soils, vegetation cover, runoff and water bodies to estimate potential mean annual sediment loads from large river basins (Wilkinson et al. 2009). A dynamic (ie temporally explicit) 'Dynamic-SedNet' (D-SedNet) is under development as a Source plug-in, and represents a further development in sediment flux modelling for estimation of loads at a daily time step (Wilkinson et al. 2014, Freebairn et al. 2015).

Sediment flux modelling like D-SedNet for highly urbanised, impervious catchments such as in the current project, may not be suitable without changes to conceptual and numerical approaches of the D-SedNet model. In addition, the current project contains a reasonable amount of water quality data allowing data-driven methods to be considered.

B.3 Loads Regression Estimator and Random Forests

The Goyder Institute Mount Lofty Ranges (MLR) Water quality modelling improvement project was undertaken to investigate the effects of proposed land use change in the MLR region on the water quality of city water supplies (Kuhnert et al. 2015). The approach taken to model hydrological aspects was based on the SIMHYD rainfall runoff model applying a Bayesian method for calibration and uncertainty analysis. Two approaches were explored for modelling water quality and loads: i) a Loads Regression Estimation (LRE) method, and ii) a Random Forests decision tree method. LRE is constructed from a four step process that consists of estimation steps for flow, a predictive model for concentration (given the flow), the estimation of the load, and the quantification of the errors in the load that incorporates errors in the flow rates. For all of the sites under study, a daily time step was used using daily flow records. The Random Forests method was applied to look at land use change. This method can give a greater number of possible predictors with good predictive capabilities (Kuhnert et al. 2015). For the application of the Random Forests method to land and water quality, classes can be broad (e.g. urban, non-urban) or have increasingly more land use classes depending on water quality data availability (for runoff from different land uses).

These statistical methods were applied outside the hydrological model. They functioned as a hydrological calibration and validation tool and as a constituent generator. Similar application of these methods in the current project using SOURCE would require a way for interfacing SOURCE with an external model. This could be achieved through coding of custom SOURCE plugins. Building of the statistical models however require substantial database processing and analyses which was beyond the scope of the current project. The LRE approach on daily time step may also not be applicable for highly impervious urban catchments given the often flashy, sub-daily nature of urban hydrology.

Appendix C Summary of load information used in the AREMp

C.1 Rivers and stormwater

For the rivers, the loadings are derived from catchment model simulated loads of suspended solids (SS), nitrogen (TN) and phosphorus (TP), supplemented by some field data and expert judgement. Concentrations of the AREMp modelled substances were derived as shown in ApxTable C-1.

ApxTable C-1 Definition of AREMp modelled variables for river loads

Full name of substance(s)	Name in the model	Urban catchments	Rural catchments
Dissolved oxygen (mg/L)	OXY	9.51	same as urban
Particulate organic carbon (mgC/L)	POC	0.11 / 2.5 * SS [2, 3]	same as urban
CDOM (mgC/L)	DOC	UV-abs [4]	UV-abs [4]
Particulate inorganic matter (mg/L)	IM	0.89 * SS [2]	same as urban
Particulate organic nitrogen (mgN/L)	PON	TN * 0.59 *0.5 [3,5,6]	TN * 0.86 *0.5 [3,5,6]
Dissolved organic nitrogen (mgN/L)	DON	TN * 0.59 *0.5 [5,6]	TN * 0.86 *0.5 [5,6]
Nitrate (mgN/L)	NO3	TN * 0.38 [5]	TN * 0.13 [5]
Ammonium (mgN/L)	NH4	TN * 0.03 [5]	TN * 0.01 [5]
Particulate organic phosphorus (mgP/L)	POP	TP * 0.56 * 0.6 [3, 7]	same as urban
Dissolved organic phosphorus (mgP/L)	DOP	TP * 0.22 [7]	same as urban
Ortho-phosphate (mgP/L)	PO4	TP * 0.22 + TotP * 0.56 * 0.4 [7]	same as urban
Silica (mgSi/L)	Si	5.6 [8]	5.6 [8]

Notes

1. Concentration at 100% saturation, fresh water of 18°C (Deltares 2014 and references therein).
2. Organic fraction is mean of observations during dedicated flow event surveys during 2014-2015 of the water quality in the Gawler, Torrens and Onkaparinga Rivers by SA Water. The organic fraction is converted from dry weight to carbon equivalents by division by 2.5. The distribution over IM1/IM2 is 57:43 for rural catchments, 60:40 for urban catchments. This has been based on the following conversion of the fractions measured to modelled IM1/IM2 as in the Phase 1 AREM: IM1 = fraction > 63 um + 50% of fraction between 16 and 63 um, IM2 = fraction < 16 um + 50% of fraction between 16 and 63 um.
3. The distribution over the rapidly decaying fraction 1 and the slowly decaying fraction 2 of POC, PON and POP is 50:50, based on the assumption that the material will be moderately degradable.

4. Mean value from dedicated flow event surveys during 2014-2015 of the water quality in the Gawler, Torrens and Onkaparinga Rivers by SA Water where available. Otherwise, the mean value from additional analyses of NRM composite samples in rivers. A value of 0.2 cm^{-1} is used for discharges without data.
5. Distribution of TN over different species is as observed during dedicated flow event surveys during 2014-2015 of the water quality in the Gawler, Torrens and Onkaparinga Rivers by SA Water. The Torrens samples are used to characterise urban catchments, while the Gawler and Onkaparinga samples are used to characterise rural catchments.
6. Assumption: 50% of organic N is in dissolved form.
7. Share of filterable reactive P in total P is 22%, as observed during dedicated flow event surveys during 2014-2015 of the water quality in the Gawler, Torrens and Onkaparinga Rivers by SA Water. We assume that there is a similar amount of P in dissolved organic form, and that 40% of the remaining particulate fraction (56%) is organic (Deltares, 2014 and references therein).
8. High end value of streams draining common rock types (Deltares, 2014 and references therein).

C.2 Wastewater treatment plants

For the WWTPs, the loadings are derived from measurements of suspended solids (SS), nitrogen (TN), phosphorus (TP), dissolved oxygen (DO), UV-abs and silica (Si), provided by SA Water. Concentrations of the modelled substances were derived as shown in ApxTable C-2.

ApxTable C-2 Conversion of measured to modelled variables for water treatment plants

Full name of substance(s)	Name in the model	Assumption Bolivar Lagoon	Assumption other effluents
Dissolved oxygen (mg/L)	OXY	9.0	5.0 1
Particulate organic carbon (mgC/L)	POC	SS * 0.8 * / 2.5 [2, 3]	SS * 0.2 * / 2.5 [2, 3]
CDOM (mgC/L)	DOC	UV-abs [4]	UV-abs [4]
Particulate inorganic matter (mg/L)	IM	SS * 0.2 [2]	SS * 0.8 [2]
Particulate organic nitrogen (mgN/L)	PON	TN * 0.12 [3,5]	TN * 0.13 [3,5]
Dissolved organic nitrogen (mgN/L)	DON	TN * 0.12 [5]	TN * 0.13 [5]
Nitrate (mgN/L)	NO3	TN * 0.74 [5]	TN * 0.66 [5]
Ammonium (mgN/L)	NH4	TN * 0.02 [5]	TN * 0.08 [5]
Ortho-phosphate (mgP/L)	PO4	TP * 0.8 [6]	TP * 0.94 [6]
Particulate organic phosphorus (mgP/L)	POP	TP * 0.1 * 0.5 [3,6]	TP * 0.03 * 0.5 [3,6]
Dissolved organic phosphorus (mgP/L)	DOP	TP * 0.1 * 0.5 [6]	TP * 0.03 * 0.5 [6]
Silica (mgSi/L)	Si	Si [4]	Si [4]

Notes:

1. A value of 5.0 mg/L has been assumed for Bolivar HS (average value of DO measured at Christies and Glenelg).
2. Distribution of SS over inorganic and organic fractions is based on expert judgement. SA Water has collected data to calibrate this assumption, but this was not available at the time of model runs for this project. The organic fraction is converted from dry weight to carbon equivalents by division by 2.5. The distribution over IM1/IM2 is 70:10 for Bolivar High Salinity, 17:3 for Bolivar Lagoon, 65:15 for Christies STP, 58:22 for Glenelg STP. This has been based on the following conversion of the fractions measured to modelled IM1/IM2 as in the Phase 1 AREM: IM1 = fraction > 63 um + 50% of fraction between 16 and 63 um, IM2 = fraction < 16 um + 50% of fraction between 16 and 63 um.
3. The distribution over the rapidly decaying fraction 1 and the slowly decaying fraction 2 of POC, PON and POP is 20:80, based on the assumption that most of the material will be relatively slowly degradable.
4. Mean values per effluent, as measured by SA Water.
5. Distribution of TN over NH4, NO3 and organic N as measured by SA Water. 50% of organic N is assumed dissolved.
6. Share of PO4 in TP as measured by SA Water. Remaining part is assumed organic, half dissolved, half particulate.

C.3 Penrice

The Penrice soda ash plant is situated on the west bank of the Port River. The Penrice outfall started operation in the 1930s (Pfennig 2008) but no data is available for the 1940s. The Adelaide Coastal Waters Water Quality Improvement Plan (WQIP) reports a SS discharge of 100,000 t/y between 1975 and 1985 (Miller 1987, Pfennig 2008, McDowell and Pfennig 2013). According to a development application for the settlement ponds of Penrice, the particulate inorganic matter retained in test ponds was mostly fine grained calcite ($CaCO_3$) (~65% < 20 mM, ~75% < 63 mM). This mineral is very poorly soluble at a normal seawater pH and has a specific density of 2.71 kg/m^3. The assumption is made that the organic fraction in SS is negligible.

Prior to the installation of settling ponds, much of the discharged solids remained in the Port River close to the discharge point, eventually impeding the passage of ships, with only a fraction of the material travelling to the Adelaide's coast. This local trapping is the result of small scale physical processes as affected by the discharge characteristics (i.e. density of the slurry substantially higher than seawater) in combination with the local Port River cross section and bathymetry. Historically, Penrice used to dredge this material from the Port River main channel every few years and discharge it directly into Adelaide's coastal waters near Outer Harbor. This practice has ceased since 1993.

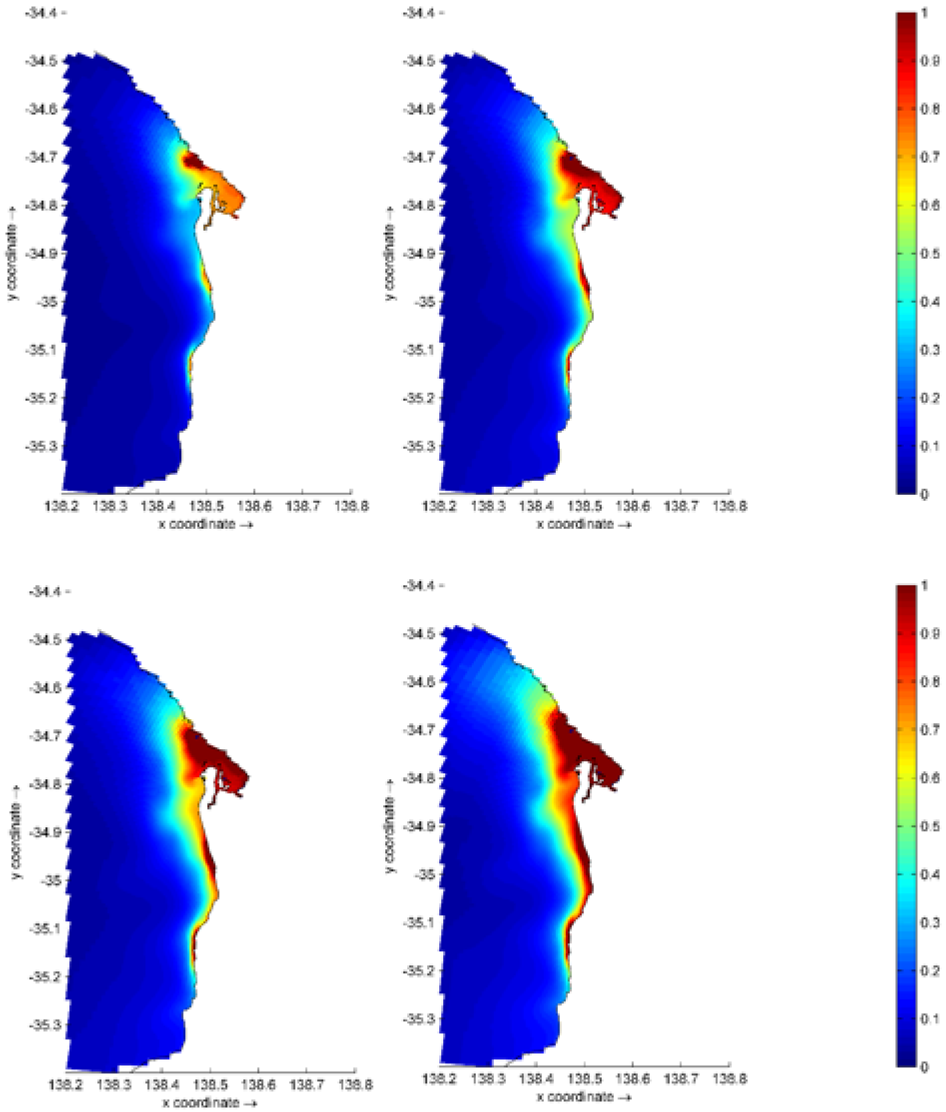
The AREMp cannot resolve the local retention of particles without detailed information about the discharge salinity, temperature and particle content as well as the ambient salinity and temperature. The fraction of the discharged particles which would not be trapped inside the Port River and find its way to Adelaide coastal waters prior to the installation to the settling ponds is estimated at 5% based on expert knowledge (Peter Pfennig, EPA, personal communication). This implies a load of 5000 tonnes/year in 1975.

A large reduction of this load occurred when Penrice developed settlement ponds in 2002. The WQIP indicates that the total load dropped to 1780 tonnes in 2003 and 810 tonnes in 2008. Daily discharge data from bimonthly reports submitted by Penrice to the EPA indicate a total suspended solids load of 10,300 tonnes in 2011. If allowed to move into coastal waters, this load would have a pronounced effect on seagrass habitat suitability in the coastal area affected by the Port River outflow.

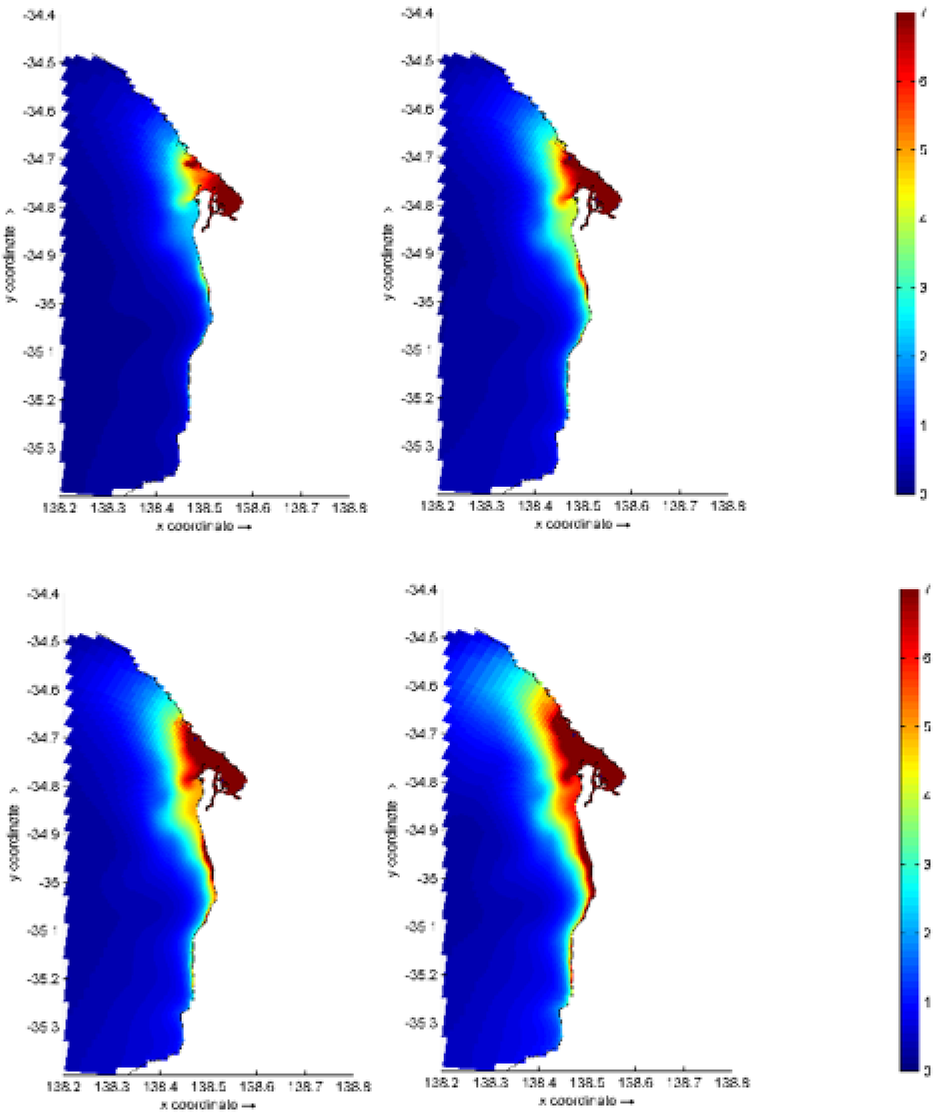
We have assumed a 50% retention of the 2011 load in the Port River. The dumping grounds for the material dredged from the Port River are in deeper water (8-10 m) with limited resuspension. Based on diving observations by the EPA, most of the dumped material remains intact (Peter Pfennig, EPA, personal communication). We consider resuspension of this material as less relevant, and assume it is included in our overall representation of resuspension.

Appendix D Additional AREM outputs

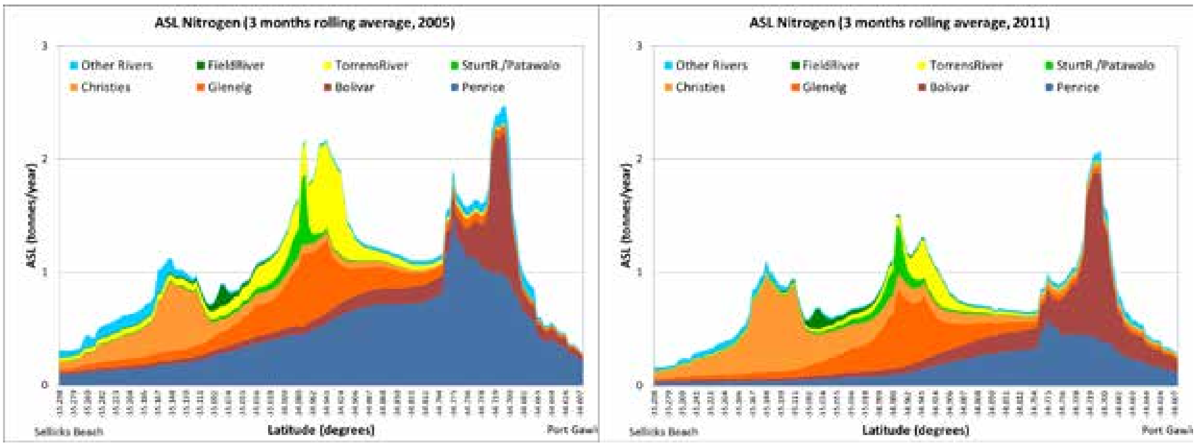
D.1 Area specific load contours



ApxFigure D-1 Nitrogen ASLs (t/km^2) in 2011 from all rivers and WWTPs, including Penrice; mean values (top left), 6 month running average (6mRA) values (top right), 3 month running average (3mRA) values (bottom left) and 1 month running average (1mRA) values (bottom right



ApxFigure D-2 Suspended solids ASLs in 2011 (tonnes/km^2) from all rivers and WWTPs, including Penrice; mean values (top left), 6 month running average (6mRA) values (top right), 3 month running average (3mRA) values (bottom left) and 1 month running average (1mRA) values (bottom right)



ApxFigure D-3 Contribution of each source to nitrogen ASLs in 2005 (left) and 2011 (right) in a line extending along the coast from Sellicks Beach in the south to Port Gawler in the north. Values shown are the 3-month running average (3mRA), chosen as indicative of pressure to the more sensitive seagrass species such as *Amphibolis*

The Goyder Institute for Water Research is a partnership between the South Australian Government through the Department of Environment, Water and Natural Resources, CSIRO, Flinders University, the University of Adelaide, the University of South Australia and ICE WaRM (The International Centre of Excellence in Water Resources Management).