APPENDIX B
RISKS TO MARINE HABITATS IN THE PORT GAWLER REGION FROM STORMWATER FLOWS

Marine Ecosystems

Risks to marine habitats in the Port Gawler region from stormwater flows

K. H. Wiltshire

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SOUTH AUSTRALIAN RESEARCH AND DEVELOPMENT INSTITUTE

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Author(s):	K. H. Wiltshire
Reviewer(s):	J.E. Tanner and M. Deveney
Approved by:	S. Goldsworthy Science Leader – Marine Ecosystems
Signed:	
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ALL ENQUIRIES
South Australian Research and Development Institute - SARDI Aquatic Sciences
2 Hamra Avenue West Beach SA 5024
PO Box 120 Henley Beach SA 5022
P: (08) 8207 5400 **F:** (08) 8207 5415
E: pirsa.sardiaquatics@sa.gov.au **W:** http://www.pir.sa.gov.au/research

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Cover image: Mangroves at Port Gawler. Image by "Fairv8", CC BY-SA 3.0, via Wikimedia Commons (https://commons.wikimedia.org/wiki/File:Pt._Gawler_wharf.JPG)

EXECUTIVE SUMMARY

Local governments need to manage stormwater to protect residents and infrastructure from flooding and to minimise impacts of stormwater runoff on receiving waters. Flooding associated with the Gawler River system poses a risk to several areas and is a constraint to development in Northern Adelaide. The Gawler River Floodplain Management Authority (GRFMA), in partnership with the Stormwater Management Authority of South Australia and the Department for Environment and Water is developing a stormwater management plan (SMP) for the Gawler River to address stormwater and flooding risks. As part of this SMP, habitats and key species of the receiving waters from the Gawler River catchment need to be identified, and risks posed by stormwater runoff to these habitats and species assessed.

The Gawler River discharges into Gulf St Vincent (GSV) at Port Gawler, which is at the northern end of Barker Inlet. Available data sources were reviewed to determine the marine habitats and key species within the receiving waters of the Gawler River outfall. The general risks of stormwater to these habitats and species were reviewed, including potential impacts of nutrients, sediment, and other common stormwater pollutants.

The area in the immediate vicinity of the Gawler River outfall predominantly comprises mangroves with some saltmarsh, while in the broader area dense seagrass occurs subtidally. The mangrove, saltmarsh, and seagrass habitats of the region support a range of fauna, including important fisheries species, threatened shorebirds, and a population of bottlenose dolphins.

Nutrient, freshwater and sediment inputs from stormwater may favour mangroves and introduced coastal plant species over saltmarsh. Nutrient inputs to GSV could adversely affect seagrass through promotion of epiphytic growth and may encourage the growth of introduced or other nuisance algae or of toxic phytoplankton, while sediment may impact seagrass through light reduction due to turbidity. Organisms living in GSV in the vicinity of Port Gawler could be directly impacted by a range of stormwater pollutants, including metals, PAHs, and organic contaminants.

Keywords: Stormwater, Nutrients, Sediments, Pollution, Mangroves, Seagrass.

1. INTRODUCTION

1.1. Background

Local governments need to manage stormwater to protect residents and infrastructure from flooding and to minimise impacts of stormwater runoff on receiving waters. Flooding associated with the Gawler River system poses a risk to several areas and is a constraint to development in Northern Adelaide. A stormwater management plan (SMP) is being developed for the Gawler River to address flooding and other stormwater-related issues in the region.

The Gawler River Floodplain Management Authority (GRFMA) is developing the Gawler River SMP in conjunction with the Stormwater Management Authority of South Australia and the Department for Environment and Water, including the Green Adelaide and Northern and Yorke landscape boards. Preparation of the SMP requires information on the habitats and key species that occur in receiving waters of the Gawler River outflow, and the level of risk associated with stormwater outflows to the marine environment at the outflow.

Stormwater from the Gawler River enters Gulf St Vincent (GSV) at Port Gawler, which is located at the northern end of Barker Inlet (Figure 1). The Adelaide Coastal Waters Study (ACWS) considered anthropogenic impacts on the Adelaide Metropolitan waters of GSV (Fox *et al.* 2007). The ACWS determined that nutrients, particularly nitrogen (N) from stormwater and wastewater, are likely to be responsible for broad-scale seagrass loss along the Adelaide metropolitan coast. Turbidity due to sediments carried by stormwater is also likely to have contributed to seagrass loss, especially in the near-shore zone (Fox *et al.* 2007). Nutrients and sediment loads are also implicated in the loss of large brown canopy algae from temperate reefs, and a shift to turf-dominated assemblages (Gorgula and Connell 2004; Turner 2004). The Adelaide Coastal Waters Quality Improvement Plan (ACWQIP), which considers both GSV waters and the Port River waterways (including Barker Inlet), has adopted the 50% reduction in sediment loads and a 75% reduction in N from 2003 levels targets recommended by the ACWS (McDowell and Pfennig 2011).

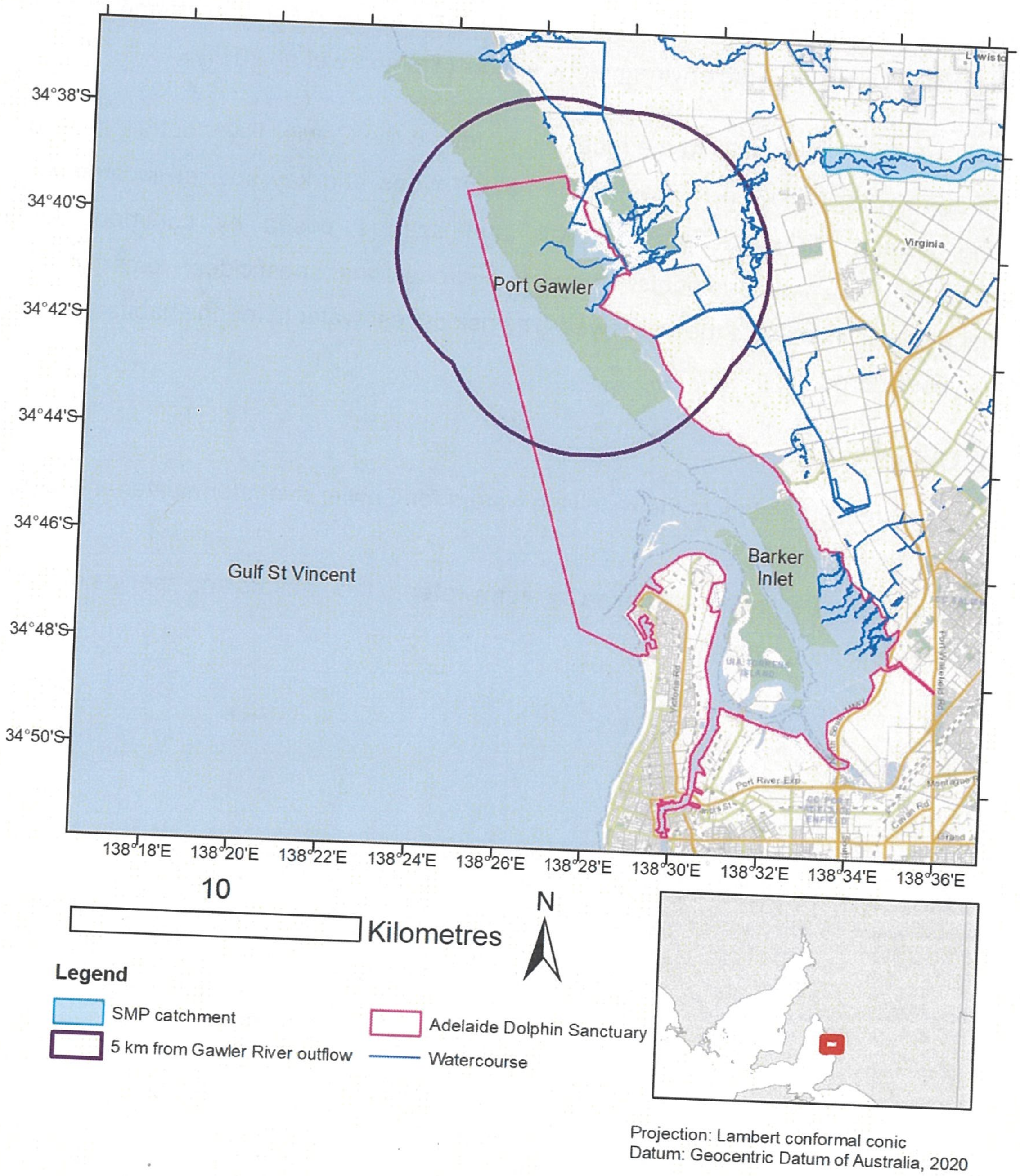

Figure 1. Map of study area showing locations mentioned in the text.

Heavy metals and other contaminants that are also carried in stormwater have periodically exceeded levels of concern in Adelaide waters; although they are not considered an important factor in historical seagrass decline (Fox *et al.* 2007), these pose a risk to receiving environments if present in sufficient concentrations (Mills and Williamson 2008; Gaylard 2009). Emerging organic contaminants, including endocrine disrupting chemicals (EDCs), pharmaceuticals and personal care products (PPCPs), and their metabolites, may be present in stormwater and have the potential to cause adverse environmental effects (Tremblay *et al.* 2011).

To determine stormwater risks to the receiving waters at the Gawler River outfall, available data and literature were reviewed to identify the habitat types and key species located within the receiving waters. Risks to these habitats and species posed by common stormwater contaminants: sediment, nutrients, metals, hydrocarbons, pesticides, emerging organic contaminants and litter, were then reviewed. The risk of freshwater to marine habitats and species was also considered.

1.2. Objectives

- Characterise habitats of the receiving waters for Gawler river stormwater and identify key species in the area.
- Identify and describe risks posed by stormwater to the habitats and species of the Gawler River receiving waters.

2. METHODS

The region of interest was taken to be contiguous water bodies within 5 kilometres of the Gawler River outfall, including the intertidal and subtidal. The Gawler River discharges through a small delta at Port Gawler, consequently a 5 km area encapsulating the discharge outfall was considered rather than the point source of the main channel Stormwater discharged from the Gawler River enters the marine environment of GSV at the northern end of Barker Inlet (Figure 1).

Information on marine benthic and intertidal habitats surrounding the study area was collated from existing data sources and a review of published literature. Data sources used in the assessment include benthic habitat classifications from the Department of Environment and Water (DEW, formerly Department for Environment and Heritage) for the Adelaide and Mount Lofty Ranges NRM region (DEH 2008b; Bryars 2013), mapping of coastal fisheries habitats (Bryars 2003) and data collected by SARDI Aquatic Sciences during seagrass monitoring of the Adelaide coast (Bryars and Rowling 2008) and Port River region (Tanner *et al.* 2014; Tanner 2020). The locations of these data points are shown in Figure 2.

Literature was reviewed to obtain further information on the habitats in the study area and to identify key species occurring in the area, including commercially and recreationally fished species, introduced species, and species of conservation concern. Potential impacts of stormwater on the habitats and species occurring in the area were identified and risks assessed by a review of additional relevant literature.

3. RESULTS AND DISCUSSION

3.1. Habitats and species of the Gawler River receiving waters

3.1.1. Habitat composition

Seagrass dominates the area, comprising 79.9% of the habitat within 5 km of the Gawler River outflow (Figure 2). Intertidal mangroves and saltmarsh comprise 10.3% and 5.0% of the total habitat respectively, with 4.8% being bare substrate (Figure 3).

3.1.2. Habitat structure and additional species

Key habitat forming species

The dominant seagrasses in the Gawler River outflow area are the meadow-forming taxa *Posidonia* spp. and *Amphibolis antarctica* (SARDI data). *Posidonia* spp. are difficult to distinguish to species from the available video data, but the main species known to form meadows in the Adelaide region are of the *Posidonia australis* group, comprising *P. australis, P. sinuosa* and *P. angustifolia*, with *P. australis* mainly restricted to shallow water (Bryars *et al.* 2008). *Amphibolis* dominated at some deeper video sites in the north-west of the area of interest, with *Posidonia* dominant in remaining areas (Figure 4; SARDI data[1]). *Posidonia* was the only seagrass recorded in transect data from 2020, while *Posidonia* dominated the transect in 2014 with a small (< 5%) contribution from *Zostera* spp. (Tanner *et al.* 2014; Tanner 2020). Small patches of *Zostera* spp. seagrass were recorded within *Posidonia* beds at four seagrass video sites but similarly comprised a small proportion (< 3%) of total seagrass cover in each case (SARDI data[1]). *Zostera* spp. are also difficult to identify from video, but *Z. nigricaulis* (formerly *Heterozostera tasmanica*) and *Z. muelleri* are the species most common in SA (State Herbarium of SA data). Seagrass cover in the Gawler River outfall region is patchy with medium to dense cover inshore, with continuous medium to dense cover further offshore and at the northern end of the area of interest (SARDI[1] and DEW data; Figure 3). Overall cover of seagrass was similar between 2014 and 2020 transects and seagrass condition was good in both years, but with improved condition in 2020 due to a reduction in epiphyte cover (Tanner 2020). Recent analysis of satellite data, however, suggest that some areas of dense seagrass have been lost off the Port Gawler coast since 2013 (Fernandes *et al.* 2022). This area of loss is directly offshore from the Gawler River outfall, to the north of the transects recorded in 2014 and 2020.

[1] Data from Bryars and Rowling (2008)

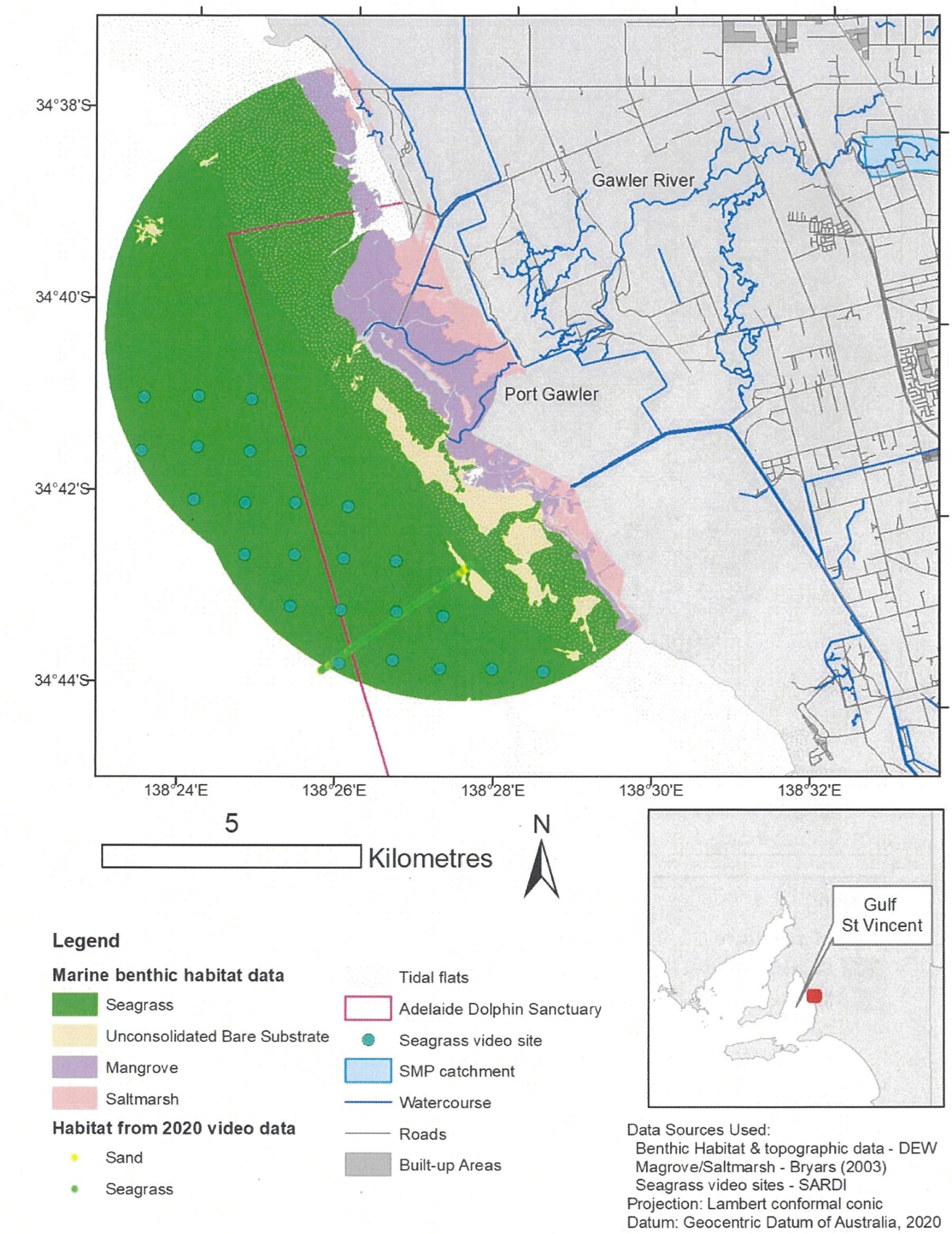

Figure 2. Map of study area showing marine benthic habitat classification and video transect data for the Gawler River SMP area and location of additional seagrass video data points.

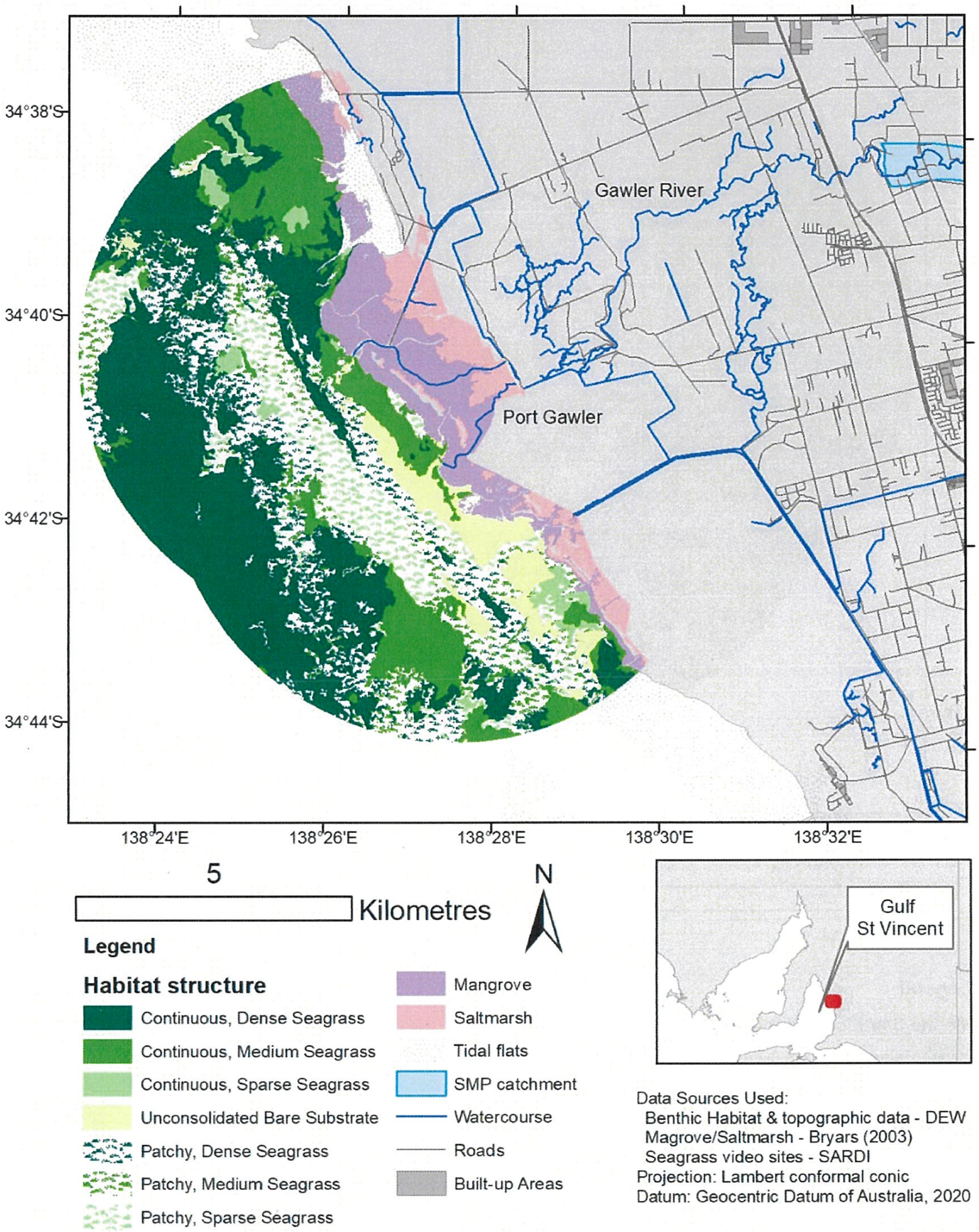

Figure 3. Map of study area showing marine benthic habitat structure and biota.

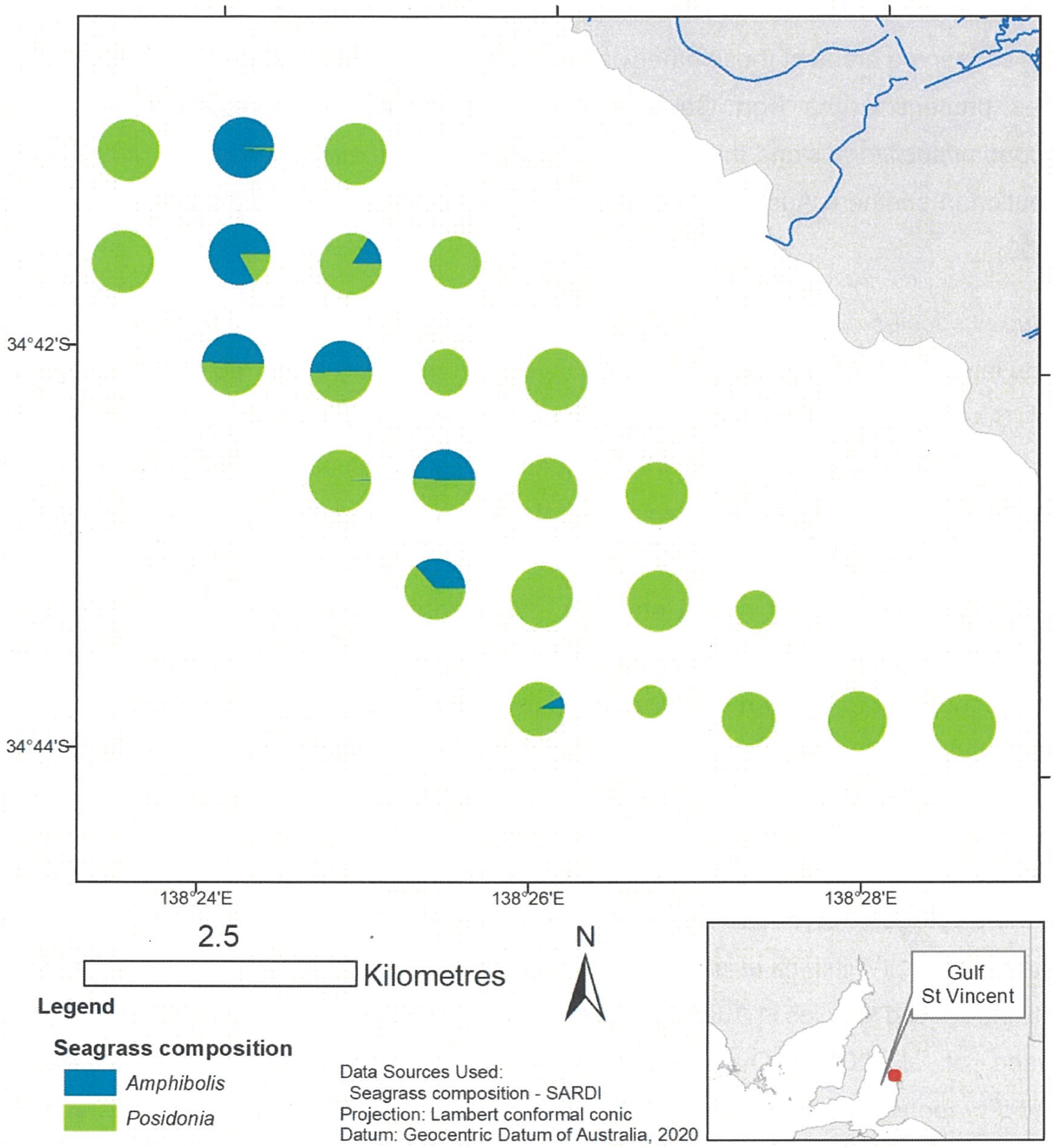

Figure 4. Composition of habitat forming seagrass from video site data. Each chart shows the relative contribution of *Amphibolis* and *Posidonia* to total seagrass cover, with size of each chart scaled to total seagrass cover of each site (range 33 – 100%; Bryars and Rowling 2018 data).

The only mangrove species occurring in SA is the grey mangrove, *Avicennia marina*, with SA populations believed to comprise the same subspecies (var. *marina*) as Western Australia (Barker and Orchard 2008). Cover of mangroves in the Port Gawler region is continuous and dense (Johnston and Harbison 2005; Caton *et al.* 2009). Saltmarshes of southern Australia support high plant diversity and provide important habitat for feeding and reproduction of commercially important fish and crustaceans (Fotheringham and Coleman 2008; Caton *et al.* 2009).

Saltmarshes at Port Gawler consist of inter- and supra-tidal samphires and provide one of the largest conserved areas of these otherwise depleted habitats in the state (Caton *et al.* 2009). Key species present in the Port Gawler saltmarshes include the threatened bead samphire, *Tecticornia flabelliformis* and the coastal bitter-bush *Adriana quadripartita*, which has a restricted distribution in southern Australia and is at risk from development and invasive weeds (Caton *et al.* 2009).

Introduced species

Several introduced species of macroalgae have been recorded in the region, including the green algae: *Codium fragile* ssp *fragile, Cladophora prolifera* and *Ulva fasciata,* and the red alga *Antithamnionella spirographidis*; a further 9 species of green algae, 3 brown and 8 red algae found in the area are considered cryptogenic, i.e., of uncertain native or introduced origin (Wiltshire *et al.* 2010; Baker and Gurgel 2011). The invasive macroalgae *Caulerpa taxifolia* and *Caulerpa cylindracea* occur in the Port River – Barker Inlet system, with *C. cylindracea* recorded as far north as the Port Gawler region (Wiltshire 2010; Wiltshire *et al.* 2010), which also provides suitable conditions for growth of *C. taxifolia* (see Deveney *et al.* 2008). Several introduced and potentially toxic dinoflagellates are also recorded in the Port River – Barker Inlet, including bloom-forming species of *Alexandrium* (see Wiltshire *et al.* 2010).

The priority introduced fanworm *Sabella spallanzanii* is recorded from the coast at Port Gawler, while the cryptogenic bryozoa *Amathia verticillata* and *Bugula neritina* occur within saltmarsh and mangrove areas (Wiltshire *et al.* 2010). The green shore crab *Carcinus maenas,* another high priority introduced species in Australia, is common in the Port Gawler region (Dittmann *et al.* 2016; Dittmann *et al.* 2017). Over 2019 – 2020, three individuals of the introduced crab *Charybdis japonica* were caught in GSV, including one near Port Gawler, but this species does not appear to have established in South Australia (Wiltshire *et al.* 2020; Wiltshire *et al.* 2022). A number of introduced coastal plants also occur in the area including Kikuyu (*Pennisetum clandestinum*) and berry seablite (*Sueda baccifera*); and introduced plants are a conservation risk for the important samphire habitats at Port Gawler (Caton *et al.* 2009).

Important native species

The Port River-Barker Inlet system supports a resident pod of approximately 30 bottlenose dolphins (*Tursiops truncatus*), and the Adelaide Dolphin Sanctuary, which includes the Port River, Barker Inlet, North Haven marina and extends to Port Gawler, was declared to protect the habitats of this species in the area (AMLRNRMB 2013).

GSV is recognised as a nationally important site for shorebirds, supporting 52 species, including 37 migratory species (DEWNR 2018; Purnell 2018; Lees *et al.* 2020). The Adelaide International Bird Sanctuary (AIBS), which extends along the coast from Barker Inlet to Port Gawler, is the southern end of the East Asia-Australasian Flyway (EAAF) and is utilised by more than 27,000 migratory birds in the peak summer season (DEWNR 2018). Mangrove and saltmarsh of the region also support many waterbirds, including the culturally important Australia pelican (*Pelecanus conspicillatus*) and black swan (*Cygnus atratus*), and important predatory species such as osprey (*Pandion haliaetus*) and white-bellied sea eagle (*Haliaeetus leucogaster*) (Caton *et al.* 2009; DEWNR 2018).

3.1.3. Values and ecosystem services of key habitats

Seagrass meadows are globally recognised as important coastal habitats; they are a key habitat for many fish species as a nursery, feeding, and breeding ground. Additionally, seagrasses trap sediment and stabilise the seabed with their extensive root and rhizome systems, baffle water flow (hence reducing erosion), and provide important ecosystem services such as nutrient cycling and gas regulation, i.e., providing oxygen and removing carbon dioxide (Short and Neckles 1999; Scott *et al.* 2000; Duarte *et al.* 2004; Orth *et al.* 2006). Seagrass communities consist of both seagrass species and a wide array of macroalgae and sessile invertebrates (e.g. sponges and bryozoa) growing on (epiphytes) or intermingled with the seagrass, and a large range of motile fauna. This diverse assemblage forms the basis of many food webs (Jernakoff and Nielsen 1998; Short and Neckles 1999).

Mangroves and salt marshes are important coastal ecosystems for fisheries, wildlife, nutrient cycling and global carbon balance (Weisberg and Lotrich 1982; Zann 1995; Kristensen *et al.* 2008; Nagelkerken *et al.* 2008; Walters *et al.* 2008). These habitats also provide important physical protection of the coast from waves, tidal bores, and tsunamis, and thus dampen shoreline erosion (Alongi 2008).

Low-level salt marsh forms an important food resource due to the daily tidal flushing which allows fish to harvest abundant invertebrates in this zone (Weisberg and Lotrich 1982). Salt marsh habitats in the region are utilised by a range of crustacea and several fish species, including congolli (*Pseudaphritis urvillii*), a diadromous fish species classed as vulnerable in the Adelaide region (Connolly *et al.* 1997; DEH 2008a). Fisheries species of the region do not directly utilise salt marsh habitats as extensively as they do mangroves and seagrass, potentially due to relatively short periods of inundation (Connolly *et al.* 1997; Bloomfield and Gillanders 2005).

Indirect benefits of salt marshes for these species are, however, likely, including through support of food chains and ecosystem services such as sediment stabilization (Bryars 2003; Bloomfield and Gillanders 2005).

Mangroves and salt marshes also provide important habitat for foraging birds and migratory shorebirds (McComb *et al.* 1995; Harbison 2008; Lees *et al.* 2020). Shorebirds, while all classed in the order Charadriiformes, are a diverse group of species, characterised predominantly by their reliance on wetland habitat (Purnell 2018). A high proportion of shorebirds are threatened, with habitat loss being a key risk (Lees *et al.* 2020). Several species of conservation concern utilise the AIBS (DEWNR 2018; Lees *et al.* 2020), including the IUCN listed endangered Lesser Sand Plover (*Charadrius mongolus*), vulnerable Greater Sand Plover (*Charadrius leschenaultia*) and Black-tailed Godwit *(Limosa limosa*); and the following listed as nationally threatened under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act):

Critically endangered

- Bar-tailed Godwit subspecies *menzbieri* (*Limosa lapponica menzbieri*)
- Great Knot (*Calidris tenuirostris*)
- Curlew Sandpiper (*Calidris ferruginea*)
- Eastern Curlew (*Numenius madagascariensis*)

Endangered

- Red Knot (*Calidris canutus* ssp. *rogersi* and *canutus*)
- Lesser Sand Plover (*Charadrius mongolus*)

Vulnerable

- Bar-tailed Godwit subspecies *baueri* (*Limosa lapponica baueri*)
- Greater Sand Plover (*Charadrius leschenaultii*)
- Hooded Plover (*Thinornis cucullatus*)

In addition to the internationally and nationally recognised species, a number of birds that occur in the AIBS are recognised as threatened under South Australia's *National Parks and Wildlife Act 1972*, including the endangered and vulnerable species of resident shorebirds: fairy tern (*Sternula nereis*) and banded stilt (*Cladorhynchus leucocephalus*), the vulnerable waterbird: freckled duck (*Stictonetta naevosa*), and vulnerable and endangered terrestrial birds: osprey (*Pandion haliaetus*), white-bellied sea eagle (*Haliaeetus leucogaster*) and the GSV subspecies of slender-billed thornbill (*Acanthiza iredalei rosinae*), also known as the samphire thornbill (DEWNR 2018). Saltmarshes of the Port Gawler area provide valuable habitat for the samphire thornbill

(Caton *et al.* 2009). Mangrove and saltmarsh in Barker Inlet also support the only known breeding colony of the threatened (rare) little egret, *Egretta garzetta*, in South Australia (Johnston and Harbison 2005).

Mangroves are known to provide essential nursery and feeding areas for many fish and invertebrates, including several species that utilise seagrass habitats at different life stages (Tanner and Deakin 2001; Bloomfield and Gillanders 2005; Harbison 2008). Seagrass and mangrove areas of the Gawler River outfall region are recognised as important habitats for several fish species, including the commercially important King George whiting (*Sillaginodes punctata*), yellow-eye mullet (*Aldrichetta forsteri*) and sea garfish (*Hyporhamphus melanochir*) (see Jones *et al.* 1996; Bloomfield and Gillanders 2005; Harbison 2008; Jones *et al.* 2008), and the blue swimmer crab, *Portunus armata,* a commercially and recreationally important fishery species in GSV (Johnston and Harbison 2005; Tanner 2007). Seagrass in the region also provides habitat for protected sygnathid fishes (Connolly 1994).

3.2. Potential risks from stormwater

Potential risks from stormwater to the marine environment are: increased suspended sediments, which have impacts through light reduction (turbidity) and sedimentation; nutrients; other contaminants such as metals, pesticides, hydrocarbons, and emerging organic contaminants; and reduced salinity due to freshwater inputs (Gaylard 2009). The ACWS and other investigations on the Adelaide coast and within GSV have demonstrated negative impacts to reef and seagrass habitats, particularly from sediments and nutrients (Gorgula and Connell 2004; Turner 2004; Fox *et al.* 2007; Gorman and Connell 2009).

The general risks to the environments receiving stormwater from the Gawler River are discussed below, with a focus on impacts on the benthic habitat forming species that support these environments. Motile species can often move to escape contaminants but may be impacted through loss or degradation of habitat. Where direct impacts on fauna are possible, however, these are also discussed.

3.2.1. Suspended Sediments

Sediments carried by stormwater are the main cause of turbidity in shallow waters (< 5 m) along the Adelaide coast. Because discharged stormwater in this area tends to move along-shore with minimal mixing with deeper water, turbidity may persist for several days, increasing impacts on near-shore habitats (Fox *et al.* 2007; Gaylard 2009). Turbidity increases light attenuation and scattering, leading to a lesser proportion of light penetrating to a given depth (Collings *et al.*

2006b). Light limitation has negative impacts on seagrass including reducing maximum depth range for growth (Abal and Dennison 1996), and causing decreased biomass, shoot density and productivity, and depletion of starch resources, an indication of poor condition (Ruiz and Romero 2001, 2003; Mackey *et al.* 2007). Macroalgae are similarly impacted by light reduction due to turbidity (Turner and Collings 2008; Gaylard 2009). Even in areas where average light intensity is in the range sufficient for seagrass growth, variability in available light due to periodic sediment influxes may reduce productivity and contribute to loss of seagrass (Collings *et al.* 2006b). Interactive effects between turbidity and nutrients may also contribute to seagrass loss and shifts in benthic community composition (De Casabianca *et al.* 1997; Wear *et al.* 2006).

Sediments also have impacts through siltation, e.g., restricting light due to sediment accumulation on the surface of seagrass leaves, or by preventing gas exchange and smothering plants (Ralph *et al.* 2006). Burial of shoots and seeds, and erosion by sediment movement can also cause damage to or loss of seagrass (Marba and Duarte 1995; Preen *et al.* 1995; Duarte *et al.* 1997; Bryars *et al.* 2008). Smothering and/or erosion can also lead to loss of macroalgae and other biota, and sedimentation can also reduce the availability of hard substrate for these organisms, thereby reducing or preventing recruitment (Airoldi 2003). Sedimentation can cause changes in unvegetated soft bottom habitats by altering sediment structure, smothering, or burying organisms, and clogging gills and filter feeding structures (Mills and Williamson 2008; Gaylard 2009). Mangroves may beneift from sedimentation where sediment accumulation creates suitable habitat for recruitment and growth (Ellis *et al.* 2004), but excessive sedimentation can result in smothering of mangrove roots, leading to reduced performance and potentially tree death (Ellison 1999). In mangrove habitats, increased sedimentation is associated with lower diversity of benthic invertebrates, particularly fewer bivalve and small crustacean species, and a shift to deposit-feeding species (e.g. detritivorous polychaetes, echinoderms), rather than suspension-feeding species (e.g. fan-worms, bivalves) (Ellis *et al.* 2004).

3.2.2. Nutrients

Wastewater effluent is a major source of nutrients entering GSV, but the contribution from stormwater is also substantial (Gaylard 2009; McDowell and Pfennig 2011). With inputs from industrial sources decreasing since 2013, reducing stormwater nutrient inputs remains a high priority to improve coastal water quality in GSV (McDowell *et al.* 2018). Elevated nutrients promote the growth of epiphytic algae on seagrass, eventually causing loss of above-ground seagrass biomass (Collings *et al.* 2006a; Bryars and Rowling 2008). High concentrations of water column nutrients can have acute toxic effects on seagrass (Collings *et al.* 2006a; Ralph *et al.* 2006) or promote microalgal blooms that reduce available light (De Casabianca *et al.* 1997; Ralph *et al.* 2006) and can lead to low oxygen availability (Gillanders *et al.* 2008).

Opportunistic green macroalgae such as *Ulva* spp. can also bloom in high nutrient conditions (Baker and Gurgel 2011; Imgraben *et al.* 2019), causing 'green-tides', in which large masses of macroalgae wash up on the shore and, if not removed, decompose, becoming anoxic and producing malodorous and toxic hydrogen sulphide gas (Smetacek and Zingone 2013). Excessive *Ulva* growth can be detrimental to mangroves by smothering pneumatophores (aerial roots important for gaseous exchange), causing loss of recruiting juveniles and stress in adult trees (Edyvane 1999; Harbison 2008). Macroalgal cover on intertidal flats can also negatively impact mangrove recruitment by preventing propagules anchoring to the sediment between tidal cycles (Clarke and Myerscough 1993). Nutrients also promote blooms of toxic microalgae such as *Alexandrium* spp. (Edyvane 1999) and growth of the invasive macroalgae such as *Caulerpa taxifolia, C. cylindracea* and *Codium fragile*. High nutrient levels favour the growth of these species over that of seagrasses and native macroalgae, facilitating intensification of invasive populations, further spread or potentially invasion of new areas (Ceccherelli and Cinelli 1997; Ceccherelli *et al.* 2002; Burfeind and Udy 2009; Gennaro and Piazzi 2014; Gennaro *et al.* 2015).

Sediment-bound nutrients have fewer toxic effects than water column nutrients, but in high concentrations these can lead to sediment anoxia and production of sulphides, both of which have negative impacts on seagrasses (Ralph *et al.* 2006). Nutrients and sediments have interactive impacts that are greater than either factor acting alone (Abal and Dennison 1996; De Casabianca *et al.* 1997; Gorgula and Connell 2004). The impact of nutrients is greatest in waters that are usually oligotrophic (relatively low in nutrients), such as those of GSV (Russell *et al.* 2005; Gorman *et al.* 2009; Bryars 2013).

Stormwater nutrients may benefit mangroves, but in saltmarshes, nutrient inputs from stormwater have been linked to increased abundance of introduced and invasive plant species and decreased cover of native saltmarsh, with associated freshwater inputs also favouring growth of exotic species (Geedicke *et al.* 2018).

3.2.3. Other Contaminants

Other contaminants often found in stormwater include trace metals; hydrocarbons, including polycyclic aromatic hydrocarbons (PAHs); pesticides (including insecticides, herbicides and fungicides); emerging organic contaminants such as pharmaceuticals, personal care products, endocrine disrupting chemicals and per- and poly-fluorinated alkyl substances (PFAS); and litter (Burton *et al.* 2000; Mills and Williamson 2008; Tremblay *et al.* 2011; Gaylard 2017). Stormwater also causes localised reduction of salinity (Mills and Williamson 2008; Gaylard 2009). Specific information on several of these contaminants is provided below; detailed aquatic toxicity data and guideline values for many toxicants are provided by the *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* (ANZG 2018), which update several guideline values of ANZECC and ARMCANZ (2000a, b).

Metals, hydrocarbons, and pesticides have acute and chronic toxic effects, and many accumulate in sediments or in tissues, leading to bioaccumulation and magnification through the food chain (Mills and Williamson 2008; Gaylard 2009). Many toxicants bind to sediment or organic matter and hence are found at highest concentrations in stormwater that also carries high sediment and nutrient loads and accumulate in depositional environments (Mills and Williamson 2008). Sediment-bound toxicants are generally less toxic to flora than soluble forms (Ralph *et al.* 2006) but can accumulate until they become acutely toxic to benthic fauna, e.g., flounder in a contaminated Auckland estuary showed evidence of poor health, including higher incidences of liver lesions than those from unpolluted sites (Mills and Williamson 2008). Sporadic pulses of contaminants, such as occur in stormwater discharges, can lead to greater toxic effects than exposure to constant concentrations, therefore, applying guidelines based on toxicity data from constant exposure studies could lead to underestimating risks associated with stormwater (Burton *et al.* 2000). Synergistic effects between co-occurring contaminants in stormwater can also lead to greater toxicity than exposure to a single toxicant alone (Burton *et al.* 2000).

Metals

Copper, lead, and zinc are the metals most commonly found at elevated levels in stormwater and are derived from road dust and roof run-off (Burton *et al.* 2000; Mills and Williamson 2008; Gaylard 2009). The concentrations of these metals in stormwater increase with the number of dry days preceding a given rainfall event. They have all been regularly recorded at above the ANZECC and ARMCANZ (2000a) trigger levels in Adelaide stormwater, although concentrations have decreased since the mid-1990s (Gaylard 2009). The ANZG (2018) guidelines provide a lower trigger value (8 *c.f.* 15 $\mu g\ L^{-1}$) for zinc than ANZECC and ARMCANZ (2000a) while trigger values for copper and lead are unchanged. Copper, lead, and zinc have many acute and chronic toxic effects, including on seagrass species and the canopy-forming macroalga *Ecklonia radiata* (see Gaylard 2009). Due to chemical similarity, non-essential metals can mimic required elements and bind to receptors, facilitating uptake, and then alter several biological processes (Gauthier *et al.* 2014).

Metals inhibit metabolic pathways, disrupt enzymes, and promote formation of reactive oxygenated compounds (Prange and Dennison 2000; Gauthier *et al.* 2014). In plants including seagrasses, metals have negative effects on photosynthetic processes, leading to reduced growth and potentially plant death (Prange and Dennison 2000). Although required for biological processes, essential metals exhibit toxicity when present in excessive concentrations (Gauthier *et al.* 2014).

Toxicity of metals depends largely on bioavailability, which is dependent on water chemistry and sediment organic content (Mills and Williamson 2008; Gaylard 2009). Copper and lead are most likely to be toxic in soft, acidic freshwater with low organic content. Increasing water hardness, alkalinity and pH, and natural dissolved organic matter (e.g., humic acids) generally reduce toxicity, but interactions are complex. Toxicity of lead is also reduced by chloride complexing in saline waters; lead bioaccumulates but is rarely present in sufficient quantities for this to occur (ANZECC and ARMCANZ 2000b; Mills and Williamson 2008; Gaylard 2009).

Zinc toxicity similarly decreases with increasing hardness, alkalinity, and salinity, but pH effects are not linear. Below pH 8, zinc toxicity increases with decreasing pH, with conflicting results found at higher pH. Zinc binds to clay and organic matter, but the effects of sediment-binding on zinc toxicity are variable. Copper and zinc are essential trace elements, and most organisms have mechanisms for regulating sub-lethal concentrations of these metals. They are therefore unlikely to bioaccumulate (ANZECC and ARMCANZ 2000b; Gaylard 2009).

Other metals that may be found in elevated concentrations in stormwater are cadmium, iron, chromium, nickel, antimony, platinum, and molybdenum (Mills and Williamson 2008). Several of these metals pose a risk to marine mammals due to biomagnification, with species at higher trophic levels being at greatest risk (Das *et al.* 2003). Cadmium is of concern in Adelaide metropolitan waters because it has been implicated in toxic effects observed in bottlenose dolphins (Lavery *et al.* 2009). Birds may also accumulate these metals, with resulting impacts including egg-shell thinning, developmental issues and neurological effects (Burger and Gochfeld 2001). Toxicity of the typical metals found in stormwater, and of other metals and metalloids, with the exception of platinum, is discussed in ANZECC and ARMCANZ (2000b) and guideline values are provided by ANZG (2018). Metals often bind to sediments and accumulate in depositional environments (Mills and Williamson 2008; Gaylard 2009). Mangroves trap fine sediments, and mangrove muds in the Barker Inlet region have high trace metal concentrations, with release of sediment-bound metals to the water column occurring during extended slack water periods due to the conducive pH and redox conditions that develop at these times (Harbison 1986).

Hydrocarbons

PAHs may be present in stormwater and are of concern due to their potential for acute toxicity and ability to bioaccumulate (Mills and Williamson 2008; Gaylard 2009. PAHs in stormwater are derived primarily from vehicle emissions, with some contribution from tyre wear (Mills and Williamson 2008). Typically, PAHs become toxic when they are metabolised because resulting derivatives include genotoxic, carcinogenic and reactive oxygenated compounds. Some PAHs are also directly toxic to aquatic organisms, primarily through damaging membranes and so affecting ion transport (Gauthier *et al.* 2014).

PAHs, especially longer-chained compounds, bind strongly to sediment, particularly fine sand (125-250 µm size fraction), and to organic matter (ANZECC and ARMCANZ 2000b; Mills and Williamson 2008). They accumulate in depositional environments such as estuaries (Mills and Williamson 2008). Lower molecular weight PAHs are more soluble but are removed by volatilisation and biological degradation, so are shorter-lived in aquatic environments (ANZECC and ARMCANZ 2000b; Mills and Williamson 2008). Exposure to UV light greatly increases the toxicity of PAHs due to creation of reactive oxygenated compounds (ANZECC and ARMCANZ 2000b; Mills and Williamson 2008).

The association of both metals and PAHs with fine sediment fractions means that these contaminants often co-occur, and emerging research shows that their combined impacts are often additive and sometimes synergistic (Gauthier *et al.* 2014). Cadmium, copper, nickel and zinc all show increased toxicity to aquatic organisms in the presences of at least some PAHs, possibly because PAH-induced membrane damage may increase uptake of metals (Gauthier *et al.* 2014).

Pesticides

Pesticides are often highly toxic and able to bioaccumulate and biomagnify through the food chain (Gaylard 2009). Organochlorine pesticides (OCPs) have largely been phased out because of these properties (ANZECC and ARMCANZ 2000b), but residues remain in the environment and can be found in stormwater, particularly in runoff from historically horticultural land (Mills and Williamson 2008; Gaylard 2009). The toxicity of OCPs is generally not affected by water chemistry, but some compounds are more toxic to certain species at higher temperature, e.g., > 20°C, compared with < 10°C (ANZECC and ARMCANZ 2000b). OCPs may contribute to reproductive disorders and cancer development in marine mammals (Murphy *et al.* 2018) and can contribute to disease through immunosuppression (Marsili *et al.* 2018). In birds, OCPs negatively impact egg-shell formation, and many of these chemicals can also have neurological effects, cause developmental abnormalities, or affect parental behaviour, leading to reduced reproductive success (Burger and Gochfeld 2001).

Organophosphorus pesticides (OPPs) include some currently widely used insecticides, e.g. chlorpyrifos and malathion, and the toxicity of these and other OPPs increases with temperature; chlorpyrifos is also more toxic at higher pH (9 c.f. 7.5). In general, OPPs are much more toxic to crustacea and insects than to algae, molluscs or fish, but within taxonomic groups species show widely varying sensitivities. Some OPPs bioaccumulate (ANZECC and ARMCANZ 2000b). Pyrethroid pesticides bind to suspended matter and biological films and are rapidly removed from the water column but may pose a threat to surface-feeding species such as cladocerans (ANZECC and ARMCANZ 2000b).

Herbicides are much more widely used than insecticides and are generally more toxic to seagrasses and algae than to fish or invertebrates because they inhibit photosynthesis (ANZECC and ARMCANZ 2000b; Gaylard 2009). The toxicity of some herbicides is increased at higher pH, while toxicity of others increases with temperature. Water chemistry and temperature have little impact on the toxicity of several compounds, but there is a lack of data for most herbicides, particularly for the marine environment (ANZECC and ARMCANZ 2000b; ANZG 2018) .

Emerging Organic Contaminants

Emerging organic contaminants are anthropogenic contaminants that have been rarely monitored but which can cause adverse environmental effects. These contaminants include endocrine disrupting chemicals (EDCs), pharmaceuticals and personal care products (PPCPs), and their metabolites (Tremblay *et al.* 2011). Many of these contaminants are not new, but analytical techniques were unavailable; advances in detection now allow their measurement (Tremblay *et al.* 2011). Analysis of historical samples has shown that the occurrence of some 'emerging' contaminants has decreased recently, but several do show an increasing trend, including the EDC triclosan, PFAS, and a range of PPCPs (Maruya *et al.* 2015). Wastewater is the major source of EDCs and PPCPs entering the environment because these compounds are not removed by current water treatment processes (Fernandes *et al.* 2010; Tremblay *et al.* 2011), but they may also occur in stormwater and industrial discharges (Tremblay *et al.* 2011). For example, PFAS have been recorded at relatively high concentrations in stormwater discharging into the Port River, Adelaide (Gaylard 2017). In urban areas, potential sources of emerging contaminants in stormwater are from surfactants, and leachates from solvents, plasticisers, pharmaceuticals and petroleum products (Tremblay *et al.* 2011).

EDCs affect the operation of endocrine systems and have the potential to disrupt hormone-controlled processes, including growth, immunity and reproduction (Porte *et al.* 2006). Impacts of EDCs have been demonstrated on a range of aquatic organisms, including bacteria, algae, invertebrates (echinoderms, molluscs and crustaceans), and fish (Porte *et al.* 2006; Fernandes *et al.* 2010; Tremblay *et al.* 2011), but the mechanisms of their actions are poorly understood (Porte *et al.* 2006). Some EDCs are also carcinogenic (Tremblay *et al.* 2011). Most EDCs are resistant to degradation in the environment and bioaccumulate and magnify up the food chain, posing a threat to higher trophic levels (Porte *et al.* 2006).

PPCPs include veterinary and human medicines, with antibiotic residues being of particular concern due to the potential for development of antibiotic resistance and impacts on important bacterial ecosystem processes such as decomposition (Tremblay *et al.* 2011). Many PPCPs have biological activity and may have impacts on enzyme systems or other biological processes. Acute toxicity and sub-lethal effects have been demonstrated for aquatic fauna exposed to individual chemicals, but there is concern that complex interactions between different chemicals occur, making impacts difficult to predict where multiple chemicals are present. Impacts on top predators may occur due to bioaccumulation or indirectly due to impacts on prey availability (Boxall *et al.* 2012).

PFAS are chemicals widely used in industry and household applications, including fire fighting foams, non-stick coatings, packaging and stain repellents (Hekster *et al.* 2003). These are persistent chemicals that bioaccumulate, with relatively high concentrations recorded in some cetaceans (Fair and Houde 2018; Marsili *et al.* 2018), incuding in dolphins in the Port River, Adelaide, where PFAS have been detected in stormwater (Gaylard 2017). PFAS exhibit a range of toxic effects in mammals including neurotoxicity, liver hypertrophy and endocrine disruption, although specific studies on marine mammals are lacking due to regulatory restrictions and logistical challenges (Fair and Houde 2018). Birds also accumulate PFAS, with these chemicals associated with decreased liver phospholipid content in chicks (Robuck *et al.* 2020). Ecosystem effects of PFAS contamination are not well understood, but toxic effects on aquatic fauna, including microalgae, mysid shrimp and minnows, have been demonstrated (Hekster *et al.* 2003).

Freshwater

Marine organisms have variable tolerances to salinities above and below their optimal range, and these tolerances vary within species depending on genotype, acclimation, and condition (Nell and Holliday 1988; Westphalen *et al.* 2005; O'Loughlin *et al.* 2006; Gaylard 2009). Seagrasses are relatively tolerant of periods of lowered salinity, but long-term exposure leads to reduced photosynthetic efficiency and eventually death (Westphalen *et al.* 2005; Touchette 2007).

Many macroalgae are also tolerant of short-term low salinity exposure, but tolerance varies greatly between species; estuarine and intertidal species typically tolerate broader salinity ranges than subtidal species (Kirst 1990; Rothausler *et al.* 2016). Fish and invertebrates that live in estuaries and intertidal zones similarly show greater salinity tolerance than subtidal species (Nell and Holliday 1988; O'Loughlin *et al.* 2006). Australian water quality guidelines recommend that changes to salinity in marine environments should be less than 5% of background levels (ANZECC and ARMCANZ 2000a).

Mangroves are adapted to estuarine conditions and able to grow at a range of salinities. *Avicennia marina* requires freshwater inputs, growing best at between 50 and 75% seawater, but seedlings do not survive long term at < 5% seawater (Santini *et al.* 2014; Nguyen *et al.* 2015). Freshwater inputs into saltmarsh areas tend to favour colonisation by mangroves at the expense of more salt-tolerant saltmarsh species (Geedicke *et al.* 2018).

Litter

Litter includes rubbish (plastic bags, bottles etc.) and organic waste. Most beach litter in GSV in a 2011 survey by number (79.7%) and mass (51.3%) was plastics, with glass and ceramic comprising 10.2 % of the total number of items or 8.5% by mass (Peters and Flaherty 2011). This survey was not specifically of material carried in stormwater, but it is likely that anthropogenic litter in stormwater has a similar composition, although with the contribution of plastic likely being greater now than in 2011 given globally exponential increases in plastic consumption over recent decades (Wilcox *et al.* 2015; Fossi *et al.* 2018).

Plastic waste and ropes have been widely implicated in causing environmental harm including deaths of marine birds, turtles, and mammals (Gaylard 2009; Peters and Flaherty 2011); while organic waste may cause oxygen depletion through microbial breakdown (Gaylard 2009) (Gaylard 2009). Risks of litter to marine mammals and birds include entanglement, leading to injury and possible strangulation, starvation or drowning, and ingestion risks (Kühn *et al.* 2015; Wilcox *et al.* 2015; Fossi *et al.* 2018). Potential effects of litter, and particularly plastics, on fish and invertebrates are recognised (Kühn *et al.* 2015; Thiel *et al.* 2018; Savoca *et al.* 2021). Plastic ingestion may occur in species that mistake litter for food items, or through uptake by filter feeders, ranging from baleen whales to mussels and barnacles (Kühn *et al.* 2015; Savoca *et al.* 2021). Ingestion may directly cause mortality through blockage of the gastrointestinal tract, but indirect imapcts are also common, including partial blockage leading to reduced feeding ability, and chemical effects (Kühn *et al.* 2015; Gallo *et al.* 2018) (Kühn *et al.* 2015; Gallo *et al.* 2018). Plastic waste, including microplastics, may contribute EDCs and other persistent organic contaminants to the marine environment (Gallo *et al.* 2018). If present in large quantities, litter can also damage nearshore benthic environments through smothering, shading and scour (Kühn *et al.* 2015).

3.3. Risks to habitats and species at the Gawler River outfall

Habitats in the immediate vicinity of the Gawler River outflow are most at risk because these receive largely undiluted outputs. The load and concentration of pollutants reaching marine environments away from the outflow will be determined by local hydrodynamics, but it is likely that contaminants will be rapidly diluted with distance. The habitat within 1 km comprises 38.8% seagrass, 36.9% mangrove, 15.7% saltmarsh and 8.7% bare substrate (Figure 5). Seagrass occurring in this area is likely to be either *Posidonia australis, Zostera* spp. or a mix of these, given that these taxa are the seagrasses that are most common in shallow to intertidal areas of the region (Bryars *et al.* 2008).

In addition to contributing to chronic nutrient effects on a wider scale, local impacts from stormwater nutrients, such as increased growth of opportunistic (e.g., *Ulva* spp.), and potentially introduced (e.g., *Caulerpa* spp.) algae, could occur in the vicinity of the Gawler River outflow and the area more generally if nutrient levels are too high. Mangroves, and particularly propagules, may be adversely affected by opportunistic algae due to potential smothering effects and impacts on recruitment, although nutrient inputs can also promote mangrove seedling growth. Nutrients may promote weed establishment in saltmarsh and nutrient inputs into shallow seagrass areas may also promote epiphytic growth, and seagrass could also be impacted through competition with opportunistic macroalgae.

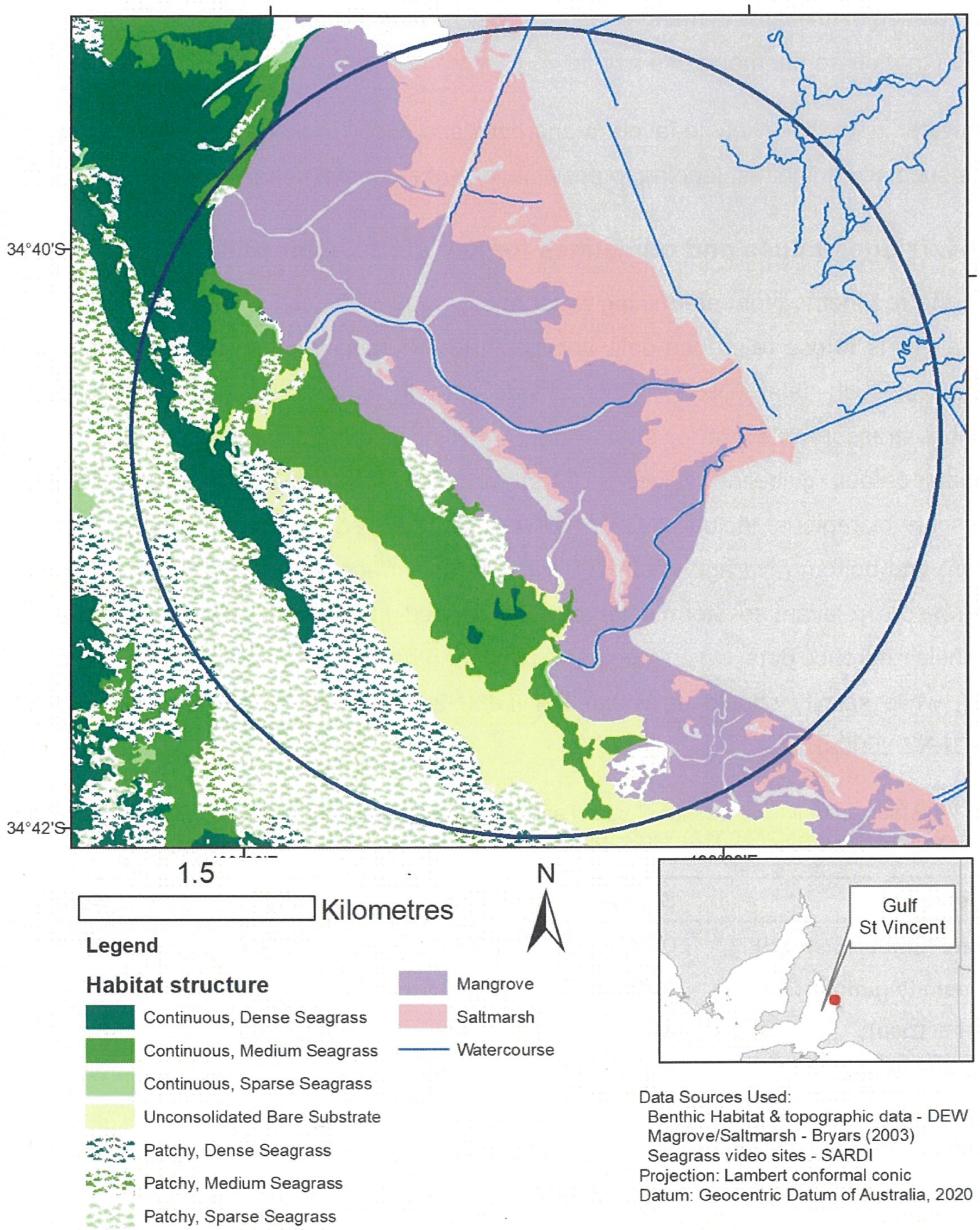

Figure 5. Map of habitats within 1 km of the Gawler River outfall.

Stormwater is likely to contribute to local turbidity, and, given the correlation between total suspended solids (TSS) and other contaminants (Mills and Williamson 2008), species using the habitats surrounding outfalls could be at risk of impacts from these pollutants, particularly metals.

Accumulation of toxicants in mangrove muds may have adverse effects on species such as crabs and fish that utilise the mangrove habitat.

Freshwater inputs are likely to favour mangroves and exotic species over salt-tolerant saltmarsh plants such as samphires, leading to possible loss of saltmarsh in the area.

3.4. Trigger values and guidelines for key stormwater pollutants

The preferred method for determining water quality trigger values for nutrients, suspended solids and salinity is to use reference data applicable to the specific ecosystem and area, but in the absence of such data, ANZG (2018) provide default trigger values for several physical and chemical stressors relevant for stormwater, including nitrate, phosphate, salinity and turbidity. Regional default guideline values (DGVs) for relevant stressors are available for several mesoscale bioregions, including GSV[2]. DGVs are provided by season and for surface (depth < 20 m) and bottom (average depth 200 m) waters. Surface water DGVs for the GSV bioregion for stressors relevant to stormwater are reproduced in Table 1. The DGVs reflect the 80th percentile reference data, which median values for nutrients and turbidity should fall below (ANZG 2018), while salinity should be within 5% (i.e., ~2 psu) of the reference level (ANZECC and ARMCANZ 2000a).

Table 1. Default guideline values for physical and chemical stressors associated with stormwater for surface waters of GSV (from ANZG 2018). Note: psu = practical salinity units, K_{d490} = diffuse attenuation coefficient.

Stressor	Summer	Autumn	Winter	Spring
Nitrate (μmol NO_3-N L^{-1})	0.337	0.351	0.504	0.239
Phosphate (μmol PO_4-P L^{-1})	0.197	0.185	0.301	0.348
Salinity (psu)	37.10	37.06	37.08	36.98
Turbidity (K_{d490})	0.101	0.130	0.114	0.091

The ANZG (2018) DGVs update the default guideline trigger values of ANZECC and ARMCANZ (2000a) , hereafter ANZECC, which, for GSV were the guidelines for “marine ecosystems in south central Australia - low rainfall areas - slightly disturbed habitats”. For nutrients, the ANZECC guideline values were 1 mg L^{-1} total nitrogen (TN) and 0.1 mg L^{-1} total phosphorus (TP). Nitrate and phosphate are the predominant components of TN and TP respectively. Assuming all TN and TP comprises these forms, the ANZECC trigger values are equivalent to 16 μmol NO_3-N L^{-1} and

[2]https://www.waterquality.gov.au/anz-guidelines/your-location/australia-marine-IMCRA/search-results?region=SVG

1.1 μmol PO_4-P L^{-1}, considerably higher than the ANZG (2018) DGVs. The ANZECC guideline values for TN and TP were based on limited data; EPA studies in South Australia subsequently demonstrated that likely nutrient impacts, including seagrass loss, have occurred in regions where nutrient concentrations were within the guidelines, indicating that the trigger values were too high to afford protection in South Australia's normally oligotrophic waters (Gaylard 2009), as reflected in the updated values.

Ambient water nutrient concentrations may also not be appropriate measures in productive environments, where nutrient loads may be high but are rapidly incorporated by algal growth (ANZECC and ARMCANZ 2000a; McDowell and Pfennig 2011). A 90th percentile water quality objective for TN of 0.2 mg L^{-1} (equivalent to 3.2 μmol NO_3-N L^{-1}) was therefore set by the Adelaide Coastal Water Quality Improvement Plan (ACWQIP) for the Port Waterways and Adelaide coastal waters, based on local data obtained by the EPA and through the ACWS (McDowell and Pfennig 2011). The ANZG (2018) guidelines suggest even that value is too high to afford sufficient protection to GSV. No specific objective was set for TP by the ACWQIP because phosphorus concentrations are generally low in Adelaide waters, and inputs from wastewater have been reduced (McDowell and Pfennig 2011), but the earlier Port Waterways WQIP (Pfennig 2008) included a target of < 0.025 mg L^{-1} TP (equivalent to 0.26 μmol PO_4-P L^{-1}), similar to the ANZG (2018) DGVs. Although phosphorus is not noted as of current concern in Adelaide waters, phosphorus inputs promote algal blooms where nitrogen is not limiting (Pfennig 2008; McDowell and Pfennig 2011).

Turbidity is correlated with TSS, but the relationship varies depending on the nature of the solids involved, making assessment of turbidity based on TSS measurements difficult (ANZECC and ARMCANZ 2000b), and no specific guideline value for TSS concentration was provided by ANZECC and ARMCANZ (2000a). ANZG (2018) provides DGVs for turbidity based on satellite diffuse attenuation coefficient (K_{d490}) data but determining how stormwater TSS relates to K_{d490} values is not straightforward. Some overseas jurisdictions specify maximum TSS concentrations of 25 mg L^{-1} or a maximum change from background levels of 10 mg L^{-1} (ANZECC and ARMCANZ 2000b). The objective for the ACWQIP is for TSS to be < 3 mg L^{-1} 90% of the time, thereby allowing higher turbidity following storm events (McDowell and Pfennig 2011).

Stormwater is recognised as the major source of TSS and other pollutants, especially metals, to Adelaide waters, and the ACWQIP calls for a reduction in stormwater input volume of 75% and of stormwater TSS by 80% from 2003 levels to assist in achieving the water quality objectives for

TSS and metals (McDowell and Pfennig 2011). TSS is a useful proxy for other contaminants in water quality modelling because concentrations of these, particularly metals and PAHs, are highly correlated with TSS (ANZECC and ARMCANZ 2000b; Mills and Williamson 2008). Recent modelling by SA Water (Van Gils *et al.* 2017) has also indicated that turbidity is still a major issue constraining seagrass recovery along the Adelaide coast, and that even when suspended solids settle out, they become resuspended during major storm events, further highlighting the importance of reducing these inputs as much as is feasible. There are no specific guidelines for gross pollutant loads, but reductions in litter and organic material are recognised as important for improving water quality (Pfennig 2008; Gallo *et al.* 2018).

4. SUMMARY AND CONCLUSIONS

Stormwater from the Gawler River SMP area discharges to GSV via a small delta at Port Gawler, at the northern end of Barker Inlet. The area in the immediate vicinity of the Gawler River outfall comprises mangroves with some saltmarsh in the intertidal to supratidal, and seagrass in the inter- to sub-tidal. Within 5 km of the outfall, there are extensive seagrass beds in the subtidal, with mangroves and saltmarsh in the intertidal regions. These habitats support a range of fauna, including important fisheries species, threatened shorebirds, and a population of bottlenose dolphins.

Nutrient, freshwater and sediment inputs from stormwater may favour mangrove growth at the expense of saltmarsh, which would also be at risk from establishment of introduced coastal plant species. Mangroves could be adversely impacted if stormwater nutrients promote excessive growth of opportunistic (e.g., *Ulva* spp.) or invasive (e.g., *Caulerpa* spp.) algae that could smother pneumatophores or negatively impact seedling recruitment on tidal flats. Nutrient inputs could adversely affect seagrass through promotion of epiphytic growth and may encourage the growth of introduced macroalgae or of toxic phytoplankton, while sediment may impact seagrass through light reduction due to turbidity. Species utilising the habitats of GSV in the vicinity of the Gawler River outfall could be directly impacted by a range of stormwater pollutants, including metals, PAHs and organic contaminants.

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