The Impact of Drain Discharges on Seagrass Beds in the South East of South Australia

Prepared for

South East Natural Resource Consultative Committee and

South East Catchment Water Management Board

by

Rachel J. Wear[1], Alison Eaton[2], Jason Tanner[1], Sue Murray-Jones[2]

[1] South Australian Research and Development Institute (SARDI Aquatic Sciences), [2] Coastal Protection Branch, Department for Environment and Heritage

SARDI Aquatic Sciences Publication No. RD04/0229-3

SARDI Research Report Series No. 129

The Impact of Drain Discharges on Seagrass Beds in the South East of South Australia

Prepared for

South East Natural Resource Consultative Committee and

South East Catchment Water Management Board

by

Rachel J. Wear[1], Alison Eaton[2], Jason Tanner[1], Sue Murray-Jones[2]

[1] South Australian Research and Development Institute (SARDI Aquatic Sciences), [2] Coastal Protection Branch, Department for Environment and Heritage

SARDI Aquatic Sciences Publication No. RD04/0229-3

SARDI Research Report Series No. 129

This Publication may be cited as:
Wear, R. J., Eaton, A., Tanner, J. E., Murray-Jones, S. (2006) The impact of drain discharges on seagrass beds in the South East of South Australia. Final Report Prepared for the South East Natural Resource Consultative Committee and the South East Catchment Water Management Board. South Australian Research and Development Institute (Aquatic Sciences) and the Department of Environment and Heritage, Coast Protection Branch, Adelaide. RD04/0229-3.

South Australian Research and Development Institute
SARDI Aquatic Sciences
2 Hamra Avenue
West Beach SA 5024

Telephone: (08) 8207 5400
Facsimile: (08) 8207 5481
http://www.sardi.sa.gov.au

Printed in Adelaide April 2006
SARDI Aquatic Sciences Publication No. RD04/0229-3
SARDI Research Report Series No. 129

Author(s):	Rachel Wear, Alison Eaton, Jason Tanner, Sue Murray-Jones.
Reviewers:	Doug Fotheringham (DEH), Dr Simon Bryars (SARDI); SECWMB; SENRCC; Mark de Jong (SEWCDB), Ron Ellis (Coastal Management Group of the SELGA).
Approved by:	Dr A. J. Fowler
Signed:	
Date:	Monday, 1 May 2006
Distribution:	Coast Protection Branch, Department of Environment and Heritage; South East Natural Resource Consultative Committee; South East Catchment Water Management Board; SARDI Aquatic Sciences Library; South Eastern Water Conservation and Drainage Board; Flinders University.
Circulation:	Public domain

TABLE OF CONTENTS

LIST OF FIGURE CAPTIONS

LIST OF TABLES

ACKNOWLEDGMENTS

The South East Natural Resource Consultative Committee and the South East Catchment Water Management Board provided SARDI Aquatic Sciences and the Department for Environment and Heritage, Coastal Protection Branch (CPB) with financial support for this project. The authors are indebted to Doug Fotheringham from CPB, who has provided support throughout the length of the project. We would also like to extend our thanks to Guy Williams and Ross Cole (CPB), Bruce Miller-Smith, Keith Rowling, Mandee Theil and Emma O'Loughlin (SARDI) for field support and Gen Mount, Erin Sautter and Michelle Roberts (SARDI) for their contributions to laboratory work. Ben Leonello and Guy Williams (CPB) assisted with collating and synthesising previous profile line data, and Jacquie Oates, Matthew Royal and Deb Canty (DEH, Environmental Information Analysis Branch) produced the habitat maps. Research was undertaken on *RV Rapid* (DEH) and *RV Odyssey* (SARDI). Simon Bryars and Greg Collings (SARDI) provided valuable technical and statistical advice. We also acknowledge Monash University and Australian Government Laboratories for undertaking nutrient and pesticide and herbicide analysis. Mark de Jong from the South Eastern Water Conservation and Drainage Board kindly collected drain water samples after the first flush of water reached the outlets in 2005, and together with Brenton Puddy provided information on drains in the South East for the report.

EXECUTIVE SUMMARY

Prior to the 1960s, much of the south east of South Australia was subject to inundation by water during winter, impacting on the productivity of agricultural land and transport through the region. Consequently, over a long period an extensive artificial drainage system was constructed. This system carries most of the excess surface water to the ocean, discharging at various locations along the coast. While terrestrial impacts from the drainage system have long been recognised, very little is known about what impacts the drain discharge may be having on the marine environment. The objectives of this collaborative project were to assess the health of seagrasses at major drain outlets, measure water quality parameters, and develop a long-term monitoring program to measure seagrass health and water quality at drain outlets.

Water within the drains was characterised by reduced salinities, increased turbidity, and high concentrations of nutrients and chlorophyll *a*, which appear to be rapidly assimilated into the marine environment. Notwithstanding this, seagrasses adjacent to the discharge points of drains in the South East appear to be impacted, as demonstrated by reduced seagrass leaf densities and leaves of reduced stature. The level of impact tends to reflect the volume of water discharged from the drains and the size of each drain and its associated subsidiaries. Drain M had a far more pronounced effect than the other drains investigated, with much of the seagrass adjacent to the drain discharge in northern Rivoli Bay already lost and with losses continuing. Seagrasses adjacent to the Maria Creek and Blackford Drain discharges also appeared to be affected by drain discharges. However, the area of impact is significantly reduced compared with that seen adjacent to Drain M. It is very difficult to determine which aspect of the drain water is contributing to reduced seagrass health in the vicinity of drain discharges. However, seagrasses are relatively tolerant of the individual effects of increased turbidity, reduced salinities and increased nutrients and it is likely that a combination of factors is affecting seagrass health.

One of the major concerns with regard to impacts on seagrasses is that initial seagrass loss in an area can destabilise the area and result in further losses, as is being observed in Rivoli Bay. The present study has revealed that the inshore limit of seagrass distribution in Lacepede Bay at Kingston has receded seaward by 84 m in the past 20 years and by 40 m near Butchers Gap in the past five years. To ensure that seagrasses in the vicinity of drain discharges in Lacepede Bay are not resulting in longer-term, and possibly irreversible effects, a monitoring program has been designed and its implementation is recommended.

Keywords: Drain discharges; seagrass; impact; South East Region; South Australia; Lacepede Bay; Rivoli Bay; Beachport; Kingston.

1. INTRODUCTION

1.1. Seagrass

Seagrass beds form a vital component of marine ecosystems. Not only do they increase the stability of the seafloor through the growth of extensive root and rhizome mats (Fonseca and Fisher, 1986), but they also play a critical role in the trapping and cycling of nutrients (Hillman *et al.,* 1989) and provide valuable habitat for a diversity of marine organisms, including commercially important fish and crustaceans, which utilise seagrass beds as nursery grounds (Bell and Pollard, 1989; Short and Wyllie-Echeverria, 1996; Connolly *et al.*, 1999; Duarte, 2002). Unfortunately, seagrass meadows are particularly sensitive to anthropogenic disturbances, which have commonly been associated with their decline in many countries (Cambridge *et al*., 1986; Shepherd *et al.*, 1989; Short and Wyllie-Echeverria, 1996). The inputs of urban and industrial wastes, and agricultural runoff, as well as coastal developments and aquaculture ventures are some of the activities that have been found to negatively impact on seagrass meadows (Short and Wyllie-Echeverria, 1996; Ralph *et al*., 2006) either through physical disturbance or through nutrient and sediment loading which can change water quality and/or clarity.

In the waters adjacent to the Adelaide metropolitan area, poor water quality has led to the loss of more than 5,200 ha of seagrass in the past five decades, with losses continuing (Seddon, 2002; EPA, 1998; Westphalen *et al.*, 2005a). Extensive losses have also been reported in other parts of Australia, including Cockburn Sound in Western Australia (Cambridge *et al.*, 1986; Shepherd *et al.,* 1989), Botany Bay in New South Wales (Shepherd *et al*., 1989; Larkum and West, 1990), and Westernport Bay in Victoria (Shepherd *et al.*, 1989; Keough and Jenkins, 2000). Given that seagrasses play an important role in stabilising sediments, initial losses of seagrasses in an area can lead to further losses via erosion and fragmentation of remaining meadows, or through a reduction in light as a result of increased turbidity due to sediment resuspension. While the severity and speed of seagrass decline is quite variable and can be quite rapid, rates of recovery are extremely slow, with poor establishment from seeds and slow growth of seedlings and established plants (Kirkman and Kuo, 1990; Marbà and Walker, 1999; Meehan and West, 2002; Bryars and Neverauskas, 2003; Meehan and West, 2004). In addition, seagrass rehabilitation is renowned for being time consuming, expensive and is not always successful (Fonseca *et al.*, 1998).

South Australia is known to support up to 13 species of seagrasses, belonging to seven different genera, including four species of *Posidonia* (*P. angustifolia, P. australis*, *P. coriacea*, *P. sinuosa*) and two species of *Amphibolis* (*A. antarctica* and *A. griffithii*). South Australia contains over 9,600 km^2 of seagrass habitat (Edyvane, 1999). While the majority of this seagrass inhabits waters of Gulf St Vincent and Spencer Gulf (82.7%), a small proportion

occurs in the shallow sheltered bays of the South East (2.7%)[1], including Lacepede Bay, Guichen Bay, Nora Creina, Stinky Bay, Rivoli Bay, Bucks Bay, Bungaloo Bay and in waters adjacent to the township of Port MacDonnell (Figure 1; Edyvane, 1999; Bryars, 2003). Seagrasses in the South East occur as large beds or as sparse meadows and are thought to be dominated by *P. australis*, *P. angustifolia*, *P. coriacea*, *P. sinuosa*, *A. antarctica* and *Zostera tasmanica* (Robertson, 1984; Edyvane, 1999).

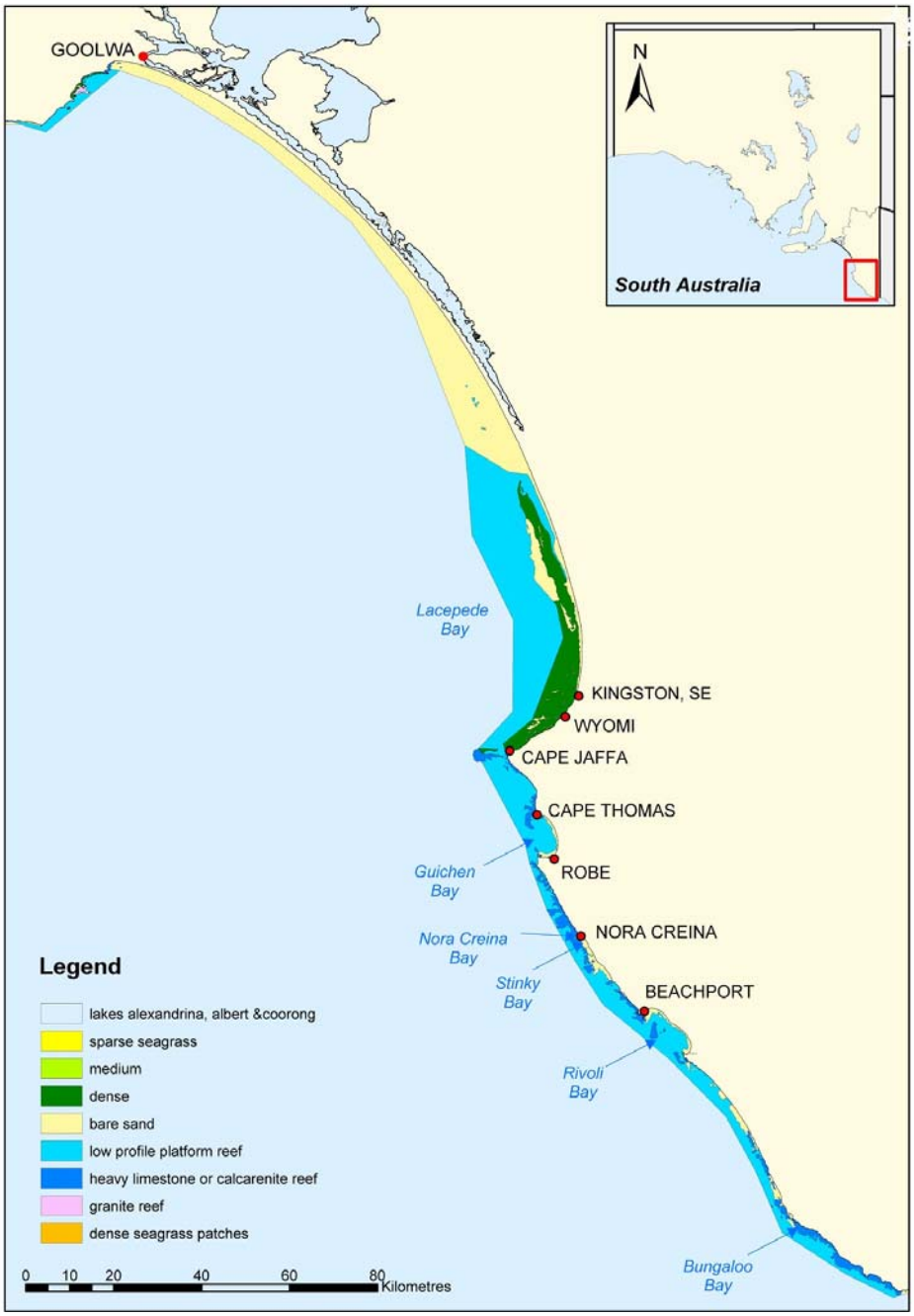

Figure 1. The distribution of seagrasses in the South East of South Australia. Small beds of seagrasses are not visible on this map, but occur in some of the smaller bays in the South East (Produced by Annette Doonan, SARDI Aquatic Sciences). Data source: Edyvane (1999).

[1] The South East Region extends from the Murray Mouth eastwards, to the Victorian border.

1.2. Drainage in the South East

The South East region consists of a series of 13 coastal dunes or ranges marking former shorelines that extend parallel to the existing coastline in a north-westerly to south-easterly direction, separated by large interdunal flats (Schwebel, 1983). The flats are at a progressively lower level from east to west and are slightly tilted from the south-east to the north-west, and consequently, the surface water moves west towards the coast until it diverts northwards along the eastern side of the adjacent ranges (SEDB, 1980). With virtually no outlet to the sea, the interdunal flats were once regularly covered with poorly defined watercourses, wetlands, swamps and lakes (Figure 2). A description of the area was made by the surveyor general George Woodroofe Goyder, in 1864, who wrote "*my opinion is that from Salt Creek southward, the area of the South-East is equal to 7,600 square miles and in every wet season half of that is under water. The depth of the water varies from 1 to 6 feet and some of it is never dry. Some swamps extend from 4 to 6 miles*" (Turner and Carter, 1989). At the time, up to 54% of land in the South East was subjected to inundation and flooding of water (SEWCDB, 2003-2006; Figure 2), and as a result, transport through the region was extremely difficult and cultivation of the land for agricultural use was impossible.

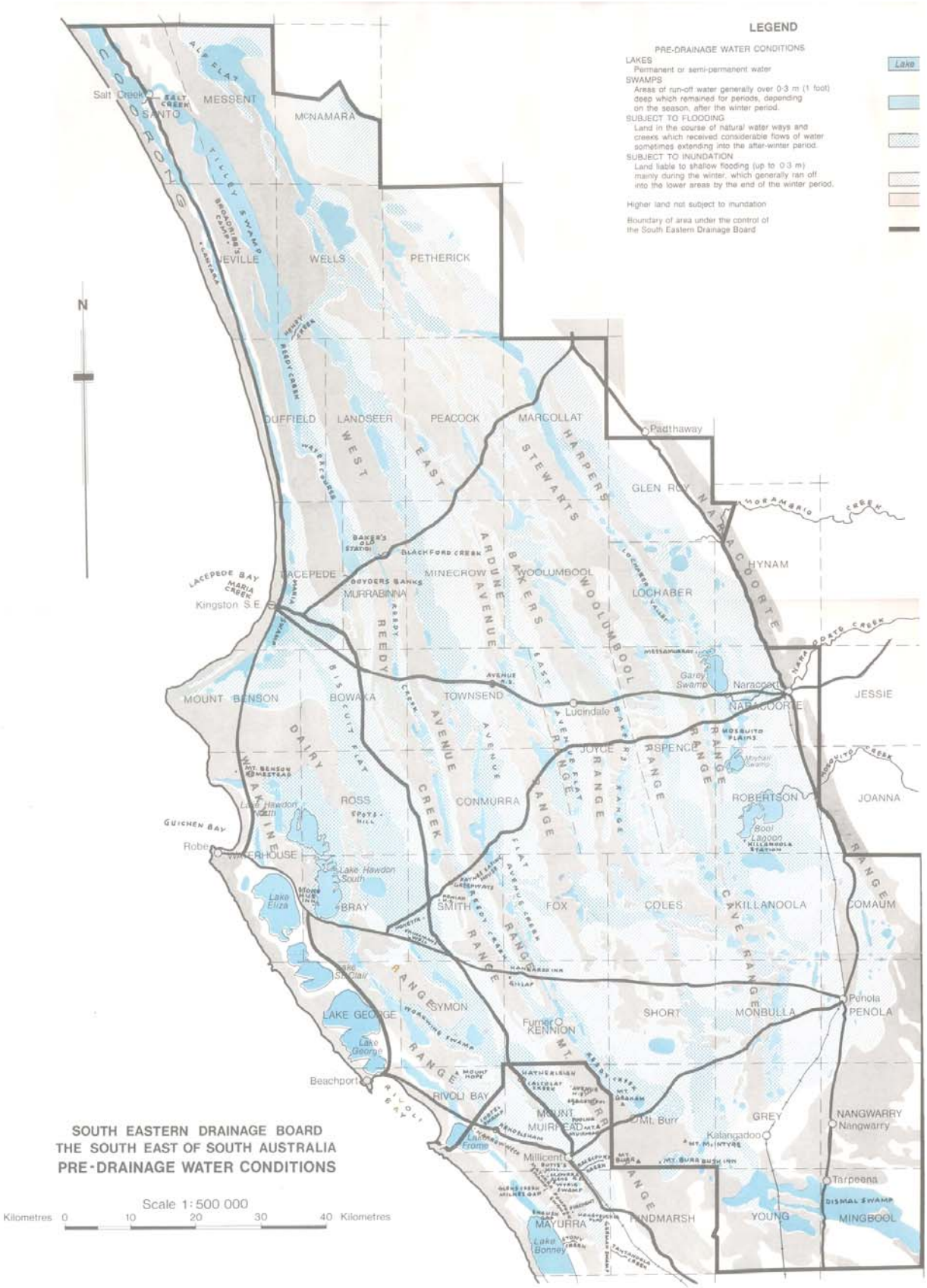

Figure 2. Pre-drainage water conditions in the South East region of South Australia (Obtained from Turner and Carter 1989).

In order to create land suitable for agricultural use and to improve transport in the region, artificial drainage in the South East commenced in 1863 with cuttings in the Woakwine Ranges to release water from Lakes Frome and Bonney. In the following two decades, around 40,000 ha of land was drained in the Millicent-Tantanoola area (Schrale *et al.*, 1987). The success of the Millicent-Tantanoola system led to the construction of four main drains and their tributaries in the middle South East. Approximately 2,000 km of artificial drains were constructed, diverting the north-westerly flowing water through the dune ranges to the coast, with work completed in the 1970s (D'Arcy *et al.,* 1984). Some drainage was accomplished by reconstructing natural watercourses. However, most effective drainage had occurred from construction of drains cut perpendicular to the stranded dune ranges (Holmes and Waterhouse, 1983). The benefits of constructing such an extensive drainage system were that 381,000 ha of flood prone land became available for grazing sheep and cattle and cultivating crops, although approximately 20,200 ha of this land along 159 km of main drains is believed to be affected by over-drainage (Schrale *et al*., 1987).

While the extensive drainage system of the South East has significantly increased the land available for agricultural purposes and improved transport within the region, development of the land has dramatically altered, and this has had significant effects on the terrestrial environment (SEDB, 1980). Impacts to the terrestrial environment have included a substantial reduction in native vegetation and a reduction in mammal species and abundance. Given a lack of reliable information on species of reptiles, fish, amphibians, crustaceans and molluscs in the South East prior to the drainage schemes and land development, the impacts to these animals are unknown. However, it is likely that drainage and subsequent development has also negatively impacted upon them (SEDB, 1980). While terrestrial impacts as a result of drainage and development have been recognised in the past several decades, potential impacts to the adjacent marine environment have only recently been considered (Seddon *et al.*, 2003).

1.3. Drains Discharging into Seagrass Meadows in the South East

The South East Region is represented by some of the most diverse and productive waters of South Australia and is characterised by the most significant nutrient-rich coastal upwellings along the southern Australian coastline (Edyvane, 1999). The presence of these upwellings contributes to the high algal diversity found in the region, as well as the productivity of local southern rock lobster and abalone fisheries (Edyvane, 1999). The diversity of the region can also be attributed to the variety of marine and estuarine habitats, including rocky shores, seagrass meadows, sandy beaches and dunes, estuaries and lagoons, sheltered embayments, and subtidal and offshore reefs (Edyvane, 1999). Given the importance of the region to the State's marine biodiversity and economy, it is imperative that we understand the effects of drainage and land development on these systems. The present project is one of the first projects aimed at assessing such impacts and is specifically aimed towards assessing the health of seagrasses in the vicinity of drain discharges. The project follows on from an

earlier study in which the loss of seagrass in Rivoli Bay, Beachport, was investigated (Seddon *et al*., 2003).

There are a number of activities and practices occurring in the South East that have the potential to impact on seagrass communities (e.g. finfish aquaculture, coastal developments), including the discharge of agricultural runoff via drains. The South East drainage system grew progressively between the 1860s and 1970s, and it is estimated that the combined length of the drains is approximately 2,000 km (Figure 3; SEDB, 1980). Eight of these drains are known to discharge into or near seagrass meadows; these being Blackford Drain, Maria Creek (or Kingston Main Drain), Butchers Gap Drain (all in Lacepede Bay), Mount Benson Drain (Guichen Bay), Drain M (Rivoli Bay) (Figure 3), and three small unnamed drains at Port MacDonnell.

Figure 3. Map of the South East region of South Australia, showing the extensive drainage system (Produced by Annette Doonan, SARDI Aquatic Sciences).

While the impact of drain discharges in the South East on seagrass meadows and their associated communities is largely unknown, some research has been undertaken in Rivoli Bay, adjacent to the Drain M discharge at Beachport, where seagrass losses have recently been reported (Hart and Clarke, 2002; Seddon *et al*., 2003). In 1951, aerial photographs revealed that the protected area of the Bay consisted of relatively dense, continuous, seagrass meadows. However, between 1951 and 1997, aerial photographs provide evidence of a consistent decline of this seagrass and recent surveys demonstrate that losses are continuing (Seddon *et al.,* 2003). Rapid erosion is occurring along margins of the remaining raised *Posidonia* beds, as a result of increased water movement. In particular, channels perpendicular to the shoreline have widened in several places and some sections of the seaward edge have retreated shoreward (Seddon *et al.,* 2003). Hart and Clarke (2002) estimated that in 1951, 25.8 ha of seagrass was present, but by 1997 this had reduced to just 6 ha (Figure 4). Seddon *et al.* (2003) re-evaluated these figures and determined that the seagrass area was originally 36.4 ha and was reduced to 7.7 ha in the same period (a total loss of 28.7 ha). The highest rate of decline took place during the 1960s and early 1970s and occurred at approximately 1.15 ha y^{-1} (Seddon *et al.,* 2003). While the present loss of seagrass in the area is almost certainly due to high wave energies and seabed instability, caused by a lack of protection from deeper *Posidonia* beds, earlier losses were more than likely due to discharges, with losses coinciding with the construction of Drain M (starting in 1949) and major works and enlargement of the Drain throughout the 1960s (Seddon *et al.,* 2003).

The seagrass loss at Beachport has reduced wave attenuation in northern Rivoli Bay, resulting in erosion of the seabed where seagrasses once grew. The change in sea floor shape and depth have also changed the angle at which the prevailing swell reaches the beach, and while the waves once came into the shore directly parallel to it, now the angle is more oblique, causing the orientation of the beach to change (Seddon *et al.*, 2003; Figure 5). To prevent coastal erosion, groynes have been constructed and as a result, sand appears to build up on the southern side of the groynes. In an effort to protect remaining seagrass patches from the prevailing conditions and almost certain loss, between mid to late 2003 a breakwater consisting of numerous sand-filled geo-textile bags was constructed on the southern side of the Beachport Jetty by the Coastal Protection Branch, Department for Environment and Heritage.

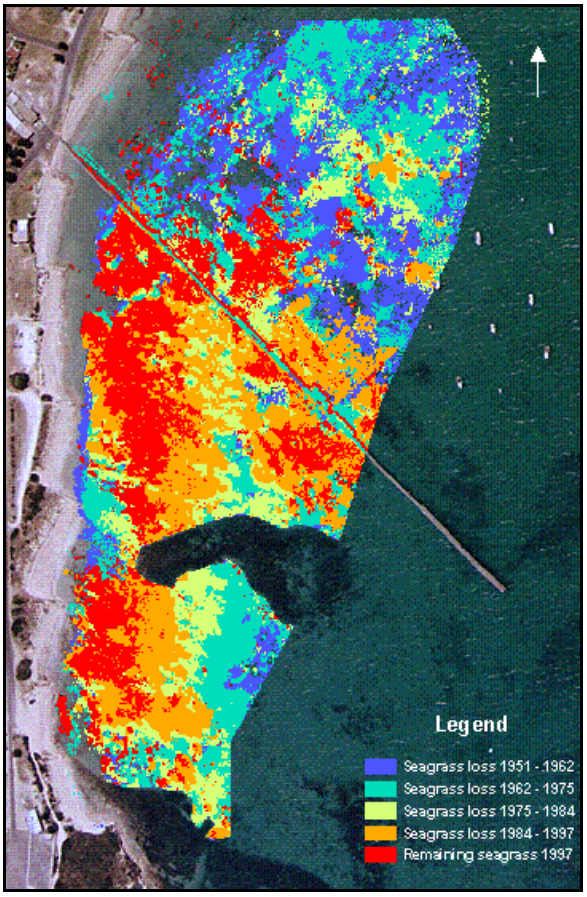

Figure 4. The loss of seagrass at Beachport between 1951 and 1997 (from Hart and Clarke, 2002).

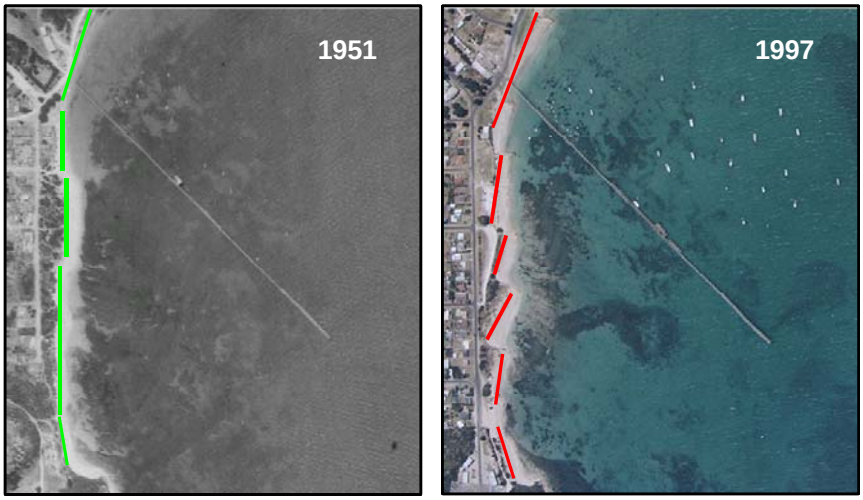

Figure 5. Beach orientation at Beachport in 1951 and 1997, following seagrass loss and various coastal activities such as groyne construction. Green and Red lines indicate the seaward limit of dune vegetation in 1951 and 1997, respectively (from Seddon *et al.,* 2003).

In addition to the loss of seagrass in northern Rivoli Bay, it is possible that seagrasses have also been lost in the southern end, adjacent to Southend. While there is no seagrass meadow present there now, there is historical evidence that seagrass meadow was present in the past. Firstly, an early nautical survey of Rivoli Bay in 1871 reported at Southend "*weeds or grass showing in places during fine weather similar to those observed in the northern end of the bay*" (Howard, 1871). Secondly, a 1956 Marine and Harbours Board survey line shows what appears to be a remnant seagrass bank. This feature was no longer evident when resurveyed by the Coast Protection Board in 1987 and 2005 (Figure 6). Loss of a seagrass bank may account for the coastal erosion that has been an issue at Southend for many years (D. Fotheringham, pers. comm.).

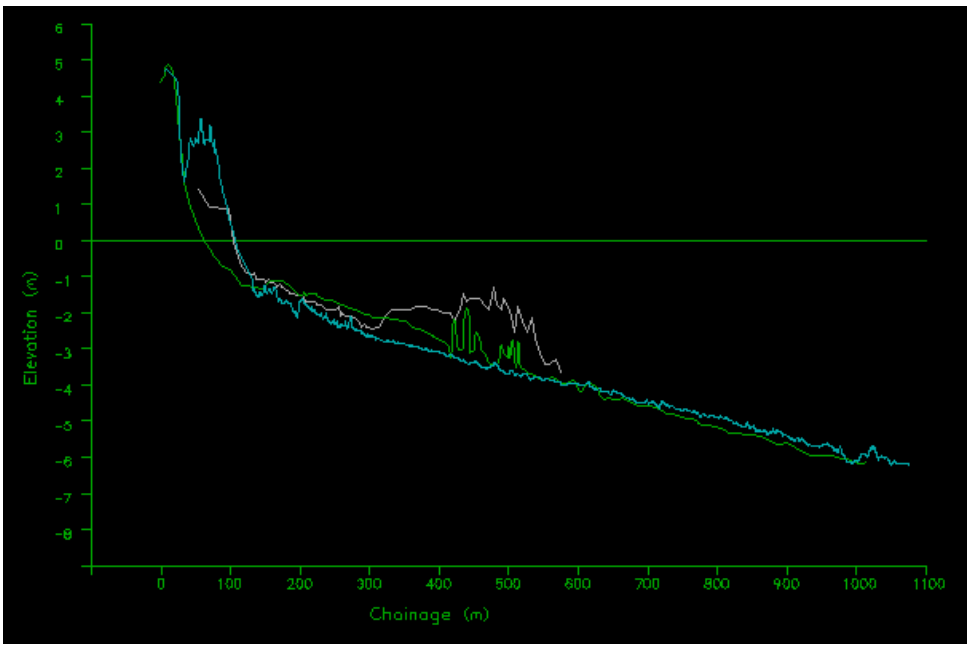

Figure 6. Seabed heights for profile line 725005 during 1956 (purple), 1987 (green) and 2005 (blue). Figure courtesy of the Coastal Protection Branch, Department for Environment and Heritage.

Previous studies investigating the impacts of drain discharges in Rivoli Bay, together with historical information on the area, have identified that they may have been responsible for, or at least contributed to, the loss of seagrass (Seddon *et al.*, 2003; D. Fotheringham, pers. comm.). Previous research has also demonstrated the importance of seagrass meadows for seabed stability and protection of the coastline. Seagrasses are also ecologically important as nursery and feeding areas for fish and crustaceans (Edyvane, 1999). There are concerns that if seagrass loss has occurred as a result of Drain M, then other drains in the South East have the potential to cause similar problems to those seen in Rivoli Bay. Hence, we must endeavour to understand the impacts of drain discharges on the seagrasses present in the South East. Of the eight drains that discharge directly into (or in the vicinity of) seagrass communities in the South East, four were investigated as part of the present study. These are Blackford Drain, Maria Creek and Butchers Gap Drain, which all discharge into the largest

seagrass meadows in the South East in Lacepede Bay, and Drain M, which discharges into Rivoli Bay (Figure 7). The drains vary in length, catchment size, the volume and quality of water they discharge, and the nature and composition of the seagrass habitats they discharge into.

Figure 7. Aerial photographs of the outlets to (a) Drain M, (b) Blackford Drain, (c) Maria Creek, and (d) Butchers Gap Drain, in the South East Region of South Australia. Photographs courtesy of the Coast Protection Board.

1.3.1. Drain M (Lake George Outlet)

Lake George is located adjacent to the Beachport township in the South East Region of South Australia. A network of constructed channels, including Drain M, currently transport excess surface water from the Wattle Range Catchment into Lake George and ultimately into the northern end of Rivoli Bay. Prior to the 1900s, there was no connection between Lake George and the ocean. However, in 1914, a narrow channel was cut from Lake George through to the sea (Figure 8). The outlet to the sea was completed by 1916, and since this time a number of other drains have been constructed (Table 1). Drain M is currently 75 km long and has approximately 210 km of major subsidiary drains (Drain M minor drains, Drain B, Drain C, Symon Main Drain, Killanoola Drain, Bellinger Swamp Drains and Southern Bakers Range Drain) (M. de Jong pers. comm.).

Figure 8. Cutting a channel through coastal dunes from Lake George to the sea in 1914. Photo courtesy of the Beachport Tourism Information Centre (from Seddon *et al.,* 2003).

Table 1. Time line of construction of Drain M. Summarised from SEDB (1980). Numbers in brackets represent the length of the construction.

Year	Construction
1863	Construction of drains began in the South East
1914	Lake George Outlet to the sea (0.4 km)
1916	Lake George Outlet to the sea completed (3.9 km)
1922	2/3 of Drain M constructed (34.2 km)
1949	Drain M to Lake George enlarged (0.7 km)
1963	Drain M enlarged to Legges Lane (23.2 km)
1964	Drain M extended to Bakers Range Drain (14.3 km)
1966	Drain M extended to Bool Lagoon (18.4 km)
1968	Mosquito Creek Channel + drains, completed (4.8 km)

Drain M is one of the largest drains in the South East and was built with the shortest possible distance between the sea and the inundated flats. The South Eastern Drainage Board has collected information on flow volumes within Drain M since 1972. Annual flow volumes since this time have varied from 2,194 ML (1997) to 186,179 ML (1989), and averaged 58,418 ML. As expected, flow within the Drain varies with annual rainfall in the area, and generally peaks between September and December each year (Figure 9). Monthly flow volumes since 1972 have averaged 4,868 ML and reached a maximum of 94,272 ML in September 1981 (Figure 10). At the time of the sampling during the present study in October 2004, the highest monthly volume since 1991 was recorded (40,577 ML) (Figure 10).

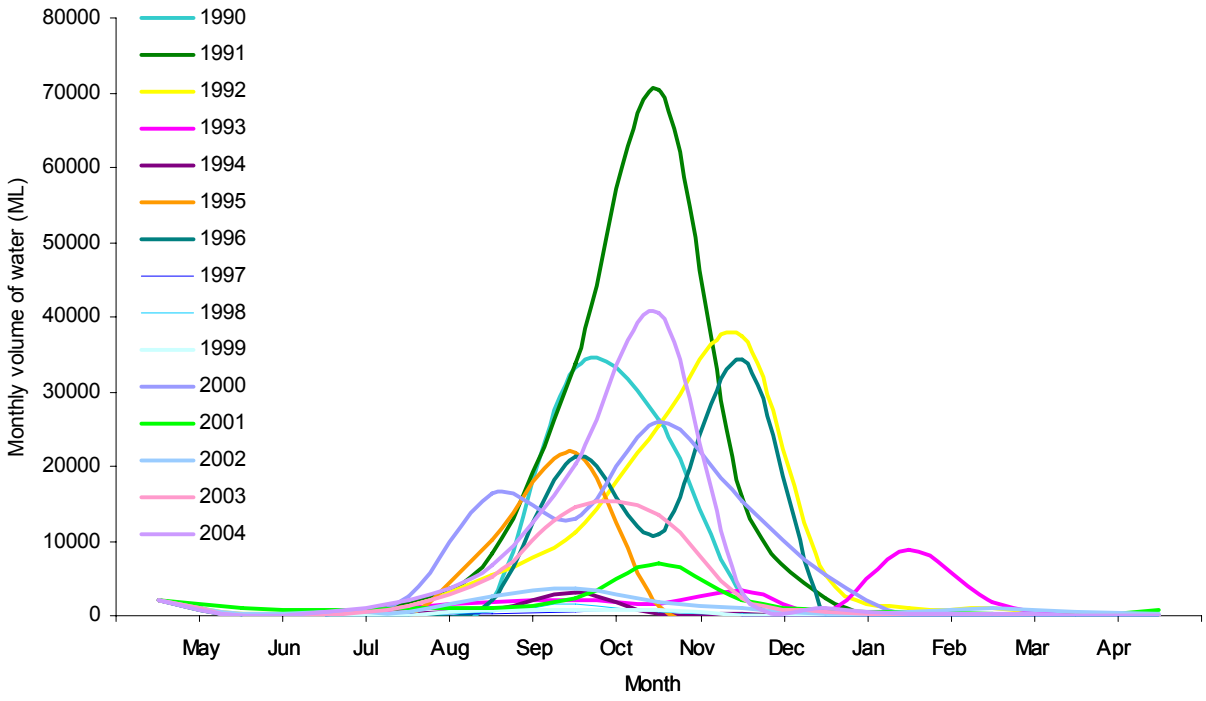

Figure 9. Monthly volume of water in Drain M between 1990 and 2004, showing that water flow generally peaks between September and December each year. Data provided by the South Eastern Water Conservation and Drainage Board.

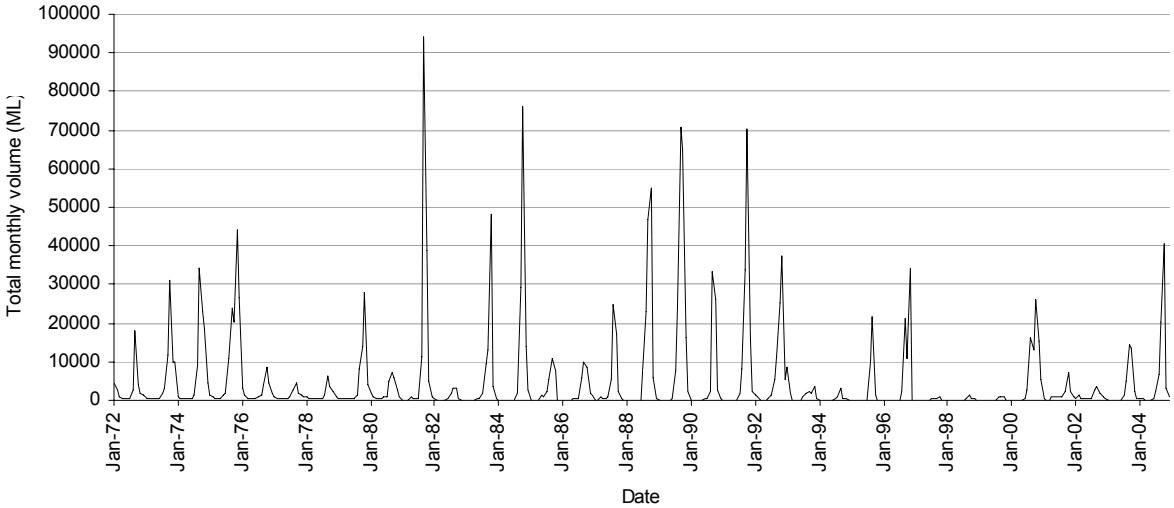

Figure 10. Total monthly volume in Drain M at Woakwine Range between January 1972 and December 2004. Data provided by the South Eastern Water Conservation and Drainage Board.

The construction of drains in the South East has significantly reduced the amount of salt in the upper proportion of the soil profiles, which has created a better environment for plant growth on interdunal flats (Schrale *et al*., 1987), and is thought to benefit future generations of farmers (Holmes and Waterhouse, 1983). Holmes and Waterhouse (1983) reported that Drain M alone discharged almost 25,000 tonnes of salt to the sea in 1980. However, Schrale *et al*. (1987) estimated that on average, 103,400 tonnes of salt was discharged each year between 1971 to 1976. Salinity within Drain M ranges between 500-1,000 mg L^{-1} TDS (total dissolved salts) at high flows and 1,000-1,600 mg L^{-1} TDS at lower flows (Jensen *et al.*, 1980). Random water samples collected by the South East Water Drainage Board between 1974 and 1985 revealed that TDS has varied between 294 mg L^{-1} and 1,238 mg L^{-1}, and averaged 895 mg L^{-1} (n = 29) over this period. Holmes and Waterhouse (1983) provide some data on the main constituents of the drain water in July 1980. At this time, salinity was recorded as 855 mg L^{-1}. Other water quality parameters and their values were; sodium (Na), 192 mg L^{-1}; potassium (K), 5 mg L^{-1}; calcium (Ca), 65 mg L^{-1}; magnesium (Mg), 38 mg L^{-1}; chlorine (Cl), 327 mg L^{-1}; sulphate (SO_4^{2-}), 73 mg L^{-1}; nitrate (NO_3^-), 1.4 mg L^{-1}. Dissolved oxygen in water samples within the Drain has varied from 8.1 to 13.5 mg L^{-1}, and averaged 10.7 mg L^{-1} (data from the South East Water Drainage Board from samples collected between 1980 and 1985).

1.3.2. Blackford Drain

The Blackford Drain outlet is located approximately 4 km north of the township of Kingston, and discharges into Lacepede Bay. Lacepede Bay is a large, relatively sheltered embayment, which contains the largest single seagrass meadow in the South East (25,062 ha; Edyvane, 1999). Seagrasses present include *P. australis*, *P. angustifolia*, *P. coriacea*, *P. sinuosa*, *A. antarctica* and *Z. tasmanica* (Robertson, 1984; Edyvane, 1999). Seagrasses in the bay are thought to be located in waters between 0.5 and 12 m depth, and have been recognised as ecologically important nursery and feeding areas for various fish and crustaceans (Edyvane, 1999).

Construction of the Blackford Drain began in 1963, almost a century after initial drainage efforts in the South East began. Construction of the main drain as well as other smaller adjoining drains (Murrabinna-Blackford 1–8) was completed by 1970. Blackford Drain is 55 km long, and has approximately 108 km of subsidiary drains (Blackford minor drains, Blackford-Murrabinna Drains system, Jacky Whites Drain and sub-system drains) (M. de Jong pers. comm.). Flow within the Blackford Drain varies greatly with season, and is generally at its greatest during November to April (Figure 11). Monthly flow volumes since 1972 have averaged 1,688 ML and reached a maximum of 19,331 ML in March 1982 (Figure 12). Water flow generally occurs all year around, however on seven occasions during the past 33 years the Drain has dried up.

Despite the fact that Drain M discharges almost three times more water than Blackford Drain, Schrale *et al*. (1987) estimated that the amount of salt discharged by Blackford Drain between 1971 and 1976 was approximately 238,000 tonnes per year; more than double the amount discharged from Drain M during the same period. While the majority of this salt originates from the groundwater interception, it has been estimated that 3% comes from rainfall (Schrale *et al*., 1987).

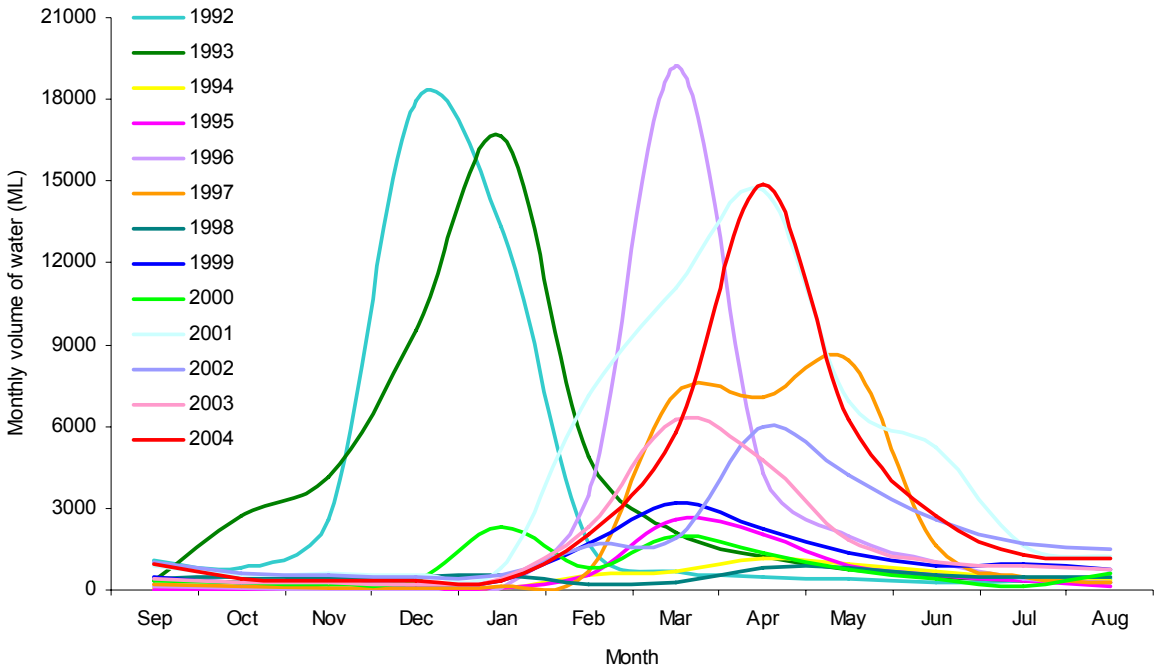

Figure 11. Monthly volume in Blackford Drain between 1992 and 2004, showing that water flow generally peaks between November and April each year. Data provided by the South Eastern Water Conservation and Drainage Board.

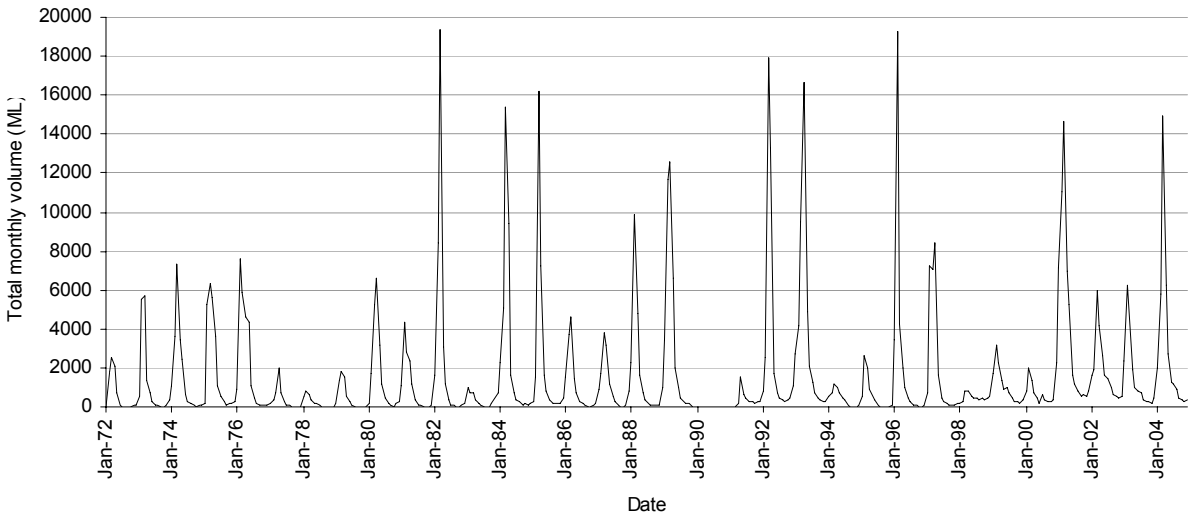

Figure 12. Monthly volume in Blackford Drain between January 1972 and December 2004. Data provided by the South Eastern Water Conservation and Drainage Board.

1.3.3. Maria Creek

Maria Creek is located adjacent to the township of Kingston and drains into Lacepede Bay, approximately 3 km south of the Blackford Drain outlet. Construction of a small channel from Maria Swamp to Maria Creek in 1863 represented one of the first cuts made in the South East (Turner and Carter, 1989), second only to cuttings made in the vicinity of the Port MacDonnell township in 1862 (SEDB, 1987). The initial cuttings were completed by late 1863 and in 1884 construction of the Kingston Main Drain began (SEDB, 1980). The outlet to Maria Creek was completed by 1887 (SEDB, 1980) and no further changes have been made since that time. The total length of Maria Creek and its subsidiaries is 28.6 km (M. de Jong pers. comm.). Flow rates within Maria Creek have not been measured.

1.3.4. Butchers Gap Drain

The Butchers Gap Drain is located approximately 6 km south of the Kingston township and divides Salt Lake and Butchers Lake, both of which encompass the Butcher Gap Conservation Park. Construction of the Butchers Gap Drain began in 1899 and by 1905 construction of the first drain was completed. In 1906, the Butchers Gap Drain was extended to include the Wongolina and Barooka Drains. Widening of Butchers Gap Drain took place in 1912 and 64 years later the Butchers Gap outlet structure was replaced. Currently the Butchers Gap Drain and associated drains (Wongolina and Barooka Drains) have a combined length of approximately 15.2 km (M. de Jong pers. comm.). Flow rates within the Butchers Gap Drain have not been measured, but are thought to be considerably lower than those of the other drains investigated as part of the current study.

1.4. Impacts to Seagrass

Given that seagrasses are thought to contribute up to 15% of the world's annual net carbon production (Duarte and Chiscano, 1999), and play a key role in coastal processes such as nutrient cycling, sediment stabilisation and the provision of habitat for numerous marine organisms (Fonseca and Fisher, 1986; Bell and Pollard, 1989; Hillman *et al.,* 1989; Short and Wyllie-Echeverria, 1996; Connolly *et al.,* 1999; Duarte, 2002), seagrass loss has generally been well documented (e.g. Shepherd *et al*., 1989; Seddon and Murray-Jones, 2002; Seddon *et al*., 2003). While some seagrass losses have been attributed to natural causes such as hurricanes, earthquakes and diseases (e.g. Clarke and Kirkman, 1989), the majority of losses have been associated with changes in water quality and clarity resulting from increased coastal urbanisation and development (Short and Wyllie-Echeverria, 1996). Some of the mechanisms thought to affect the quality of the water and the health of seagrasses include elevated nutrient concentrations and associated excessive epiphyte growth (e.g. Cambridge *et al*., 1986; Shepherd *et al*., 1989; Ralph *et al*., 2006), increased turbidity (e.g. Preen *et al.,* 1995; Campbell and McKenzie 2004) and sedimentation (e.g. Gacia and Duarte, 2001), and herbicides and pesticides (e.g. Schwarzchild *et al*., 1994; Scarlett *et al*., 1999; Haynes *et al.*, 2000b). For the current study, in order to identify if the drains in the South East are impacting upon nearby seagrass beds, various water quality parameters were assessed in the drain

water and the receiving environment, and then compared to values at control sites. A range of seagrass health parameters were also measured, and seagrass distributions were mapped to determine if any losses may have occurred previously.

2. MATERIALS AND METHODS

2.1. South East Region

2.1.1. Study sites

Field work was undertaken at various sites along the south-east coast of South Australia. Seagrasses in the South East are limited to a few areas including Lacepede, Rivoli, Guichen, Stinky and Nora Creina Bays, around Carpenter Rocks, and in waters adjacent to Port MacDonnell (Figure 1). Drains that discharge directly into seagrass beds occur in only two of these areas; Lacepede and Rivoli Bays. Consequently, in order to assess the impacts of drain discharges on seagrasses, these two bays were studied. Four 'Impact' and four 'Control' sites were used in the study. Impact sites were located adjacent to the discharges of Drain M, Blackford Drain, Maria Creek and Butchers Gap Drain. Control sites were located at least 3 km away from each of the Impact sites (Figure 13). Seagrass and water samples were collected at each of these sites in spring 2004 and autumn 2005. Water samples were also collected from within the four drains at the junction of major roads (see Appendix 1). At the time of sampling, flow within each of the drains varied substantially. During spring 2004, flows were greatest in Drain M, followed by Blackford Drain and Maria Creek. While Butchers Gap Drain contained water in spring 2004, flow was non-existent, and it had no water in it at the time of sampling in autumn. In autumn 2004, Blackford Drain was flowing, albeit slowly (flow volume averaging 5.1 ML/day), Drain M was reduced to small pools of water, while it was difficult to determine flow within Maria Creek because of tidal influences.

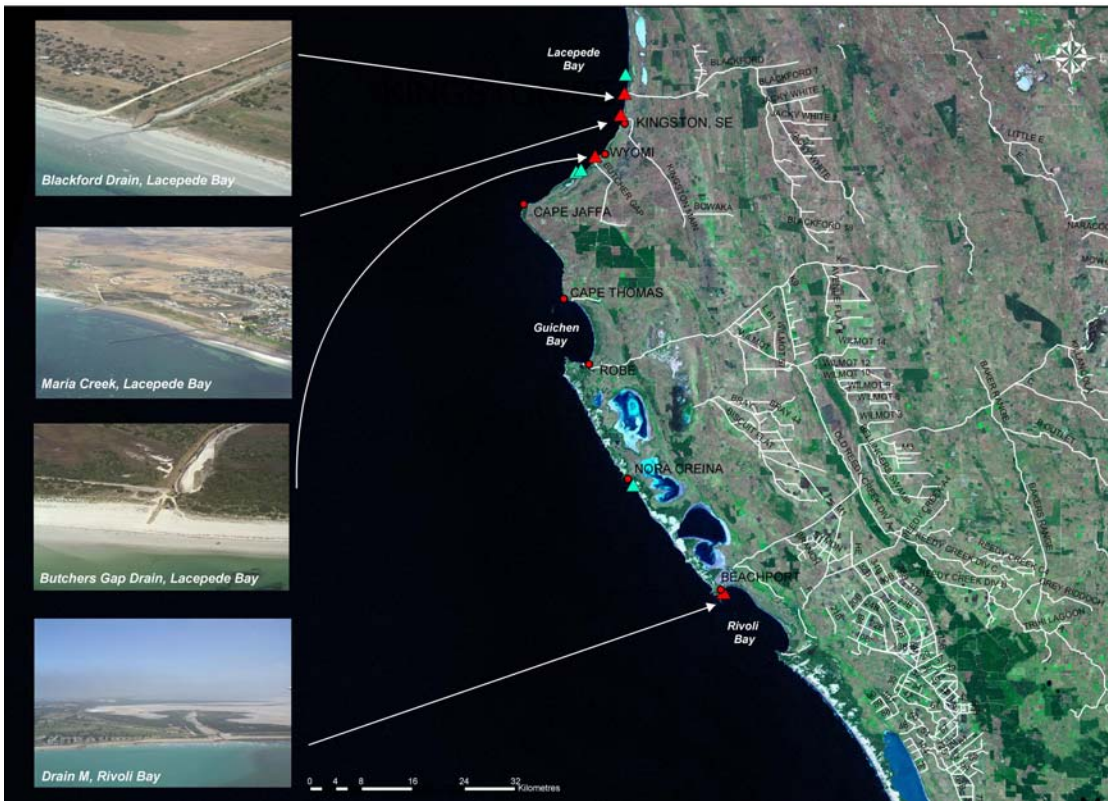

Figure 13. Map of the South East Region of South Australia and the location of four Impact and Control sites used in the study. Red triangles represent the location of Impact sites, while green triangles represent the location of Control sites.

2.1.2. Hydrodynamics

The South East coast runs in a northwest to southeast direction, with very little protection from the energy of the open ocean. Consequently the South East coast is very dynamic, and is subjected to persistently high southwest ocean swells and onshore westerly winds. During storm events, usually during winter, extreme wave energies can cause extensive beach erosion (Edyvane, 1999; Short and Hesp, 1984 in Seddon *et al.*, 2003). Wave energies vary substantially along the coast; in some areas wave energies are extremely high (e.g. at the northern end of Younghusband Peninsula), while in others wave energies are low (e.g. Bucks and MacDonnell Bays). Wave energies are considered to be moderate in Rivoli Bay and low in Lacepede Bay. Low wave energies in Lacepede Bay can be in part attributed to the protection offered by Cape Jaffa, and the orientation of the beach, but mainly due to the presence of shallow nearshore reefs (Sprigg, 1979). Greater wave energies are experienced in Rivoli Bay, which faces in a south-westerly direction. Few data on wave heights exist for the South East; however, visual observations of wave height from Cape Northumberland indicate that 62% of the year there was a 2-4 m swell dominating from the SW, for 6% of the year swell was over 4 m, and for the remaining 32% of the year, the swell was less than 2 m (Short and Hesp, 1984 in Seddon *et al.*, 2003). Tidal and wind movements generate a northerly flow of water within Lacepede Bay in summer and a southerly flow of water in winter (SKM, 2001). Current speeds are likely to reach 12 – 15 cm s^{-1} in summer and 10 – 12 cm s^{-1} in winter (SKM, 2001). Unfortunately there is no information available on circulation within Rivoli Bay.

2.2. Seagrass Health

2.2.1. Leaf density, leaf area, leaf length and width and epiphyte load

Seagrass parameters that are commonly used to assess the health of seagrass meadows, and that have been found to vary with changes in environmental conditions/anthropogenic inputs, include seagrass leaf length and width, leaf area, leaf density, epiphyte load, and photosynthetic activity (Neverauskas, 1987; Fitzpatrick and Kirkman, 1995; Frankovich and Fourqurean, 1997; Udy and Dennison, 1997; Delgado *et al*., 1999; Wood and Lavery, 2000; Ruiz and Romero, 2001; Ruiz *et al*., 2001; Bryars *et al*., 2003; Macinnis-Ng and Ralph, 2003a; Ruiz and Romero, 2003). Consequently, these parameters were assessed in the present study. Seagrass samples were collected at the four Impact and four Control sites (Figure 12). *Posidonia sinuosa* and/or *P. australis* was found at all sites. At each site, ten haphazardly placed, 0.0625 m^2 (250 x 250 mm) quadrats of seagrass were harvested at the sediment level during one-week periods in spring 2004 (18 – 22 October) and autumn 2005 (25 April – 4 May). Seagrass samples were placed in labelled plastic bags and frozen.

After defrosting, the length and width of every leaf was determined (length to the nearest 1 mm and width to the nearest 0.5 mm) to obtain the average and maximum leaf length and average leaf width, seagrass leaf density, and leaf area index at each site. Leaf area index (m^2 m^{-2}) within each quadrat was calculated using the following formula:

$$leaf\ area\ index\left(m^{2}\ m^{-2}\right)=\left[\sum_{n=1}^{quadrat} each\ leaf\ length \times width \times 2\right]\times 16$$

To obtain epiphyte dry weight, the epiphytes from approximately 20 leaves[2] from randomly selected shoots were scraped off using a blunt blade. The seagrass leaves were then rinsed with water to ensure that all epiphytes were collected and the solution was combined with the epiphytes that were scraped off the leaves. The resultant epiphyte and water solution was then vacuum filtered using Whatman glass fibre filters (0.7 µm pore size) and then placed in an oven for approximately 12 hours at 65ºC. The filters were weighed and the initial filter paper weight removed to give the dry weight of epiphytes. This process proved to be extremely time consuming, especially considering that the seagrasses generally contained high proportions of calcareous algae which were difficult to remove, and as a consequence the methodology used to remove the epiphytes was changed during processing of the spring 2004 samples. The second method involved placing the leaves from randomly selected shoots in an oven for approximately 48 hours at 65ºC. The combined dry weight of leaves and epiphytes was recorded. Epiphytic algae were then removed by washing in a 5%

[2] Seagrass shoots were selected to obtain seagrass leaves of varied ages and consequently varied epiphyte loads. The number of leaves per shoot varied between 1 and 3 and as a result exactly 20 leaves could not be obtained in every sample.

hydrochloric acid solution (HCl) and, if required, scraping the leaves with a blunt blade (Neverauskas, 1987). After rinsing in demineralised water, the resultant seagrass leaves were then re-dried and re-weighed as above and epiphyte weight was calculated by removing the seagrass dry weight from the combined weight of the seagrass and epiphytes. Weights of the leaves and epiphytes were recorded to the nearest 0.0001 g. Epiphyte dry weight (in g m^{-2} leaf area) was calculated using the following formula:

$$epiphyte\ load\ \left(g\ dry\ weight\ m^{-2}\ leaf\ area\right) = \frac{epiphyte\ dry\ weight\ (g)}{leaf\ \ area\ \left(m^2\ of\ \ seagrass\ leaves\right)}$$

In order to make reliable comparisons between the two epiphyte removal techniques, at three sites, epiphytes were removed from seagrasses from five quadrats using the scraping technique and the other five using the HCl technique. Differences in epiphyte weight between the two techniques (average epiphyte weight scraping – average epiphyte weight HCl) were calculated and compared. On average, the HCl technique removed 6.97 g DW m^{-2} more epiphytes than the scraping technique (Figure 14). In order to directly compare epiphyte loads at each site over each time period, the average difference in epiphyte load between techniques was added to those samples in which only the scraping technique was utilised. These samples are those from the Butchers Gap and Maria Creek Control sites and the Butchers Gap Impact site during spring 2004.

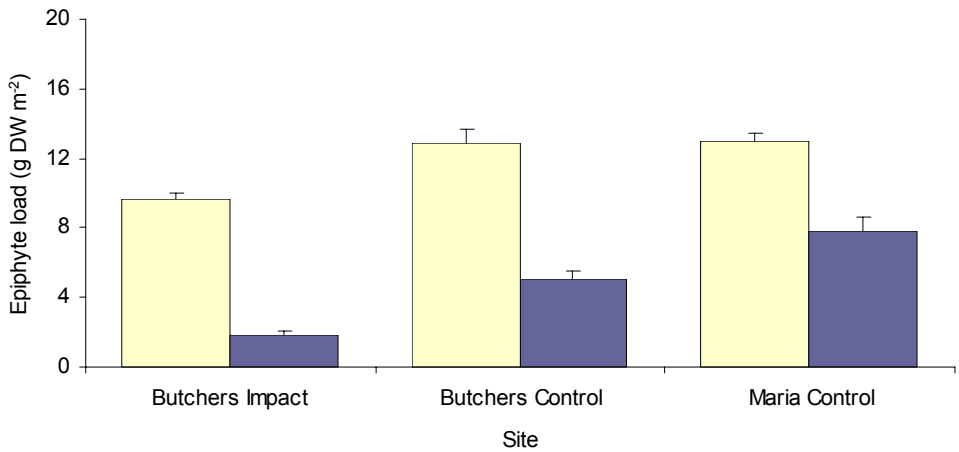

Figure 14. A comparison of methods used to quantify epiphyte loads. Average epiphyte loads (± standard error) on seagrasses at three sites using the scraping technique (blue bars) and the HCl technique (yellow bars) during spring 2004 (n = 5).

All parameters were analysed using univariate ANOVA in SPSS (SPSS Inc., Illinois). Treatment (Impact versus Control) and season were fixed factors, while site was nested within treatment. Levene's test indicated that the assumption of homogeneity of variances was rarely met. However, Underwood (1997) states that the ANOVA is robust to deviations from this assumption, and therefore, parametric ANOVAs were subsequently performed. For all tests, a significance level of $\alpha = 0.05$ was used.

2.2.2. Photosynthetic efficiency

As an indicator of stress, chlorophyll *a* fluorescence of seagrass leaves was measured using a pulse amplitude modulated (PAM) fluorometer (Diving-PAM, Walz, Germany 1998). Chlorophyll *a* fluorescence has been used to determine the stress of terrestrial plants and more recently has been applied to marine plants using the Diving-PAM (e.g. Seddon and Cheshire, 2001; Macinnis-Ng and Ralph, 2003a; Macinnis-Ng and Ralph, 2003b; Macinnis-Ng and Ralph, 2004). Chlorophyll *a* fluorescence measurements using the Diving-PAM were taken *in situ* on five haphazardly selected *Posidonia* leaves within each of the ten quadrats at each site in spring 2004 and autumn 2005. Diving PAM settings used throughout the study were: measuring intensity = 6, saturating intensity = 8, saturating width = 0.8, and gain = 2. All other settings were default levels.

Measurements of effective quantum yield (an indicator of photosynthetic efficiency) were used in analysis. Prior to analysis, the data was natural log transformed using the formula: Ln yield = ln (1 - yield), to homogenise the variance associated with the means (Cheshire, 2004 in O'Loughlin, 2004). A univariate ANOVA was performed in SPSS (SPSS Inc., Illinois). Treatment (Control verses Impact) and season were fixed factors, while site was nested within treatment. Levene's test indicated that the assumption of homogeneity of variances was rarely met. However, Underwood (1997) states that the ANOVA is robust to deviations from this assumption, and therefore, parametric ANOVAs were subsequently performed. For all tests, a significance level of $\alpha = 0.05$ was used.

2.3. Water Quality

2.3.1. Salinity, temperature and turbidity

To determine general water quality within the four drains and at Impact and Control sites, a HORIBA Water Quality Metre was used to measure turbidity, salinity and temperature. At Control and Impact sites, measurements were taken just above the seagrass beds, while in the drains, measurements were taken mid water where possible. Due to instrument error during the spring 2004 sampling period, turbidity measurements were not obtained for all sites.

2.3.2. Nutrients

Water samples for nutrient analysis were collected from the middle of the water column within the four drains using a 1 L glass beaker or a Niskin bottle (depending on water depth), during spring 2004 and autumn 2005. During the same periods, a Niskin bottle was used to collect water samples approximately 30 cm above the canopy height in areas of seagrass cover at Impact and Control sites (Figure 13). Following the collection of each water sample, the sampling container and lids were rinsed with the filtered (0.45 μm) sample water, before 40 ml of filtered water was collected. Immediately after collection, samples were placed on ice in the dark, before being transferred to a freezer on shore. Within two weeks of collection,

samples were sent to the Water Studies Centre at Monash University, where they were analysed for total nitrogen (TN), total phosphorus (TP), ammonia (NH_3), and nitrite plus nitrate (NO_X) using flow injection analysis (FIA) on a QuickChem 8000 Automated Ion Analyser. In cases where nutrient concentrations were below detectable limits (i.e. <0.01 mg L^{-1} for TP, and < 0.001 mg L^{-1} for NH_3 and NO_X), and no value was obtained, the detectable limit value was halved for data analysis and graphs. Two additional water samples were collected for nutrient analysis at the time of the first major flush from each drain during 2005 (Table 2). These additional samples were also sent to the Water Studies Centre for analysis. A description of the drain and catchment characteristics used to determine timing of the 'first flush' is provided in Appendix 1, together with observations made by Mark de Jong at regular site visits to each of the drains.

All nutrient parameters (TN, TP, NH_3 and NO_X) were analysed using univariate ANOVA in SPSS (SPSS Inc., Illinois). Treatment (Impact verses Control) and season were fixed factors, while site was nested within treatment. Levene's test indicated that the assumption of homogeneity of variances was rarely met. However, Underwood (1997) states that the ANOVA is robust to deviations from this assumption, and therefore, parametric ANOVAs were subsequently performed. For all tests, a significance level of $\alpha = 0.05$ was used.

Table 2. The date and location of additional water samples collected by Mark de Jong from the South Eastern Water Conservation and Drainage Board for analysis of various nutrients.

Date of Collection	Number of samples	Location
18/07/05, 1445 hours	2	Drain M, Robe to Beachport Road Bridge.
11/08/05, 1130 hours	2	Blackford Drain, Princes Highway
11/08/05, 1100 hours	2	Maria Creek, East Terrace (Robe-Kingston Road) Bridge
11/08/05, 1015 hours	2	Butchers Gap Drain, Cape Jaffa Rd Bridge

2.3.3. Pesticides and herbicides

Water samples for analysis of a range of herbicides and pesticides were also collected during spring 2004 and autumn 2005. Water samples from within the four drains were collected midwater where possible. Samples taken at Impact and Control sites were collected approximately 30 cm from the substrate using a Niskin bottle. All samples were stored in 1 L brown glass jars provided by the Australian Government Analytical Laboratory (AGAL) and were placed immediately on ice following collection and transferred to a fridge as soon as possible. Within seven days of collection, samples were sent to AGAL for analysis of a range of organochlorine and organophosphate pesticides and triazine herbicides (Table 3). Additional water samples for pesticide and herbicide analysis were also collected at the time of the first major flush from each drain (Table 4). Two replicate samples were collected, although some of the sample jars cracked, reducing the number of samples to one at Drain M and Blackford Drain and none at Butchers Gap Drain.

Table 3. The various organochlorine and organophosphate pesticides and triazine herbicides tested for in water samples taken within four drains and at Control and Impact sites.

Organochlorine Pesticides	Organophosphate Pesticides	Triazine Herbicides
Hexachlorobenzene	Demeton-S-Methyl	Artazine
Heptachlor	Dichlorvos	Hexazinone
Heptachlor epoxide	Dlazinon	Metribuzine
Aldrin	Dimethoale	Prometryne
gamma Benzene Hexachloride	Chlorpyrifos	Simazine
alpha-Benzene Hexachloride	Chlorpyrifos Methyl	
beta-Benzene Hexachloride	Malathion	
delta Benzene Hexachloride	Fenthion	
trans-Chlordane	Azinphos Ethyl	
cis-Chlordane	Azinphos Methyl	
Oxychlordane	Chlorfenvinphos (E)	
Dieldrin	Chlorfenvinphos (Z)	
p,p-Dichlorodiphenyldichloroethylene	Ethion	
p,p-Dichlorodiphenyldichloroethane	Fenitrothion	
p,p-Dichlorodiphenyltrichloroethane	Parathion (Ethyl)	
Endrin	Parathion Methyl	
Endrin Aldehyde	Pirimiphos Ethyl	
Endrin Ketone	Pirimiphos Methyl	
alpha-Endosulfan		
beta-Endosulfan		
Endosulfan Sulfate		
Methoxychlor		

NOTE: Detectable concentrations: Organochlorine pesticides ≥ 0.01 µg L^{-1} ; organophosphate pesticides ≥ 0.10 µg L^{-1} ; triazine herbicides ≥ 0.10 µg L^{-1}.

Table 4. The date and location of additional water samples collected by Mark de Jong from the South Eastern Water Conservation and Drainage Board for analysis of organochlorine and organophosphate pesticides and triazine herbicides.

Date of collection	Number of samples	Location
18/07/05, 1445 hours	1	Drain M, Beachport, Robe to Beachport Road Bridge.
11/08/05, 1130 hours	1	Blackford Drain, Princes Highway
11/08/05, 1100 hours	2	Maria Creek, East Terrace (Robe-Kingston Road) Bridge

2.3.4. Chlorophyll *a*

Chlorophyll *a* is a green photosynthetic pigment found in plants. Chlorophyll *a* concentrations in water samples are an indicator of phytoplankton abundance and biomass in coastal and estuarine waters. Given that elevated nutrients can stimulate the growth of phytoplankton, chlorophyll *a* concentrations can be an effective measure of trophic status and are commonly used as a measure of water quality. In naturally nutrient poor systems high levels generally indicate poor water quality while low levels suggest good conditions.

Two replicate water samples were collected from within each drain, and at Control and Impact sites. Samples were collected with a Niskin bottle at all sites and were transferred to 1.25 L plastic jars and placed on ice in the dark. On the same day as collection, the samples were vacuum filtered using muffled (pre-combusted at 450ºC overnight) Whatman glass fibre filters (0.7 µm pore size). The quantity of water filtered varied substantially between sites depending on the amount of phytoplankton present. In most cases, 300 ml was filtered from samples collected within the drains, while 1000 ml was filtered from water samples collected at both Impact and Control sites. The filter papers were folded and then wrapped in aluminium foil before they were fixed and stored in liquid nitrogen. Upon return to Adelaide the samples were transferred to a -80ºC freezer until extraction.

Extraction Process: The filter papers were transferred to labelled test tubes containing 5 ml of methanol (CH_3OH) and refrigerated for 24 hours. The filter paper was removed from the test tube and the resultant extract was then centrifuged for 10 minutes (3,000 rpm). Spectrophotometer readings were then taken on a Hitachi U-2000 double beam spectrophotometer (Hitachi Ltd., Tokyo, Japan) at Adelaide University. Chlorophyll *a* concentrations were calculated as µg L^{-1}.

2.4. Video Transects

Information about the habitats adjacent to the discharges of Drain M, Blackford Drain, Maria Creek and Butchers Gap Drain was obtained from underwater video transects of the area. A Morphvision underwater video camera mounted on a paravane frame was towed behind a boat at a consistent rate of approximately 2 knots or 1.2 m s^{-1}. The video was suspended approximately 1 m above the seafloor. Nine video transects were taken perpendicular to the shore at each site with one adjacent to the drain discharge point and the remaining eight with increasing distance from the drain discharge point (50m, 150m, 400m and 900m) in both directions (Figures 14-16). Video transects were approximately 50 m long. Additional transects were also taken adjacent to the Blackford Drain and Maria Creek sites, two of which ran parallel to the shoreline and two of which radiated from the discharge point (Figure 15 and Figure 16). The number and position of transects at each site varied slightly (Figure 15 - Figure 17). Consequently, further information on the location of transects at each site, together with a more detailed outline of the methodologies, is provided in Appendix 2.

Figure 15. Map showing the location of video transects surrounding the Blackford Drain Outlet, in the South East of South Australia.

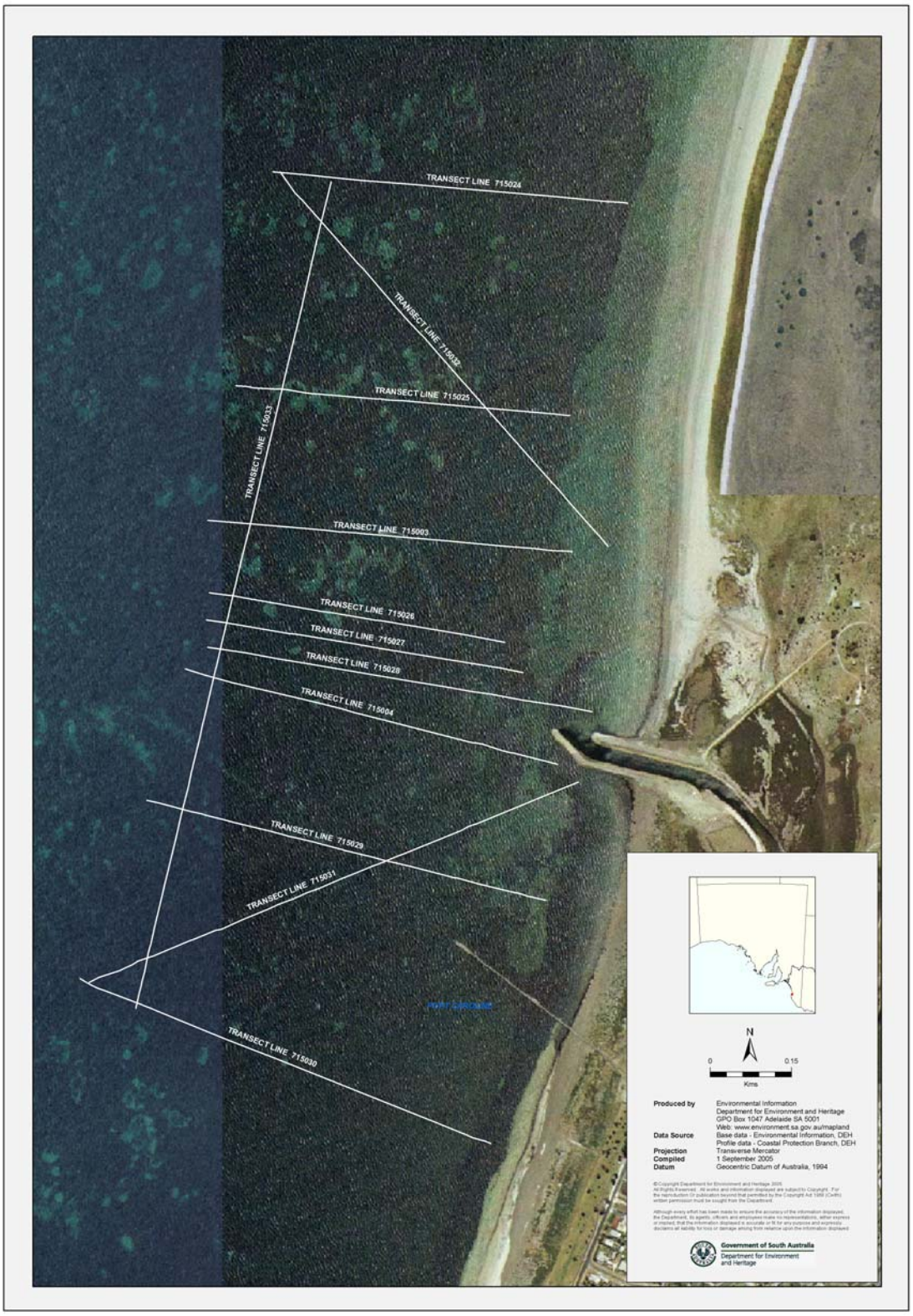

Figure 16. Map showing the location of video transects surrounding the Maria Creek Drain Outlet, in the South East of South Australia.

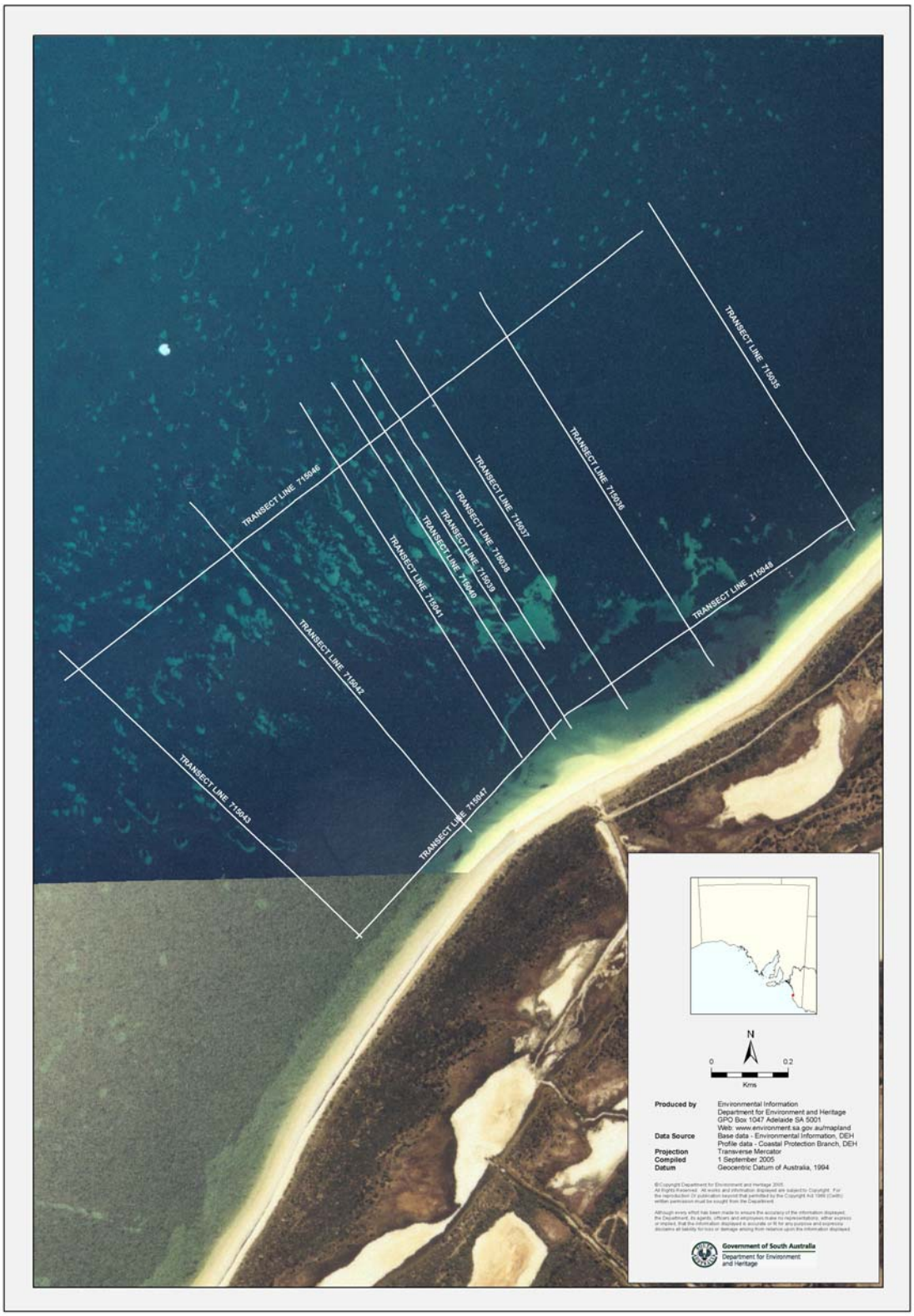

Figure 17. Map showing the location of video transects surrounding the Butchers Gap Drain Outlet, in the South East of South Australia.

Videotapes were analysed for habitat type, habitat density and epiphyte cover using a method based on the Line Intercept Transect (LIT) method (e.g. Seddon *et al.*, 2003; English *et al.*, 1994; Miller *et al.*, 1998). Along each transect, sections of habitat that appeared homogeneous were scored semi-quantitatively (Figure 5) based on the presence of major habitat types (Table 6). A semi-quantitative approach was also used to determine epiphyte cover along each transect (Table 7). When changes in habitat type, density and epiphyte cover were observed, the time on the videotape was recorded. These data were used with the GPS data collected in the field to determine the position of habitat change. Data on habitat type, habitat density and epiphyte cover were then overlaid onto ortho-rectified and geo-referenced aerial photographs for each site. Extremely poor visibility at Beachport made analysing the videos difficult and as such, the results are not provided. Nonetheless, habitat mapping of northern Rivoli Bay was undertaken by Seddon *et al.* (2003) and can be found in their report.

Table 5. Semi-quantitative scoring system used for recording habitat densities from video transects collected adjacent four drains in the South East of South Australia.

Percentage cover of habitat	Presence
0	Not present
1 - 10	Very Sparse *
11 - 30	Sparse *
31 - 70	Mid dense
71 - 100	Dense

NOTE: Habitat density classifications adapted from Muir (1977). If the seagrass bed was made up of several species of seagrass but was of uniform coverage, the habitat was scored as uniform seagrass habitat as there is no separation by bare sand i.e. the seagrass habitat was uniform, although each species was not contiguous. * = Sparse and very sparse categories were combined for mapping.

Table 6. Scoring system for recording habitat type from video transects collected adjacent to the Blackford Drain, Maria Creek and Butchers Gap Drain in the South East of South Australia.

Habitat type	Description
Seagrass	Where seagrass habitat appears continuous, i.e. all *Posidonia* sp. or all *Amphibolis* sp. Type of seagrass noted.
Mixed seagrass	Where there was an obvious mixing of seagrass genera, i.e. *Posidonia* sp. and *Amphibolis* sp. growing amongst each other with no obvious boundaries between the two.
Macroalgae	Continuous macroalgal community.
Mixed seagrass /Macroalgae	Where there was an obvious strong mix of seagrass beds with macroalgal communities.
Other	Habitats other than seagrass and macroalgae, e.g. sponge communities. Type of community noted.

NOTE: In most cases only one of these habitat types existed at any one time. On the maps that were produced this dominant habitat type has been referred to as 'species 1'. In some cases a second habitat type was present at the same location (referred to as 'species 2'), but this data was not included on the maps.

Table 7. Semi-quantitative scoring system used for recording epiphyte cover from video transects collected adjacent to four drains in the South East of South Australia.

Percentage cover of epiphyte	Presence
0	Not Visible
1 - 10	Very Sparse *
11 - 30	Sparse *
31 - 70	Mid dense
71 - 100	Dense

* = Sparse and very sparse categories were combined for mapping.

2.5. Profile Lines

Four profile lines near Kingston (715003 and 715004), Butchers Gap (715001) and Cape Jaffa (715002) (Figure 18), were established by the Coast Protection Board in 1977 to monitor changes in substrate depth. At the time, permanent marks were set up along the foreshore and these have been used to relocate the profile lines for subsequent monitoring. Profile levels are taken at these fixed marks along the shore to wading depth, after which depths are measured acoustically from a boat. Heights are measured to Australian Height Datum (AHD). Accuracy of the profile data varies with how recently the data were collected and with the collection method. Wading data are considered accurate to within 3 cm. In the past, the influence of tides and waves has reduced the accuracy of boating data to approximately 20 cm, although developments in technology have reduced this error to between 4 and 7 cm. Profile lines were surveyed each year between 1977 and 1980, 1986 and 1988, and subsequently in 1997 and 2000.

Profile lines indicate substrate depth and also tend to show obvious features such as where seagrass beds start and end. These are evident from the drops or rises in sea floor profiles. For example, the 'blue line' that marks the sharp transition between sand of the inshore environment and the point at which offshore seagrass beds begin, often shows as a sudden jump or step, due to the tendency of seagrass beds to trap sediments resulting in increases in profile height at this point. Overlaying subsequent profile years can show whether or not this seagrass line is remaining constant in its position or whether the seagrass bed is showing signs of accretion or retreat.

At the same time as video transects were taken by DEH (see section 2.4.), additional profile lines were set up. These profile lines were located along the same line as the video transects adjacent to the Blackford Drain (Figure 15), Maria Creek (Figure 16) and Butchers Gap Drain (Figure 17). Substrate depth and seagrass height were recorded with a duel-frequency (Bruttour) echo sounder and will ensure that future changes in the seagrass line can be determined adjacent to the drains.

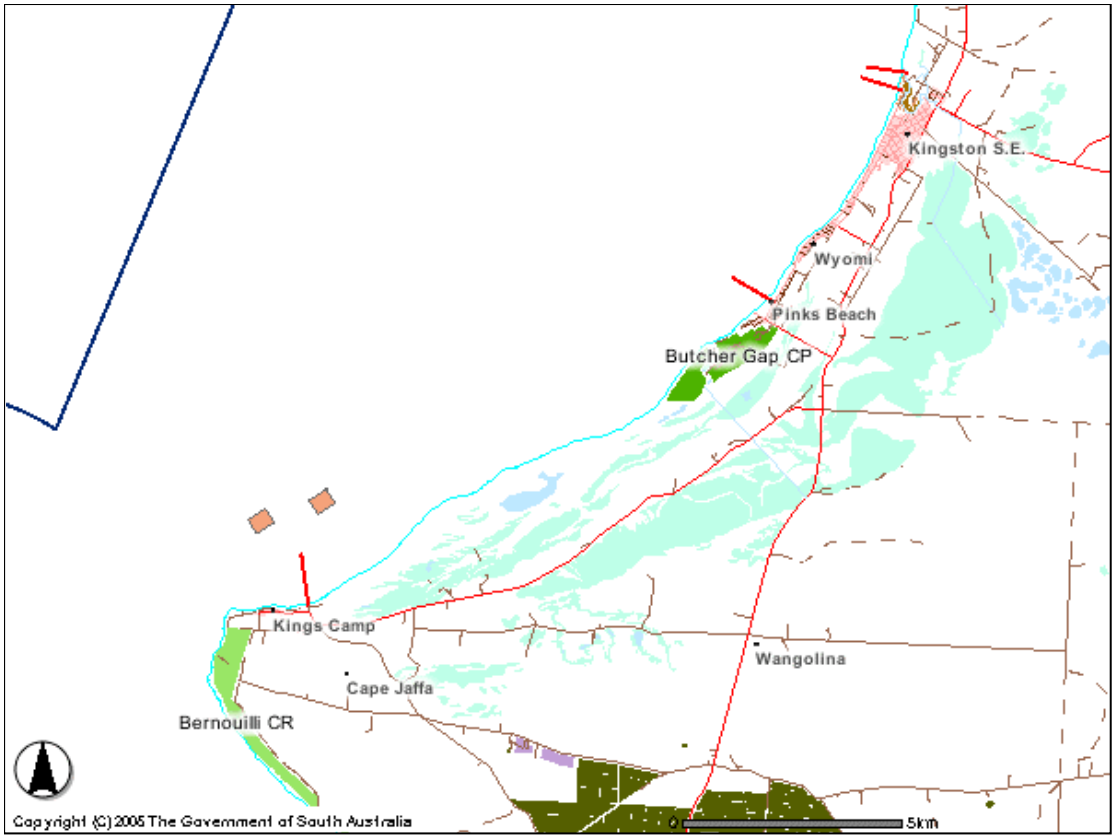

Figure 18. Map of the Kingston and Cape Jaffa region, showing the location of profile lines (in red) established by the Coastal Protection Branch, Department for Environment and Heritage.

3. RESULTS AND DISCUSSION

3.1. Seagrass Health

3.1.1. Leaf density, leaf area index, leaf length and width, and epiphyte load

The species of seagrasses sampled varied between sites and sampling periods. The majority of sites sampled entirely *P. sinuosa*. However, samples collected at the Beachport Control and Butchers Impact sites consisted of *P. australis*, while the Butchers Gap Impact site samples consisted of *P. sinuosa* and *P. australis* during spring 2004 but *P. sinuosa* only during autumn 2005 (Table 8). These differences in species have some implications for interpreting the results (see later).

Table 8. Seagrass species collected at each of the Control and Impact sites in the South East during spring 2004 and autumn 2005.

Site	Seagrass species	
	spring 2004	**autumn 2005**
Beachport Control	*P. australis*	*P. australis*
Beachport Impact	*P. australis* and *P. sinuosa*	*P. sinuosa*
Blackford Control	*P. sinuosa*	*P. sinuosa*
Blackford Impact	*P. sinuosa*	*P. sinuosa*
Butchers Control	*P. sinuosa*	*P. sinuosa*
Butchers Impact	*P. australis*	*P. australis*
Maria Control	*P. sinuosa*	*P. sinuosa*
Maria Impact	*P. sinuosa*	*P. sinuosa*

Seagrass leaf density and leaf area index

Seagrass leaf density varied substantially with season ($P = 0.001$), treatment ($P < 0.001$) and site within treatment ($P < 0.001$) (Table 9). Leaf densities were consistently higher at Control sites (average = 2,184 ± 133 leaves m^{-2})* when compared with Impact sites (average = 1,295 ± 173 leaves m^{-2}) (Figure 19a). While seagrass leaf densities vary naturally with species, site and season (Walker and McComb, 1988; Guidetti *et al*., 2002; Smith and Walker, 2002), the values found at Control sites in this study compare with those found in healthy meadows in other areas (Neverauskas, 1987; Smith and Walker, 2002; Bryars *et al*., 2003). In the present study, leaf densities at Control sites varied between 1,832 ± 98 and 2,685 ± 196 leaves m^{-2} (Figure 19a). Neverauskas (1987) found comparable values in healthy *Posidonia* meadows along the Adelaide metropolitan coast, with densities ranging from 1,682 to 2,331 leaves m^{-2}. Similarly, Bryars *et al.* (2003) identified that healthy *P. sinuosa* beds at Kangaroo Island averaged 1,312 leaves m^{-2}. Seagrass leaf densities are often reduced in degraded systems (e.g. Wood and Lavery, 2000; Bryars *et al.*, 2003). Bryars *et al*. (2003) found that *P. sinuosa* leaf densities within a eutrophic bay on Kangaroo Island varied between 608 (± 67.4) and

* NOTE: All errors given throughout the report are standard errors.

656 (± 109.3) leaves m^{-2}, while Neverauskas (1987) found that *Posidonia* beds affected by the Port Adelaide Sewage Treatment Works contained leaf densities as low as 574 leaves m^{-2}. In the current study, *Posidonia* leaf densities at sites receiving drain inputs varied between 678.4 (± 46.7) and 2,164.8 (± 198.8) leaves m^{-2}, suggesting that seagrasses at some sites subjected to drain discharges are affected, while others are not.

Seasonal variations in leaf density were apparent at some sites. Leaf densities at the Butchers Gap and Beachport Impact and Control sites and the Blackford Drain and Maria Creek Control sites were similar over time. However, there were significant increases in leaf densities observed between spring 2004 and autumn 2005 at the Maria Creek (219% increase) and Blackford Drain (45% increase) Impact sites (Figure 19a). Remarkably, during spring 2004 the Maria Creek Impact site had the lowest leaf densities obtained during the study (678.4 ± 46.7 leaves m^{-2}), yet during autumn 2005 the same site had leaf densities comparable to healthy *Posidonia* meadows in other areas (2,164.8 ± 198.8 leaves m^{-2}) (Neverauskas, 1987; Smith and Walker, 2002; Bryars *et al*., 2003). Increases in leaf densities at the Maria Creek Impact site over time are apparent in photographs taken at this site at the time of sampling (Figure 20), with the Impact site being characterised by scattered seagrass leaves surrounded by areas of root matt with no leaves in spring 2004 (Figure 20a). Similar observations were made at the Blackford Drain Impact site during spring 2004.

Leaf densities adjacent to the Butchers Gap Impact site (1,278 ± 138 and 1,438 ± 65 leaves m^{-2}) were substantially lower than those at the Butchers Gap Control site (2,661 ± 198 and 2,685 ± 196 leaves m^{-2}), at both time periods (Figure 19a). However, leaf density differences at these sites are confounded by seagrass species. *Posidonia australis* is the dominant species adjacent to the Butchers Gap Drain, while *P. sinuosa* is dominant at the corresponding Control site. Nonetheless, the absence of any significant change in leaf densities over time at these sites (unlike that seen at the Maria Creek and Blackford Drain Impact sites), and comparable leaf densities to those in healthy *P. australis* and *P. sinuosa* beds in other areas (Smith and Walker, 2002), suggests little or no effect from the Butchers Gap Drain. This conclusion is consistent with the relatively low discharges thought to occur from the Butchers Gap Drain (see Section 1.3.4.) and the non-existent flows observed during the two sampling periods of the present study.

Table 9. ANOVA results for seasonal and spatial variation in average leaf density and leaf area index of seagrasses at Control and Impact sites in the South East of South Australia.

Leaf density				
Source	df	SS	F	*P*
Season	1	3176449.6	10.691	**0.001**
Treatment	1	31641294.4	106.497	**<0.001**
Season x Treatment	1	1616040	5.439	0.21
Site (Treatment)	6	2033908.8	6.846	**<0.001**
Error	150	297110.165		

Leaf area index				
Source	df	SS	F	*P*
Season	1	0.946	0.738	0.392
Treatment	1	149.370	116.593	**<0.001**
Season x Treatment	1	0.438	0.342	0.560
Site (Treatment)	6	9.668	7.546	**<0.001**
Error	150	1.281		

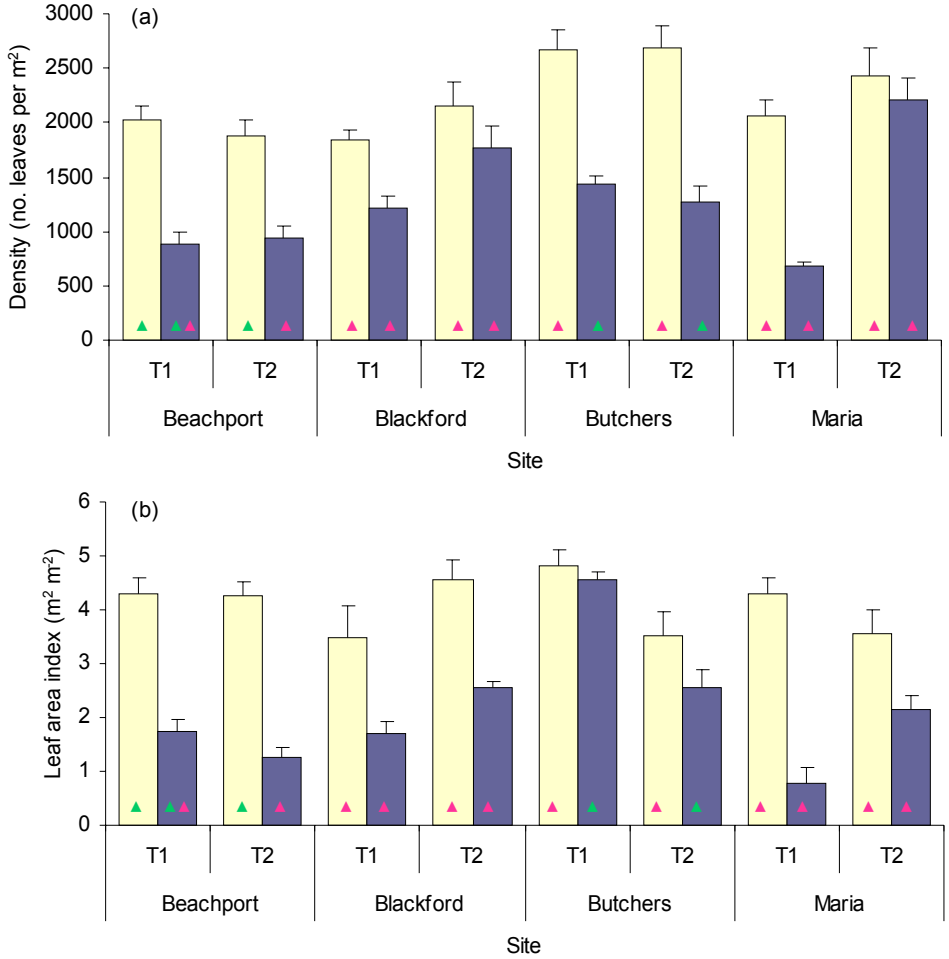

Figure 19. Average (a) leaf density and (b) average leaf area index (± standard error) of seagrasses at four Control sites (yellow bars) and four Impact sites (blue bars) in spring 2004 (T1) and autumn 2005 (T2) in the South East of South Australia (n = 10). ▲ = *Posidonia australis*, ▲ = *P. sinuosa*. ▲▲ = Mixed *P. australis* and *P. sinuosa*.

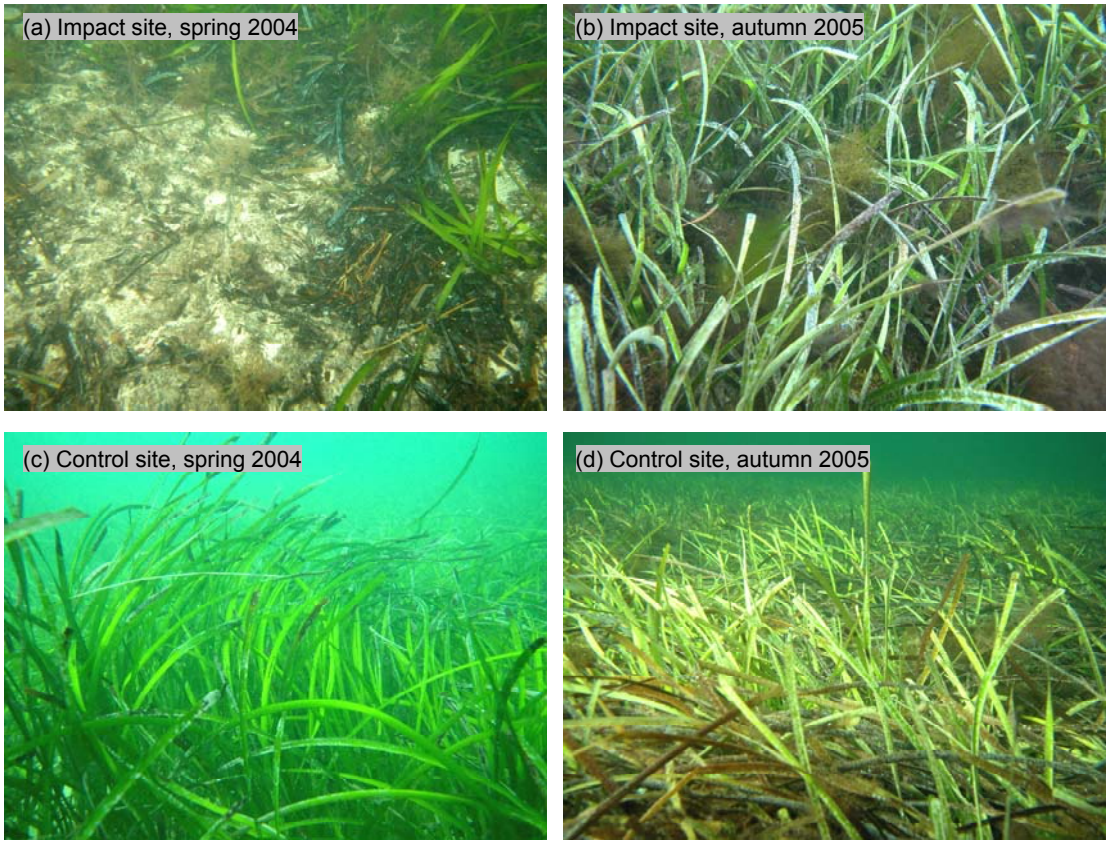

Figure 20. Photographs of *Posidonia sinuosa* adjacent to the discharge from Maria Creek (a and b) and at the Maria Creek Control site (c and d) during spring 2004 (a and c) and autumn 2005 (b and d).

Leaf area index varied significantly between treatments ($P < 0.001$) and sites within treatments ($P < 0.001$) (Table 9) and followed similar patterns to the leaf density data (Figure 19). Leaf area index values were substantially higher at control sites (4.10 ± 0.18 m^2 m^{-2}) when compared with Impact sites receiving drain discharges (2.17 ± 0.40 m^2 m^{-2}) (Figure 19b). In healthy *P. australis* and *P. sinuosa* meadows in Warnbro Sound, Western Australia, leaf index values averaged 4.1 (± 0.1) and 7.0 (± 0.2) m^2 m^{-2}, respectively (Smith and Walker, 2002). While leaf area index is likely to vary with location, the significantly reduced values observed at the Maria Creek, Blackford Drain and Beachport Impact sites (Figure 19b) suggest that drain discharges at these locations may be impacting upon the seagrasses.

Seagrass leaf density and leaf area index both decrease in response to reduced water quality and clarity and can therefore be good indicators of seagrass health (Neverauskas, 1987; Shepherd *et al.*, 1989; Wood and Lavery, 2000). Based on leaf density and leaf area index values obtained during this study, and values at healthy and degraded sites in other areas, it appears that the health of seagrasses at the Maria Creek, Beachport and Blackford Drain Impact sites was compromised in spring 2004. Seagrass leaf density and leaf area index increased from spring 2004 to autumn 2005 at the Maria Creek and Blackford Drain Impact sites, suggesting recovery during that period at the two sites. The same parameters remained significantly lower at the Beachport Impact site over time, suggesting a longer-term impact at this site. Although species differences between Control and Impact sites at Beachport mean that any conclusions should be treated cautiously.

Leaf length and width:

Seagrass leaf length has been shown to be a reliable measure of seagrass health (Wood and Lavery, 2000). Shepherd (1970) reported that partially degraded seagrasses are not only sparse but also stunted, with *Posidonia* leaves in degraded areas barely reaching 30 cm, in contrast to 60 cm in healthy conditions. Delgado *et al*. (1999) found similar values, with maximum leaf lengths of *P. oceanica* increasing from 30.9 cm to 53.4 cm with distance from a disused finfish aquaculture site. Average and maximum leaf lengths displayed similar trends in the present study, varying significantly with season, treatment, and site within treatment (Table 10). Average leaf lengths were rarely different between Control and Impact sites at Beachport and Butchers Gap, but were significantly reduced at the Maria Creek and Blackford Drain Impact sites when compared with Control sites (Figure 21a). Similar trends were obtained from maximum leaf length data (Figure 21b). Given that leaf lengths are generally lower in degraded areas (Shepherd, 1970; Delgado *et al*., 1999), reduced leaf lengths at these Impact sites, relative to Controls, are likely to be related to the drain discharges at these sites. Unlike seagrass leaf densities and leaf area index, average and maximum leaf lengths did not increase over the period of the study at the Maria Creek and Blackford Drain Impact sites, which may suggest that leaf lengths are a more sensitive measure of seagrass health.

A difference in species composition between Impact and Control sites (such as at Beachport and Butchers Gap) makes the identification of an impact difficult, as leaf lengths vary with species and the response of leaf length to changing environmental conditions may also vary. Notwithstanding this, the average and maximum *P. australis* leaf lengths found at the Butchers Gap Impact site were greater than Control sites elsewhere (Beachport). This suggests little or no impact of the Butchers Gap Drain on nearby seagrasses. Differences in *P. australis* leaf lengths between sites are more than likely related to variations in local conditions, with shorter leaves generally found in higher energy environments.

Seasonal differences in average and maximum leaf lengths were apparent (Table 10), particularly at the Butchers Gap and Maria Creek Control sites with substantial reductions in leaf lengths from spring 2004 to autumn 2005 (Figure 21a-b). Seasonal variations in leaf lengths are common in healthy systems and are related to changing growth rates (Guidetti *et al*. 2002). Guidetti *et al*. (2002) demonstrated that intermediate and adult leaves of *P. oceanica* are longest in spring and summer. Such seasonal differences correlate with the maximum growth periods of *Posidonia* species (Hillman *et al.*, 1991; Ruiz and Romero, 2003; Enríquez *et al.*, 2004). For example, Hillman *et al*. (1991) identified that the leaf production of *P. sinuosa* and *P. australis* was greatest during spring and summer and lowest during winter, with intermediate leaf production during autumn. Hillman *et al.* (1991), also demonstrated that the strength of seasonal differences varied with location. This observation may help to

explain seasonal patterns in leaf lengths at some sites in this study, yet the absence of such patterns at the Blackford Drain Control site (Figure 21a-b).

Average leaf widths varied with seagrass species (Figure 21c), but not with treatment (Table 10). Leaf widths for sites containing *P. sinuosa* averaged between 5.8 and 7.2 mm, while those sites containing *P. australis* varied between 8.8 and 11.3 mm. The one site in which a mixture of both species was collected, had an average leaf width of 9.15 mm (Figure 21c). A reduction in leaf width at the Beachport Impact site was found, but this is confounded by species differences (Figure 21a-c). Leaf widths have previously been used as an indicator of seagrass health. For example, Delgado *et al.* (1999) found that *P. oceanica* seagrass leaf widths increased from 8.3 cm to 9.8 cm with distance from a disused finfish aquaculture site. This trend, however, is not always apparent (e.g. Neverauskas, 1988). In the current study, differences between leaf widths at various sites in the South East are more likely to reflect changes in species composition rather than effects of drain discharges, and hence this indicator is not a useful one for these sites.

Table 10. ANOVA results for seasonal and spatial variation in average leaf length, maximum leaf length and average width of seagrasses at Control and Impact sites in the South East of South Australia.

Average leaf length				
Source	df	SS	F	*P*
Season	1	98581.808	47.073	**<0.001**
Treatment	1	61699.855	29.462	**<0.001**
Season x Treatment	1	317.512	0.152	0.698
Site (Treatment)	6	51971.158	24.816	**<0.001**
Error	150	2094.236		

Maximum leaf length				
Source	df	SS	F	*P*
Season	1	68724.1	8.299	**0.005**
Treatment	1	200505.6	24.211	**<0.001**
Season x Treatment	1	1.225	0	0.99
Site (Treatment)	6	262689.663	31.72	**<0.001**
Error	150	8281.441		

Average leaf width				
Source	df	SS	F	*P*
Season	1	0.284	1.162	0.283
Treatment	1	0.278	1.136	0.288
Season x Treatment	1	12.362	50.562	**<0.001**
Site (Treatment)	6	74.865	306.199	**<0.001**
Error	150	0.244		

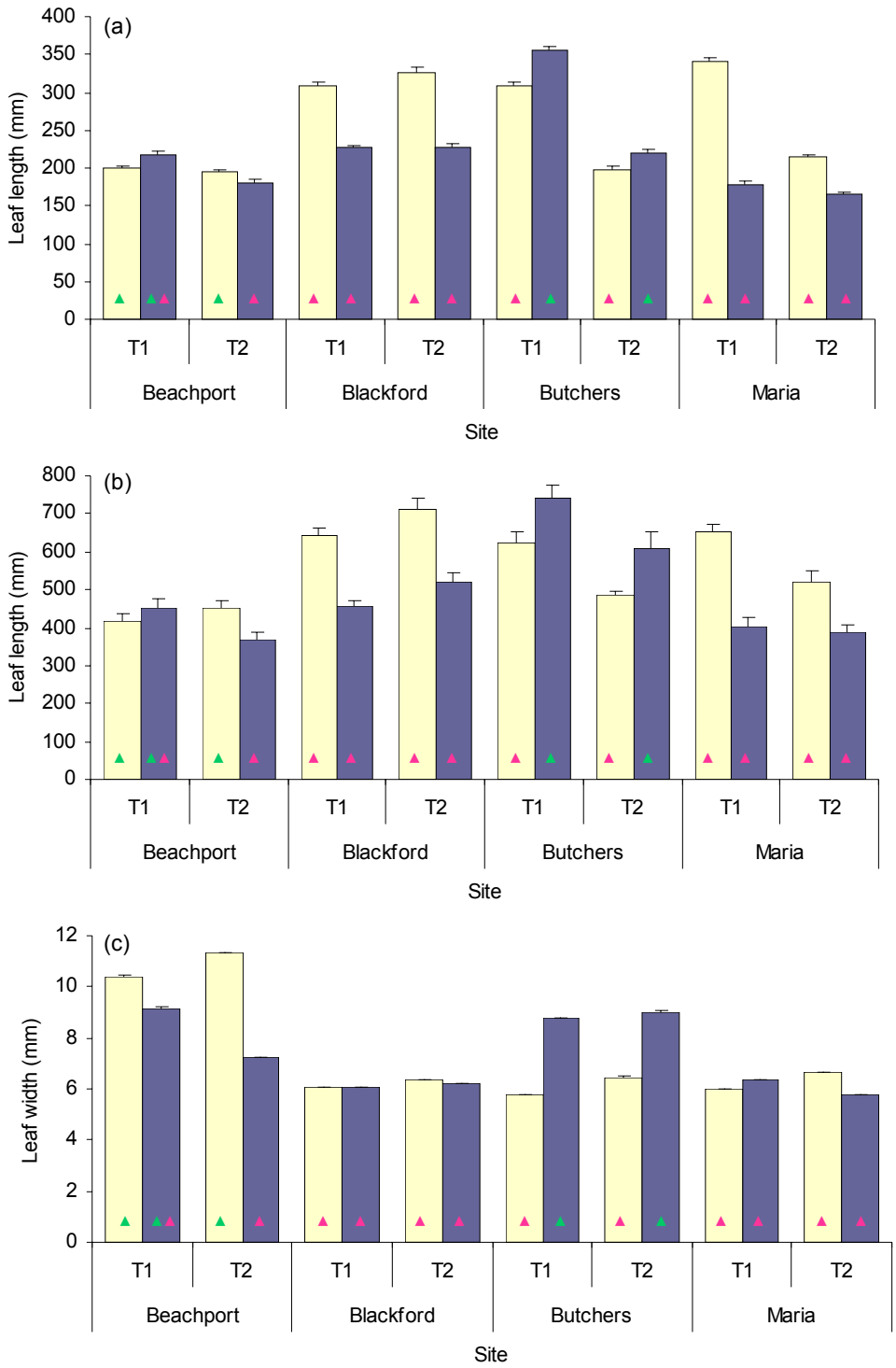

Figure 21. Average (a) leaf length, (b) maximum leaf length, and (c) average leaf width (± standard error) at four Control sites (yellow bars) and four Impact sites receiving drain water (blue bars) in spring 2004 (T1) and autumn 2005 (T2) in the South East of South Australia (n = 10). ▲ = *Posidonia australis*, ▲ = *P. sinuosa*. ▲▲ = Mixed *P. australis* and *P. sinuosa*.

Epiphyte loads

Increased epiphyte loads are common in areas receiving high levels of nutrients and have been implicated in reduced seagrass health and seagrass loss (e.g. Borum, 1985; Cambridge *et al.*, 1986; Silberstein *et al*., 1986; Tomasko and Lapointe, 1991; Bryars *et al.*, 2003). In the present study, epiphyte loads varied significantly between sites within treatments ($P < 0.001$) (Table 11). Epiphyte loads were highest at the Beachport Control (average = 20.6 ± 6.0 g DW m^{-2}) and Impact sites (average = 36.8 ± 8.1 g DW m^{-2}), while average epiphyte loads at all sites in Lacepede Bay remained below 15 g DW m^{-2} (Figure 22). Epiphyte loads found in the present study are comparatively low when compared with other studies (Neverauskas, 1987; Paling and McComb, 2000). Neverauskas (1987) identified that epiphyte loads varied between 37.7 and 81.2 g DW m^{-2} in healthy *Posidonia* meadows along the Adelaide metropolitan coast. Similarly, Paling and McComb (2000) found that epiphyte load averaged 40 and 93 g DW m^{-2} in *P. sinuosa* and *P. australis* beds, respectively, in Cockburn Sound. Notwithstanding this, both Cockburn Sound and the Adelaide coast receive treated wastewater inputs so epiphyte loads at these locations may represent overestimates for pristine areas. In addition, it is important to remember that epiphyte loads vary seasonally and are highest during summer months when temperature and light availability are increased (S. Nayar pers. comm.). The relatively low epiphyte load at the sites assessed in the present study, in particular in Lacepede Bay, was noticeable at the time of sampling, and contrasts greatly to epiphyte loads found in degraded systems in other areas of South Australia (Figure 23). Interestingly, at the time of sampling in autumn 2005, Drain M was not flowing, yet the corresponding Impact site (Beachport Impact) had the highest epiphyte loads of the study. Higher epiphyte loads at this site and the Beachport Control site may be linked to the presence of nutrient rich upwellings in this area of the South East (see Section 3.2.2. Nutrients).

Epiphyte loads generally varied little between Control and Impact areas within a site (Figure 22). A clear exception to this situation is demonstrated in autumn 2005 at the Beachport sites, where the Impact site supports a higher epiphyte load. The significant interaction effect seen in the ANOVA model (Table 11) indicates that this is not a general phenomenon. There was no overall effect of season on epiphyte load ($P = 0.248$) (Table 11).

Table 11. ANOVA results for seasonal and spatial variation in average epiphyte loads on seagrasses at Control and Impact sites in the South East of South Australia.

Average epiphyte load				
Source	df	SS	F	*P*
Season	1	21.282	1.347	0.248
Treatment	1	195.968	12.402	**0.001**
Season x Treatment	1	1389.732	87.953	**<0.001**
Site (Treatment)	6	2070.453	131.035	**<0.001**
Error	133	15.801		

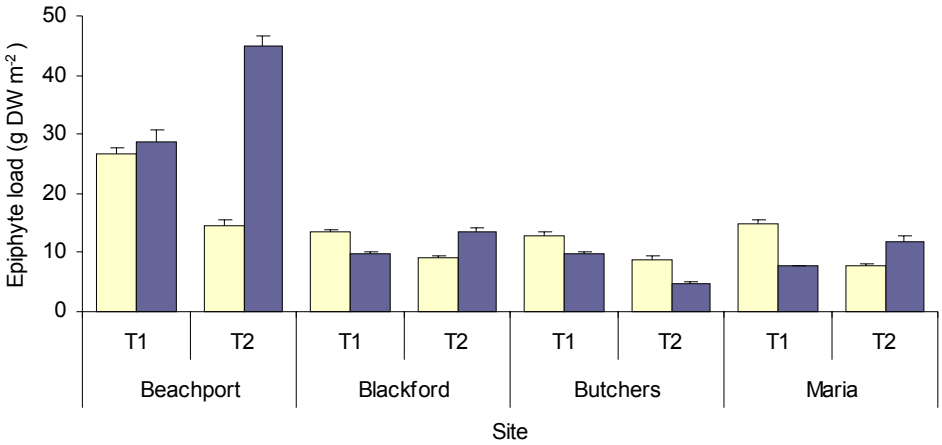

Figure 22. Average epiphyte loads (± standard error) at four Control sites (yellow bars) and four Impact sites (blue bars) in spring 2004 (T1) and autumn 2005 (T2) in the South East of South Australia. ▲ = samples in which epiphytes were removed via the scraping method and 6.97 g m^{-2} was added so epiphyte loads can be compared between samples (see section 2.2.1.).

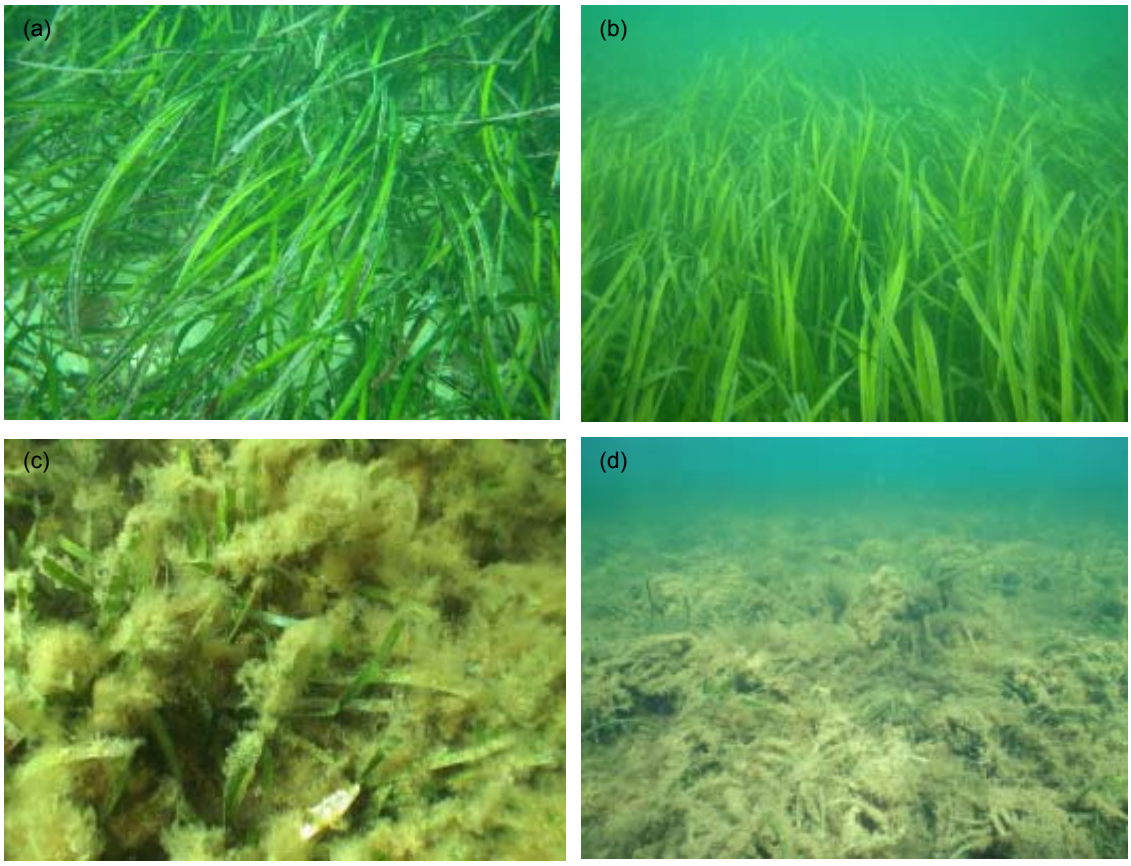

Figure 23. Photographs of seagrasses in Lacepede Bay at the (a) Blackford Control and (b) Butchers Gap Impact sites at the time of sampling in spring 2004 showing relatively little epiphyte growth, and photographs of high seagrass epiphyte loads in (c) Streaky Bay and (d) Kangaroo Island. Photographs (c) and (d) are courtesy of Dr Simon Bryars.

3.1.2. Photosynthetic efficiency

Effective quantum yield varied seasonally ($P < 0.001$; Table 12), with an average of 0.634 ± 0.0048 e photon^{-1} in spring 2004 and 0.687 ± 0.0035 e photon^{-1} in autumn 2005. Effective quantum yield also varied between treatments ($P < 0.001$) and was significantly higher at those sites receiving drain inputs when compared to control locations (Table 12; Figure 24).

Table 12. ANOVA table for seasonal and spatial variation in transformed photosynthetic yield of seagrasses at Control and Impact sites in the South East of South Australia.

Source	df	SS	F	*P*
Season	1	0.109	44.852	**<0.001**
Treatment	1	0.118	48.698	**<0.001**
Season x Treatment	1	0.001	0.556	0.457
Site (Treatment)	6	0.019	7.975	**<0.001**
Error	150	0.002		

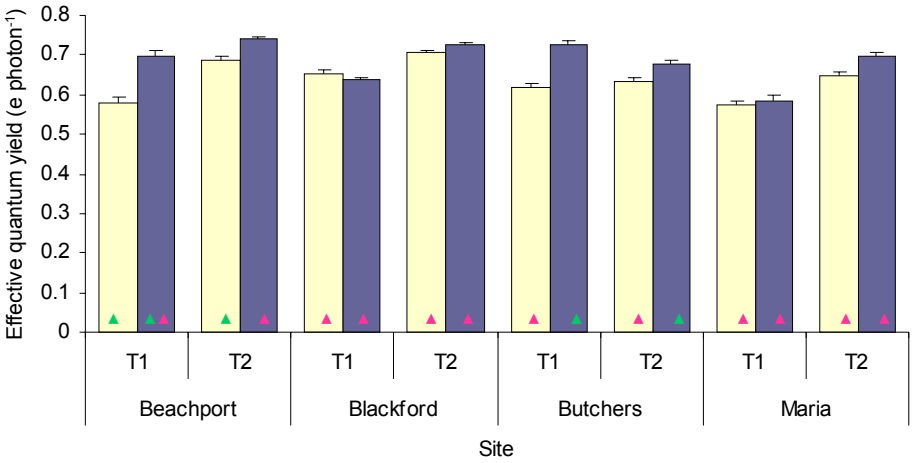

Figure 24. Average effective quantum yield (± standard error) of seagrasses at four control sites (yellow bars) and four sites receiving drain water (blue bars) in Spring 2004 (T1) and Autumn 2005 (T2) in the South East of South Australia (n = 50). ▲ = *Posidonia australis*, ▲ = *P. sinuosa*. ▲▲ = Mixed *P. australis* and *P. sinuosa*.

The high variability between treatments, sites, and seasons is similar to that observed in other studies, and is likely to be related to variations in the light climate at the time of data collection. Factors such as time of day, cloud cover and depth all affect recent light history and play a key role in seagrass photosynthetic efficiency (Ralph *et al*., 1998). The influence of varying light climate on effective quantum yield readings was observed by O'Loughlin (2004) when average yield readings fell from approximately 0.64 to 0.48 e photon^{-1} in response to a light increase of approximately 200 µmol photon m^{-2} s^{-1}. Based on previous studies in which the effective quantum yield has been used to assess seagrass health (e.g. Macinnis-Ng and Ralph, 2003a; O'Loughlin, 2004; Westphalen *et al*., 2005b), it appears that despite large variations, healthy plants generally produce effective quantum yield values

above 0.5 e photon^{-1}. Throughout the duration of this study, at both Control and Impact sites, average effective quantum yield values exceeded 0.57 e photon^{-1} (lowest values were obtained at Maria Control site in Spring 2004), suggesting that seagrasses at all sites in the South East, over both time periods, did not appear to be stressed.

3.2. Water Quality

3.2.1. Salinity, temperature and turbidity

Values of basic water quality parameters such as salinity, temperature and turbidity were measured within the four drains and at Control and Impact sites (Table 13). As expected, salinities within the four drains were substantially lower than at Control and Impact sites (Figure 25a). During spring 2004, the average salinity within the four drains was 10.4 ± 3.5 ppt, while in autumn 2005 salinities averaged 27.0 ± 9.7 ppt. The increase in salinity in autumn is due to elevated salinities in Maria Creek and Drain M (Figure 25a). Elevated salinities in Maria Creek can be explained by the movement of water from Lacepede Bay into the drain with the tides; a common occurrence during summer and autumn when flow rates are low. At the time of sampling in autumn 2005, Drain M was stagnant and samples were collected from remaining pools of water, with evaporation and concentration of salts explaining the elevated salinity values found in the drain. Salinity values in Lacepede and Rivoli Bays during spring 2004 and autumn 2005 varied between 34 and 40 ppt (Figure 25a). Flow rates were relatively high during the spring 2004 sampling period, yet salinities did not vary substantially between Control and Impact sites, suggesting that the drain water is quickly assimilated into the marine environment. Similar values were obtained by Seddon *et al*. (2003) at various sites in the northern end of Rivoli Bay, with salinities varying between 34.6 and 37.1 ppt.

The discharge of treated wastewater, stormwater and riverine inputs all represent sources of freshwater into coastal systems. While numerous studies have investigated salinity tolerances of various seagrass species, there is little information available on those species present in the South East, including species of *Posidonia* and *Amphibolis* (Westphalen *et al.*, 2005a). Nevertheless, species within these genera are thought to tolerate a wide range of salinities based on some of the locations they inhabit. For example, Walker (1985) found *A. antarctica* growing at 57 ppt in Shark Bay, Western Australia, and species of *Amphibolis* and *Posidonia* inhabit upper Spencer Gulf and Gulf St Vincent, South Australia (Edyvane 1999), where salinities exceed 40 ppt during summer months (Nunes and Lennon, 1986; de Silva Samarasinghe and Lennon, 1987). While little information is available on the tolerance of *Amphibolis* to lowered salinities, species of *Posidonia* grow in Oyster Harbour, Western Australia, where salinities vary seasonally between 25 and 37 ppt (Hillman *et al*., 1990). Experimental studies have confirmed that species of both *Amphibolis* and *Posidonia* are tolerant of reduced salinities over short time periods. Tyerman *et al.* (1984) demonstrated that salinities as low as 13 ppt did not affect the growth of *P. australis*. More recently, O'Loughlin (2005) found that exposure of adult *A. antarctica* and *P. angustifolia* to salinities

as low as 0 ppt, for 72 hours, failed to significantly reduce photosynthetic efficiency; however, photosynthetic efficiency was reduced after 10 days exposure to 0 ppt, and plant death was observed after exposure at this concentration for seven weeks. Westphalen *et al*. (2005b) identified that *A. antarctica* seedlings and *P. angustifolia* fruits were less tolerant of reduced salinities than adult plants.

The high tolerance of adult *Amphibolis* and *Posidonia* species to varied salinities as described above, and the apparent fast assimilation of drain water into the receiving environment, suggests that reductions in salinities as a result of drain discharges in the South East are unlikely to affect seagrass health at anything other than small spatial scales. Nonetheless, given that seagrass propagules appear to be more sensitive to reduced salinities (Westphalen *et al*., 2005b), it is possible that drain discharges will prevent the recruitment of seedlings into areas receiving such inputs.

Table 13. Average water quality parameters taken at various sites in the South East of South Australia. † = No measurements obtained due to instrument error. ‡ = No measurements were taken within the Butchers Gap Drain during the autumn 2005 sampling period as the drain was dry.

Site	Turbidity (NTU)	Temp (ºC)	Salinity (‰)
Spring 2004			
Beachport, Drain M	132.00	14.01	21
Beachport Impact	26.25	15.93	35
Beachport Control	182.00	16.90	40
Blackford Drain	†	19.60	9
Blackford Impact	44.75	17.00	38
Blackford Control	45.30	16.90	38
Maria Creek	†	18.90	6
Maria Impact	36.80	17.93	38
Maria Control	†	16.70	38
Butchers Gap Drain	†	17.90	6
Butchers Impact	†	16.90	38
Butchers Control	†	17.45	38
Autumn 2005			
Beachport, Drain M	224.50	17.80	40
Beachport Impact	78.75	15.60	36
Beachport Control	87.90	18.10	34
Blackford Drain	206.67	17.10	8
Blackford Impact	168.00	16.23	36
Blackford Control	113.67	16.40	36
Maria Creek	253.00	17.10	33
Maria Impact	128.67	16.50	36
Maria Control	114.67	16.27	36
Butchers Gap Drain	‡	‡	‡
Butchers Impact	78.03	16.37	36
Butchers Control	66.87	16.40	36

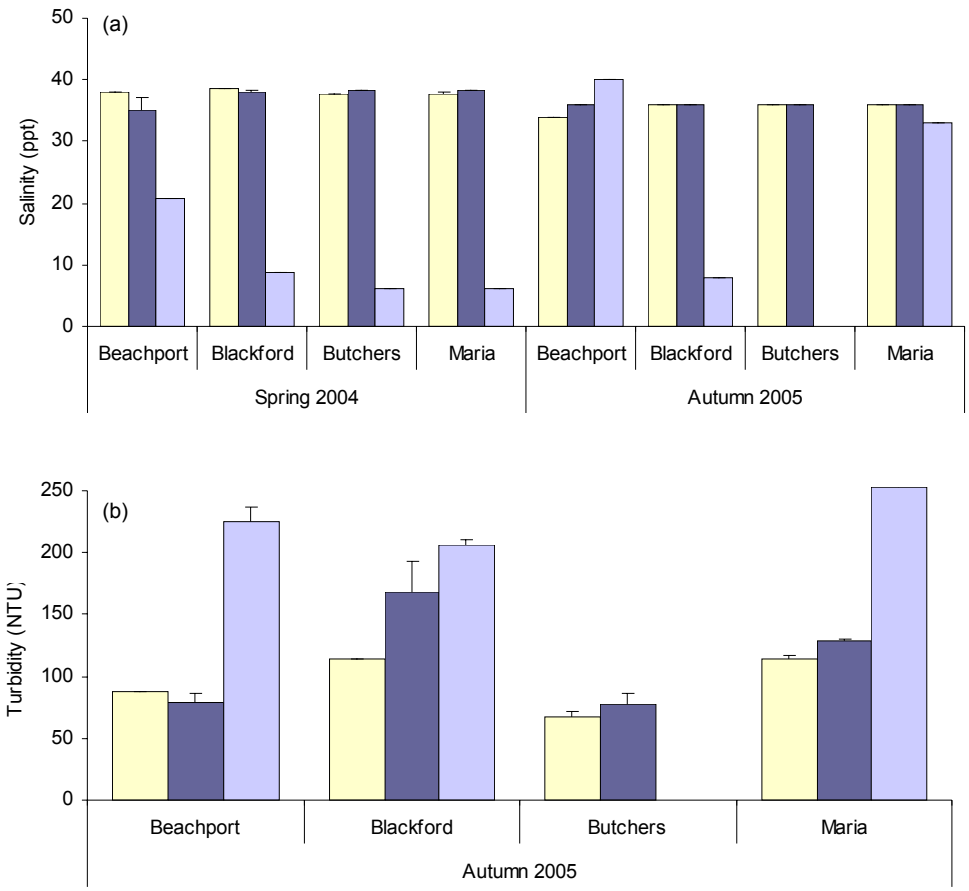

Figure 25. Average (a) salinity and (b) turbidity (± standard error) at the Control (yellow bars), Impact (dark blue bars) and drain sites (pale blue bars) at four locations in the South East of South Australia. The Butchers Gap Drain was completely dry at the time of sampling in autumn 2005, so no data are available.

During autumn 2005, turbidity levels were substantially higher within the drains than at Control and Impact sites, but there was little difference between Impact and Control sites (Table 13; Figure 25b). Due to instrument error, turbidity readings were not obtained in spring 2004. However, water within the drains was visibly turbid at this time. The turbid nature of the water within Drain M was clearly visible both within the drain and in the receiving environment, where a plume of turbid water could clearly be seen (Figure 26). The presence of such plumes is likely in the waters adjacent to drain discharges in other areas, and was previously captured on an aerial photograph of the Lake Frome drain discharge at the southern end of Rivoli Bay (Figure 27). Based on our observations made adjacent to the Drain M discharge in spring 2004, and evidence of turbid waters from aerial photographs, it is likely that the waters adjacent to drain discharges in the South East are subjected to periods of elevated turbidity, especially during winter and spring when flow rates are at a maximum.

Increased turbidity is thought to be a contributing factor to reduced seagrass health and seagrass loss, predominantly through a reduction in light. Light reductions, and subsequent declines in seagrass health have been observed in other areas. For example, Cabello-Pasini

et al. (2002) identified that frequent storms reduced light availability and led to reduced seagrass biomass and leaf carbohydrate in *Zostera marina*. Similarly, light deprivation as a result of increased turbidity has been shown to reduce the flowering intensity of the seagrass *Enhalus acoroides* (Rollon *et al.* 2003). Direct links with increased turbidity and seagrass loss have also been recognised (e.g. Preen *et al.*, 1995; Ward *et al.*, 2003; Campbell and McKenzie, 2004). For example, Preen *et al*. (1995) suggested that increased turbidity following cyclonic conditions at Harvey Bay, Queensland, caused significant losses of *Halophila spinulosa*, *H. ovalis* and *Halodule uninervis.* Similarly, the loss of 95% of a *Z. capricorni* meadow at Sandy Strait, Queensland, was directly linked with a 2-3 fold increase in turbidity following a flood (Campbell and McKenzie, 2004).

Numerous researchers have experimentally tested the effect of reduced light on seagrasses (e.g. Neverauskas, 1988; Fitzpatrick and Kirkman, 1995; Longstaff and Dennison, 1999; Moore and Wetzel, 2000; Ruiz and Romero, 2001; Peralta *et al.*, 2002). These studies have shown that reductions in light can lead to decreased seagrass biomass, leaf density, shoot density, leaf length, leaf growth rate and photosynthetic activity (Neverauskas, 1988; Fitzpatrick and Kirkman, 1995; Moore and Wetzel, 2000; Peralta *et al.*, 2002) and can ultimately lead to seagrass loss (Longstaff and Dennison, 1999). In one such study, 50% shading of *Posidonia* resulted in significant decreases in standing crop, leaf density, leaf length and shoot density over a 6 to 12 month period (Neverauskas, 1988). Shorter-term responses have been observed in other seagrass species and in response to further light reductions. The minimum light required by seagrasses for survival is thought to vary between 8 to 20% surface irradiance depending on species (Dennison *et al*., 1993; Cabello-Pasini *et al.*, 2002), and the importance of light to seagrass production can be shown by comparing the maximum depth of seagrass distribution given different degrees of water clarity. Such a comparison illustrates the relationship between high turbidity and loss of seagrass production at the lower depth limits (Figure 28).

Figure 26. Photographs of (a and b) turbid water within the Drain M outlet, and (c and d) the resultant plume of water within Rivoli Bay during the spring 2004 sampling period.

Figure 27. Aerial photograph of the southern end of Rivoli Bay adjacent to Southend, showing a turbid plume of water caused by discharge from the Lake Frome Drain. Photograph courtesy of the Department for Environment and Heritage (2000).

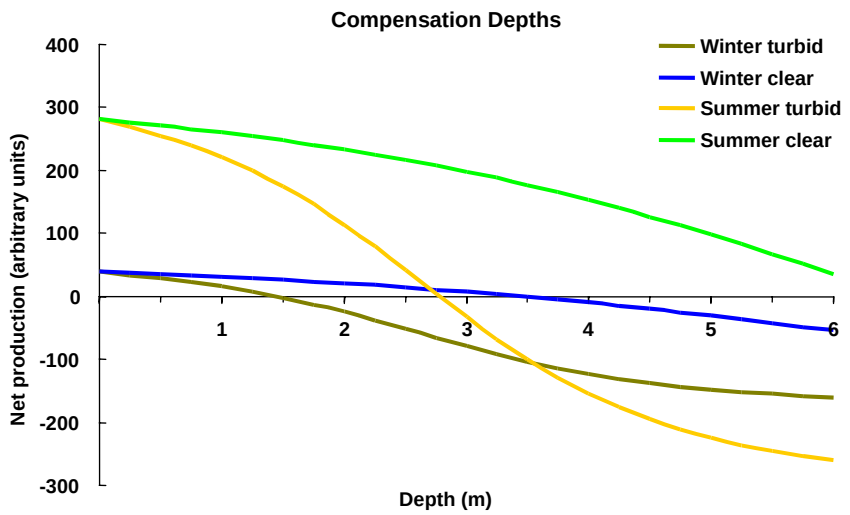

Figure 28. Estimates of production for *Zostera tasmanica* in clear and turbid water during summer and winter by modelling net 24 hour production (Cheshire and Wilkinson, 1990) using photokinetic parameters reported by Bulthius (1983). Figure obtained from Seddon *et al*. (2003).

3.2.2. Nutrients

The concentration of TN, TP, NH_3, and NO_X varied substantially with treatment. As expected, on most occasions, elevated levels of all nutrients were found inside the drains when compared with Control and Impact sites (Figure 29). No seasonal pattern in drain nutrient concentrations was found. In some cases, nutrient concentrations within the drains increased between spring 2004 and autumn 2005, and in others, concentrations decreased (Figure 29).

Despite elevated nutrient concentrations within the drains, nutrient concentrations at the Impact sites adjacent to the drain discharges were not significantly different from Control sites (Table 14), suggesting that nutrient inputs from the drains are rapidly assimilated or disseminated into the marine environment. Concentrations of TN, TP and NH_3 varied significantly between seasons at the Control and Impact sites (Table 14). Total phosphorus and nitrogen concentrations were slightly elevated during autumn 2005, while NH_3 concentrations were highest during spring 2004 (Figure 29a-c). These seasonal differences are likely to reflect natural variations in the processes occurring in the marine environment. Across both time periods, TP concentrations ranged between 'below detectable' (<0.01) and 0.02 mg L^{-1}; TN varied between 0.03 and 0.17 mg L^{-1}; NH_3 varied between 'below detectable' (<0.001) and 0.042 mg L^{-1}; and NO_X varied between 'below detectable' (<0.001) and 0.021 mg L^{-1}. While there is very little historical data on nutrient levels in coastal waters of the South East, a few studies have been conducted (Neverauskas, 1990; Seddon *et al.*, 2003). Nutrient concentrations obtained in these studies are comparable to those found in the present study (Table 15), with the exception of elevated NO_X concentrations in samples collected from the Beachport Control and Impact sites during autumn 2005 (Figure 29; Table 15).

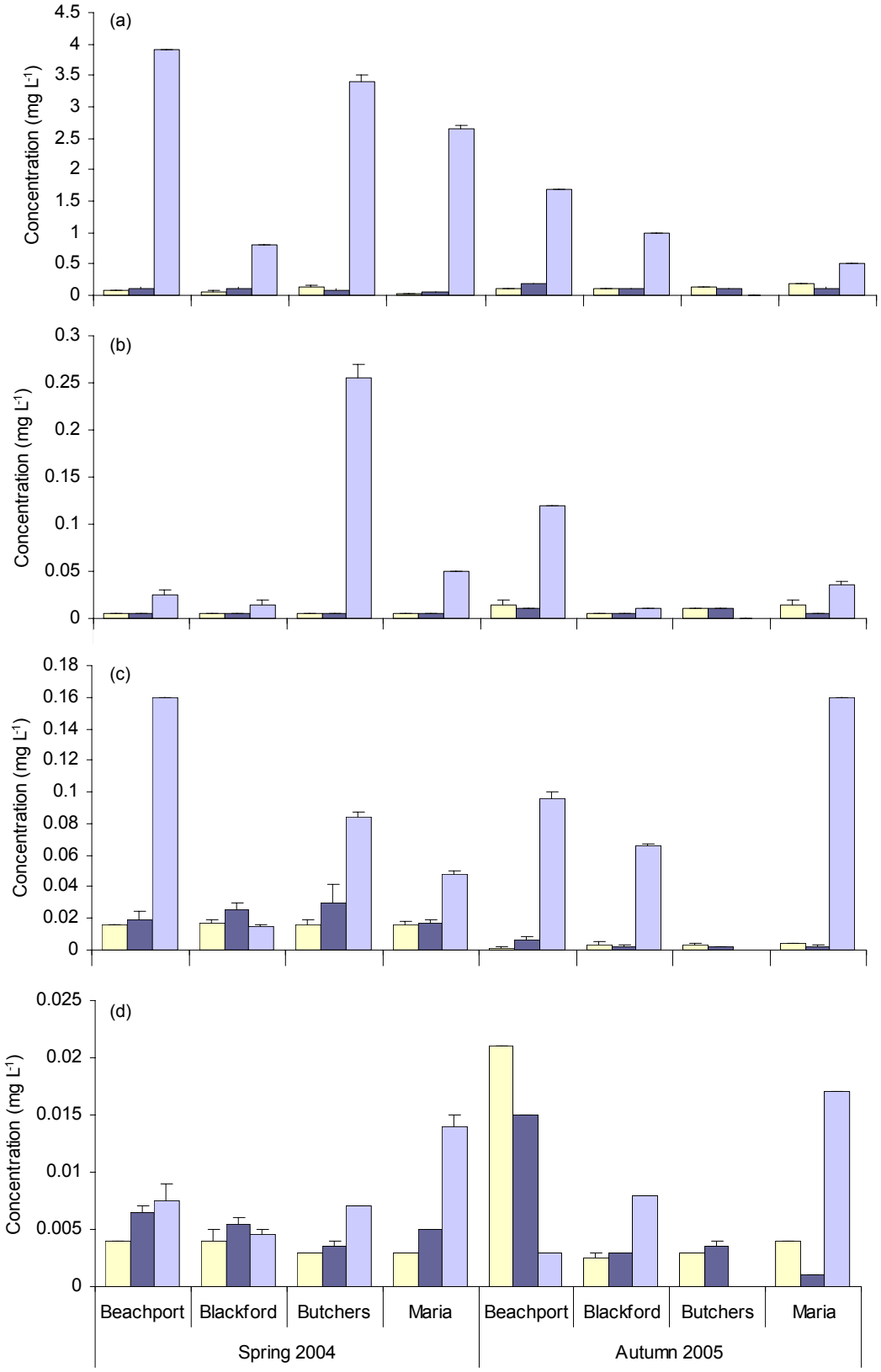

Figure 29. Concentration of (a) total nitrogen, (b) total phosphorus, (c) ammonia and (d) nitrate plus nitrite (± standard error) at Control sites (yellow), Impact sites (dark blue bars) and drain sites (pale blue bars) at four locations in the South East of South Australia (n = 3). NOTE: The Butchers Gap Drain was completely dry at the time of sampling in autumn 2005, so no data on nutrient concentrations are available during that time.

Table 14. ANOVA results for seasonal and spatial variation in nutrient concentrations at Control and Impact sites in the South East of South Australia. NOTE: Nutrient concentrations from within the drains were excluded from the analysis.

Total nitrogen				
Source	df	SS	F	*P*
Season	1	0.020	18.701	**<0.001**
Treatment	1	0.000	0.363	0.553
Season x Treatment	1	0.000	0.363	0.553
Site (Treatment)	6	0.002	2.001	0.109
Error	22	0.001		

Total phosphorus				
Source	df	SS	F	*P*
Season	1	0.000	17.387	**<0.001**
Treatment	1	2.81E-005	3.194	0.088
Season x Treatment	1	2.81E-005	3.194	0.088
Site (Treatment)	6	1.56E-005	1.774	0.151
Error	22	8.81E-006		

Ammonia				
Source	df	SS	F	*P*
Season	1	0.002	83.640	**<0.001**
Treatment	1	0.000	3.865	0.062
Season x Treatment	1	8.1E-005	3.222	0.086
Site (Treatment)	6	1.43E-005	0.536	0.775
Error	22	2.67E-005		

Nitrate plus nitrite				
Source	df	SS	F	*P*
Season	1	4.28E-005	2.921	0.101
Treatment	1	2.81E-007	0.019	0.891
Season x Treatment	1	2.63E-005	1.795	0.194
Site (Treatment)	6	6.91E-005	4.717	**0.003**
Error	22	1.46E-005		

Table 15. Comparison of average maximum and minimum nutrient concentrations in coastal waters in the South East of South Australia between the present study and other studies (table amended from Seddon *et al.* 2003). 1 = Data obtained from Seddon *et al*. 2003, 2 = data from Tanner (in Seddon *et al*. 2003), 3 = Data from Neverauskas 1990 (in Seddon *et al*. 2003) and 4 = data obtained from the current study. nd = no data collected.

Location	Ammonia ($mg\ L^{-1}$)	NO_X ($mg\ L^{-1}$)	TKN ($mg\ L^{-1}$)	Total P ($mg\ L^{-1}$)	Source
Northern Rivoli Bay	0.009 – 0.048	nd	0.19 – 0.42	0.022 – 0.044	1
Southern Rivoli Bay	0.026 – 0.031	nd	0.36 – 0.42	0.024 – 0.026	1
Southend and Point Corner, Rivoli Bay	0.018 – 0.028	nd	0.14 – 0.20	<0.02 – 0.22	2
Various bays *	0.013 – 0.027	0.01 – 0.03	0.41 – 1.21	0.037 – 0.099	3
Lacepede Bay, Rivoli Bay and Nora Creina Bay	0.0012 – 0.017	0.0025 – 0.21	nd	0.005 – 0.015	4

* = sites include Nora Creina Bay, Rivoli Bay, Southend, Bucks Bay, Blackfellows caves, Douglas Point and Port MacDonnell.

The elevated NO_X concentrations at Beachport and Nora Creina during the autumn 2005 sampling period may be explained by the Bonney upwelling, as similar results have been measured in shelf waters during the upwelling season (P. van Ruth, pers. comm.). The Bonney upwelling generally occurs during summer and autumn each year, and takes place when south-easterly winds drive warmer surface waters offshore. During such conditions, deeper, nutrient rich and less saline water is transported to the surface. The Bonney upwelling represents the most significant coastal upwelling along the whole of the Southern Australian coastline (Edyvane, 1999), and is considered to contribute greatly to the considerable coastal productivity of the region. Centres of upwelling in the South East have been identified off Robe, Southend and Port MacDonnell (Lewis, 1981). The upwellings in these regions rarely reach nearshore Lacepede Bay (Figure 30), which may explain the absence of elevated nutrients at sites there. The particularly elevated NO_X concentrations at the Beachport Control site may have been due to local upwellings, in addition to local conditions at the time of sampling, with the large amount of swell at the time of sampling possibly allowing for the release of suspended nutrients bound within the sediments.

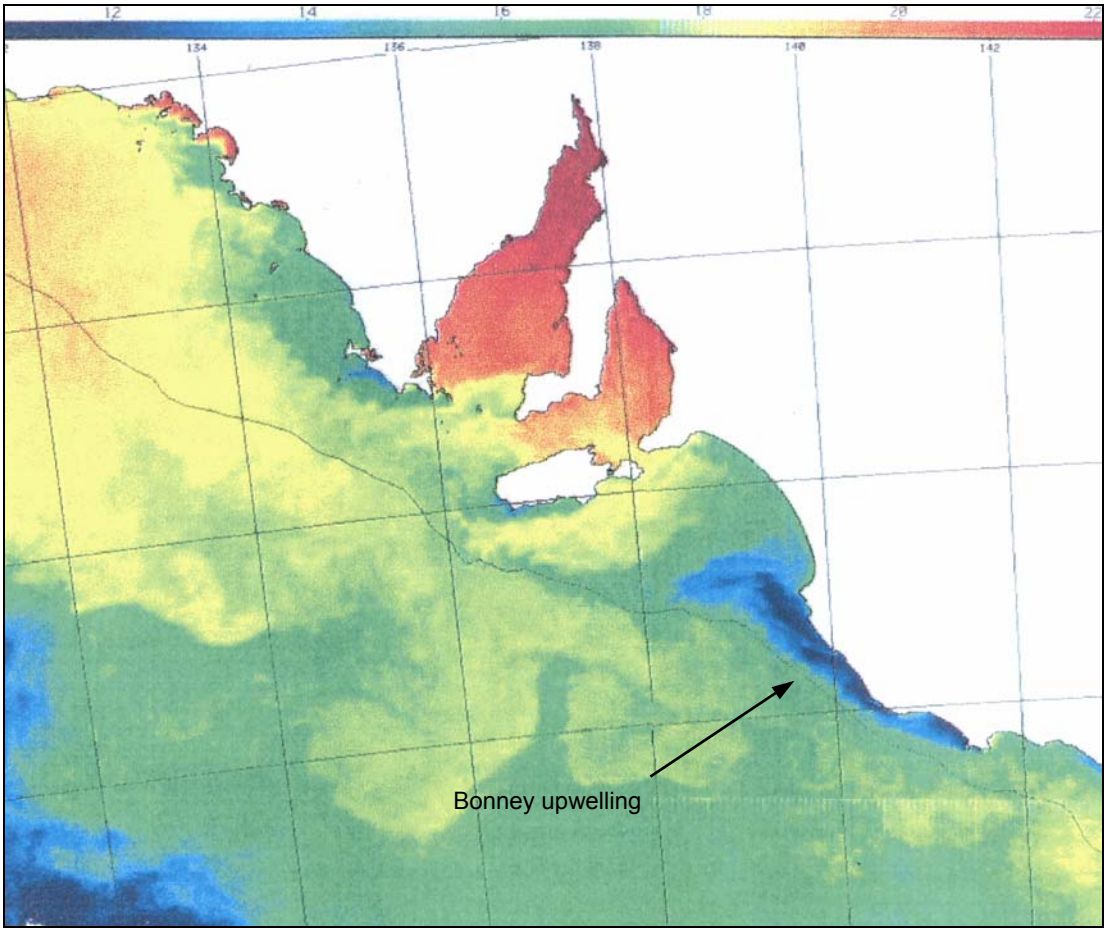

Figure 30. Sea surface temperatures throughout South Australia during February 1995. Clearly visible is the band of cool water along the South East coast, resulting from the Bonney upwelling (CSIRO Marine Laboratories, Remote Sensing Facility, Hobart (NOAA14 Orbit 779 23 Feb)).

Additional water samples were collected from within the drains during the first flush and were analysed for TN, TP, NH_3 and NO_X concentrations. The concentration of TN, TP and NH_3 within the four drains remained within the ranges obtained during the spring 2004 and autumn 2005 sampling periods. However, NO_X concentrations within Blackford Drain and Maria Creek exceeded previously recorded concentrations (Figure 29 and Figure 31). Concentrations of NO_X were particularly elevated in the Blackford Drain with an average concentration of 2.1 ± 0.4 mg L^{-1} found in August, as opposed to 0.0055 ± 0.0055 and 0.003 ± 0.0 mg L^{-1} in spring 2004 and autumn 2005, respectively (Figure 29).

On occasion, nutrient concentrations obtained within the drains exceeded the ANZECC water quality guidelines for slightly disturbed ecosystems for lowland river waters (ANZECC, 2000; Table 16). Using the ANZECC trigger values, TN concentrations were elevated in all drains on most occasions, while the trigger values for TP, NH3 and NO_X were only exceeded on some occasions (Table 16).

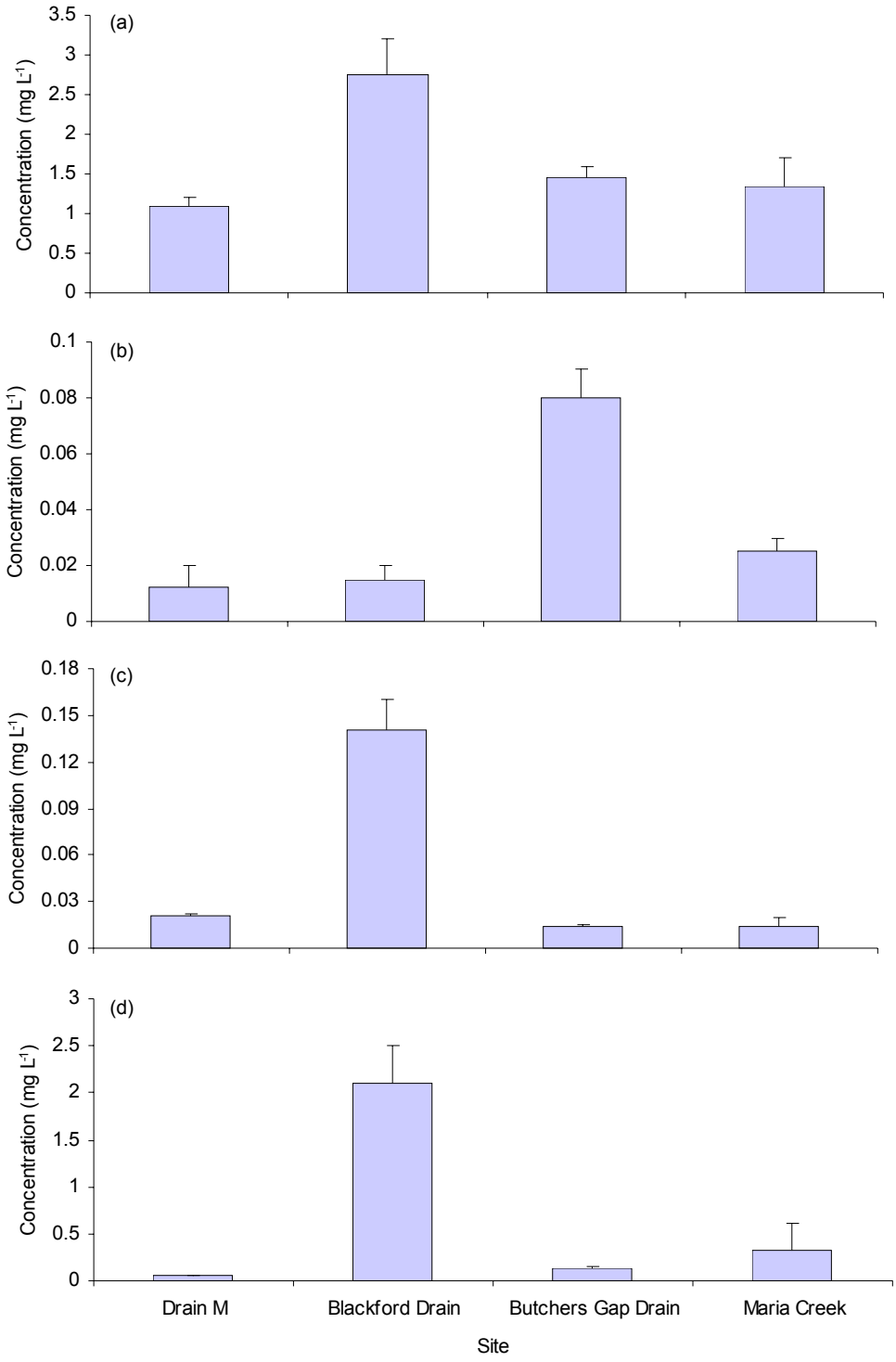

Figure 31. Average concentrations (± standard error) of (a) total nitrogen, (b) total phosphorus, (c) ammonia, and (d) nitrite plus nitrate from within four drains in the South East of South Australia during winter 2005 (n = 2).

Table 16. Average nutrient concentrations in four drains in the South East of South Australia (n = 2). Red = above ANZECC water quality guidelines: an indicator of poor water quality; yellow = below ANZECC water quality guidelines.

Site	TN (mg L^{-1})	TP (mg L^{-1})	Ammonia (mg L^{-1})	NOx (mg L^{-1})
Drain M, spring 2004	3.9	0.025	0.16	0.0075
Drain M, autumn 2005	1.7	0.12	0.096	0.003
Drain M, winter 2005	1.08	0.0125	0.021	0.051
Blackford Drain, spring 2004	0.805	0.015	0.015	0.0045
Blackford Drain, autumn 2005	1	0.01	0.066	0.008
Blackford Drain, winter 2005	2.75	0.015	0.14	2.1
Butchers Gap Drain, spring 2004	3.4	0.255	0.0845	0.007
Butchers Gap Drain, autumn 2005	nd	nd	nd	nd
Butchers Gap Drain, winter 2005	1.45	0.08	0.0145	0.135
Maria Creek, spring 2004	2.65	0.05	0.0475	0.014
Maria Creek, autumn 2005	0.515	0.035	0.16	0.017
Maria Creek, winter 2005	1.35	0.025	0.014	0.321

NOTE: Total phosphorus concentrations: <0.1 mg L^{-1} acceptable. Total nitrogen: <1.0 mg L^{-1} acceptable. Ammonia: <0.1 mg L^{-1} acceptable. NO_X: <0.1 mg L^{-1} acceptable. Trigger values obtained from the ANZECC water quality guidelines in south central Australia for lowland river waters (ANZECC 2000).

While some studies have demonstrated a positive relationship between nutrient availability and the growth of tropical seagrass species (e.g. Udy *et al.*, 1999), it is generally believed that anthropogenic increases in nutrients are detrimental to the health of seagrasses and the ecosystems they support (Ralph *et al.*, 2006). Increases in nutrient levels are commonly linked with agricultural, industrial, sewage and stormwater discharges, and are thought to affect seagrasses by promoting the growth of phytoplankton, epiphytes, and macroalgae, thereby reducing light available to seagrass for photosynthesis (Ralph *et al.*, 2006). The relationship between increased plant growth, in particular epiphytes, and nutrients has been well documented. Neverauskas (1987) demonstrated a positive relationship between proximity to the discharge of sludge from the Port Adelaide Sludge Outfall and increased epiphyte growth and seagrass loss. Similarly, seagrass losses in Cockburn Sound, Western Australia, have been attributed to nutrient inputs and a resultant increase in epiphytic growth (Cambridge *et al.*, 1986; Shepherd *et al.*, 1989). The growth of epiphytes, as a result of nutrient additions, has also been demonstrated experimentally and symptoms associated with nutrient enrichment include a reduction in growth rate, reduced biomass and shoot thinning (Short *et al.*, 1995; Ralph *et al.*, 2006).

Increased epiphytic growth reduces the amount of light available to seagrasses and the productivity of these plants (Sand-Jensen, 1977; Silberstein *et al.*, 1986), and is thought to be the main mechanism that epiphyte growth effects seagrass health. Notwithstanding this, some reports have suggested direct nutrient toxicity and sediment anoxia as mechanisms of mortality (e.g. Van Katwijk *et al.*, 1997; Burkholder *et al.*, 1992). Regardless of the mechanisms involved, increased nutrient concentrations are thought to be the major cause of

seagrass loss worldwide (Ralph *et al*. 2006), and represent one of the main threats to seagrass beds into the future (Duarte, 2002).

3.2.3. Pesticides and herbicides

During both the spring 2004 and autumn 2005 sampling periods, no detectable concentrations of any of the organochlorine (detectable concentration ≥ 0.01 µg L^{-1}) and organophosphate (detectable concentration ≥ 0.10 µg L^{-1}) pesticides and triazine herbicides (detectable concentration ≥ 0.10 µg L^{-1}) were found, either within the drains, or at Control or Impact sites. Additional sampling from within the drains at the time of the first flush also failed to show any concentrations above the detectable limits.

Herbicides and pesticides generally enter the marine environment as run-off from agricultural, forestry and municipal applications. There are a range of herbicides known to enter the marine environment from such uses, the best known and most common of which include atrazine, diuron (DCMU), cybutryne, simazine and glyphosate. While herbicides are generally not considered to have contributed to wide-scale seagrass loss, there has been little research in this area. Various compounds have been found within coastal environments at concentrations likely to impact on seagrass health (e.g. Haynes *et al.*, 2000b). Hence herbicides may play a role in seagrass decline in some areas, and should therefore be considered when assessing the potential mechanisms of seagrass loss. Notwithstanding this, it is important to realise that there is often very little information on herbicide concentrations in the marine environment as well as relatively little information on the physiological impacts of herbicides on seagrasses, particularly in terms of growth and reproduction. The majority of research to date appears to have focussed on acute (i.e. short term) responses to a range of dosages. Such research suggests that atrazine, which is rapidly absorbed by seagrasses through their leaves and roots (Schwarzschild *et al.*, 1994), can significantly reduce the photosynthetic efficiency of some seagrass species at concentrations ≥10 µg L^{-1} (Ralph, 2000, Macinnis-Ng and Ralph, 2003b), and cybutryne has similar effects at concentrations as low as 0.18 µg L^{-1} (Scarlett *et al.*, 1999a). Diuron (3-(3,4-dichlorophenyl)-1,1-dimethylurea) has been found to reduce the photosynthetic capacity of various seagrass species at concentrations as low as 0.1 µg L^{-1} (Haynes *et al.*, 2000b). Glyphosate is the most widely used non-selective herbicide in the world (PAN, 2004) and has been classified as relatively non-toxic to aquatic flora (Tomlin, 1994, in Ralph, 2000).

A range of different herbicides and pesticides are utilised in the South East region, including Glyphosate and triazine herbicides. Given that drain water samples on three occasions failed to contain any traceable concentrations of a variety of herbicides and pesticides it is unlikely that their use is affecting seagrass health in the vicinity of drain discharges. Notwithstanding this, it is important to note that ecotoxicological research on the effects of herbicides and pesticides on species of seagrass present in the South East has not yet been undertaken.

3.2.4. Chlorophyll *a*

During both the spring 2004 and autumn 2005 sampling periods there was substantially higher chlorophyll *a* concentrations in drains compared with water samples taken at Control and Impact sites (Figure 32). The highest chlorophyll *a* concentrations during spring were observed in Butchers Gap Drain (31.04 ± 0.40 µg L^{-1}) and Drain M (30.69 ± 1.30 µg L^{-1}). During autumn 2005, chlorophyll *a* concentrations were lower in all drains, when compared with samples taken during spring 2004, with the exception of Drain M, where 250.2 ± 1.21 µg L^{-1} was recorded; more than eight times the highest concentrations found during spring. Chlorophyll *a* concentrations at Control and Impact sites were rarely different during both time periods. Elevated concentrations adjacent to drain discharges were only apparent in Rivoli Bay, adjacent to the Drain M discharge and Lacepede Bay, adjacent to the Maria Creek discharge during spring 2004 (Figure 32). These sites were the only sites at which the ANZECC water quality guideline trigger values for 'slightly disturbed systems' were exceeded (>1 µg L^{-1}). The concentration of chlorophyll *a* at Control sites remained below ANZECC water quality guidelines (ANZECC, 2000), and varied between 0.2085 and 0.937 µg L^{-1}.

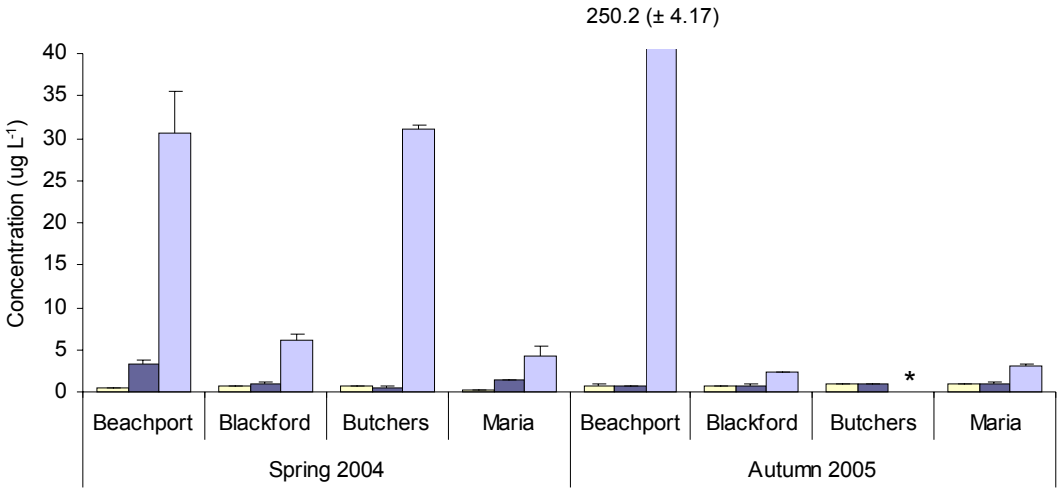

Figure 32. Average concentrations of chlorophyll *a* (± standard error) at Control sites (yellow bars), Impact sites (dark blue bars) and from within drains (pale blue bars) at four locations in the South East of South Australia during spring 2004 and autumn 2005 (n = 2). * = no data.

Elevated nutrient concentrations are known to stimulate the growth of phytoplankton, and as a consequence, strong correlations between chlorophyll *a* and nutrient concentrations have been established. For example, Shepherd *et al*. (1989) documented a strong relationship between elevated concentrations of phosphate and chlorophyll *a* concentrations in Cockburn Sound adjacent to sewage and industrial outfalls. Comparisons between our data on nutrient and chlorophyll *a* concentrations broadly support this. Particularly elevated chlorophyll *a* concentrations (> 15 µg L^{-1}) were identified within Drain M over both sampling periods and in the Butchers Gap Drain in spring 2004 (Table 17). These samples also contained elevated levels of one or more of the nutrients measured (Table 16). Importantly, the relationship

between increased nutrient concentrations and chlorophyll *a* concentrations is not always apparent. For example, chlorophyll *a* concentrations were above those recommended in the ANZECC guidelines in the Blackford Drain during spring 2004, yet all nutrient parameters tested were below ANZECC guidelines. Conversely, elevated nutrient concentrations were found within Maria Creek during both sampling periods and in the Blackford Drain in autumn 2005, yet comparatively low concentrations of chlorophyll *a* were found (Table 16; Table 17). One factor likely to discourage algal growth is increased turbidity of the water, and this may explain why Maria Creek did not contain excessive chlorophyll *a* concentrations during both time periods, despite relatively high levels of nutrients. Nonetheless, this phenomenon is not always the case as Drain M contained elevated concentrations of chlorophyll *a* in spring 2004, together with elevated nutrients when flow was high (Figure 32; Table 16; Table 17).

Table 17. Comparison of chlorophyll *a* concentrations in four drains in the South East of South Australia (n = 2). Red = above ANZECC water quality guidelines; yellow = below ANZECC water quality guidelines.

Site	Chl *a* ($\mu g\ L^{-1}$) ‡
Drain M, spring 2004	30.70
Drain M, autumn 2005	250.2
Blackford Drain, spring 2004	6.14
Blackford Drain, autumn 2005	2.40
Butchers Gap Drain, spring 2004	31.04
Butchers Gap Drain, autumn 2005	nd
Maria Creek, spring 2004	4.17
Maria Creek, autumn 2005	2.95

NOTE: Chlorophyll *a* concentrations: <5 $\mu g\ L^{-1}$ acceptable. Values based on trigger values in the ANZECC water quality guidelines in south-east Australia for lowland river waters (ANZECC, 2000).

3.3. Video Transects

Analysis of video transects in the receiving environment of the Blackford Drain, Maria Creek Drain and Butchers Gap Drain revealed that the most common seagrass genus in the area is *Posidonia* (Figure 33 - Figure 35). Patches of *Amphibolis* were also present, particularly at the Blackford Drain site (Figure 33). Small patches of macroalgae were present adjacent to Blackford Drain and Maria Creek (Figure 33 and Figure 34). Seagrass densities were mostly characterised as 'dense' at all sites. Epiphyte loads adjacent to the drains were generally 'sparse'. However, dense and mid-dense loads were observed in inshore areas and in southern offshore areas at the Blackford Drain and Maria Creek sites (Figure 33 and Figure 34), which could be related to drain discharges. Sand patches were present at all sites, particularly adjacent to the Butchers Gap Drain outlet (Figure 35). While it is possible that the pattern of sand patches at this site is associated with drain discharges, it is impossible to confirm this without hydrodynamic modelling of drain discharges, and the offshore pattern of sand patches appears to be associated more with along shore coastal processes. Maps of

the habitat type and habitat density and epiphyte density are provided in Appendix 3 (Figures 3a – 3c).

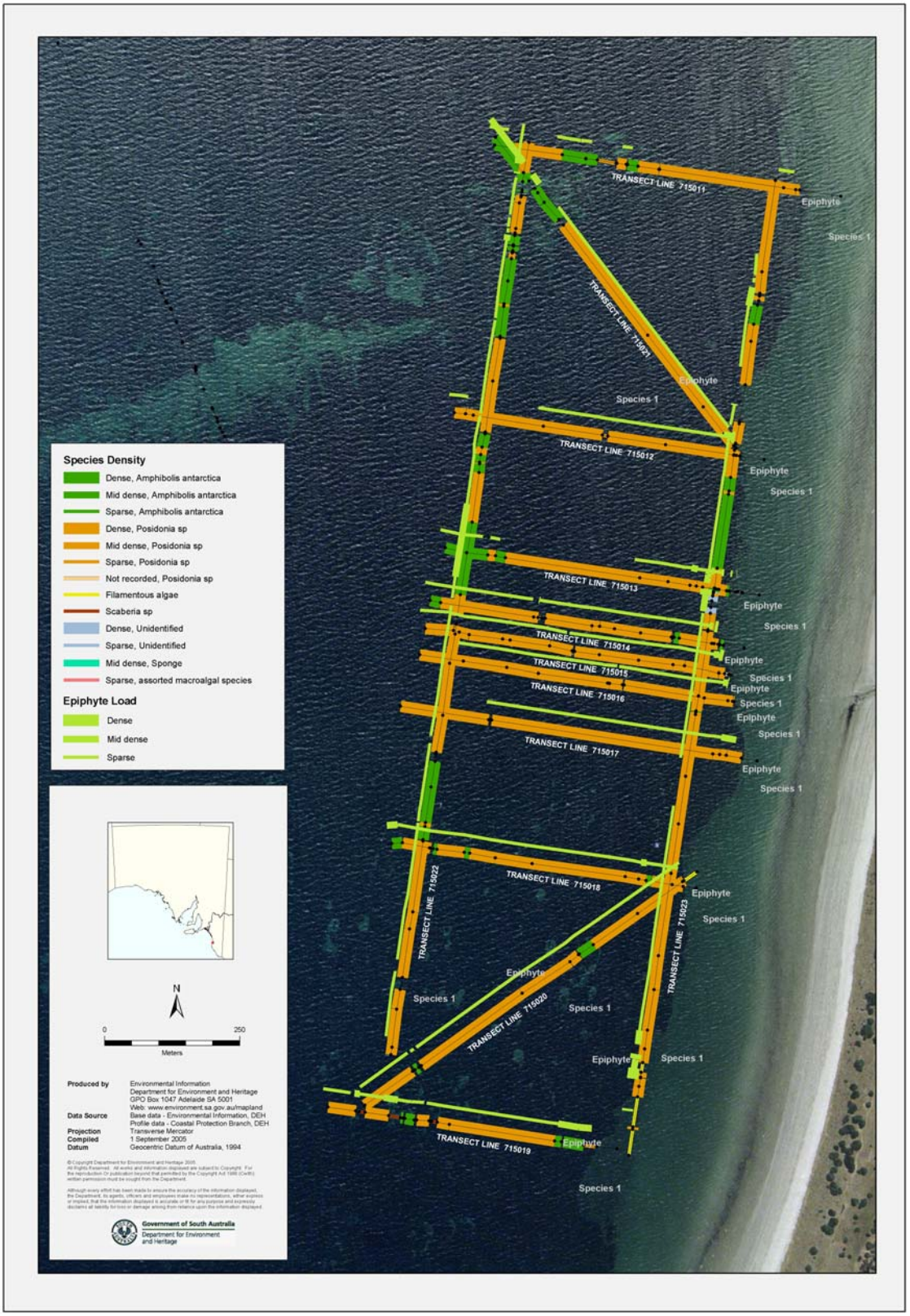

Figure 33. Species density and epiphyte loads along video transects adjacent to the Blackford Drain Outlet.

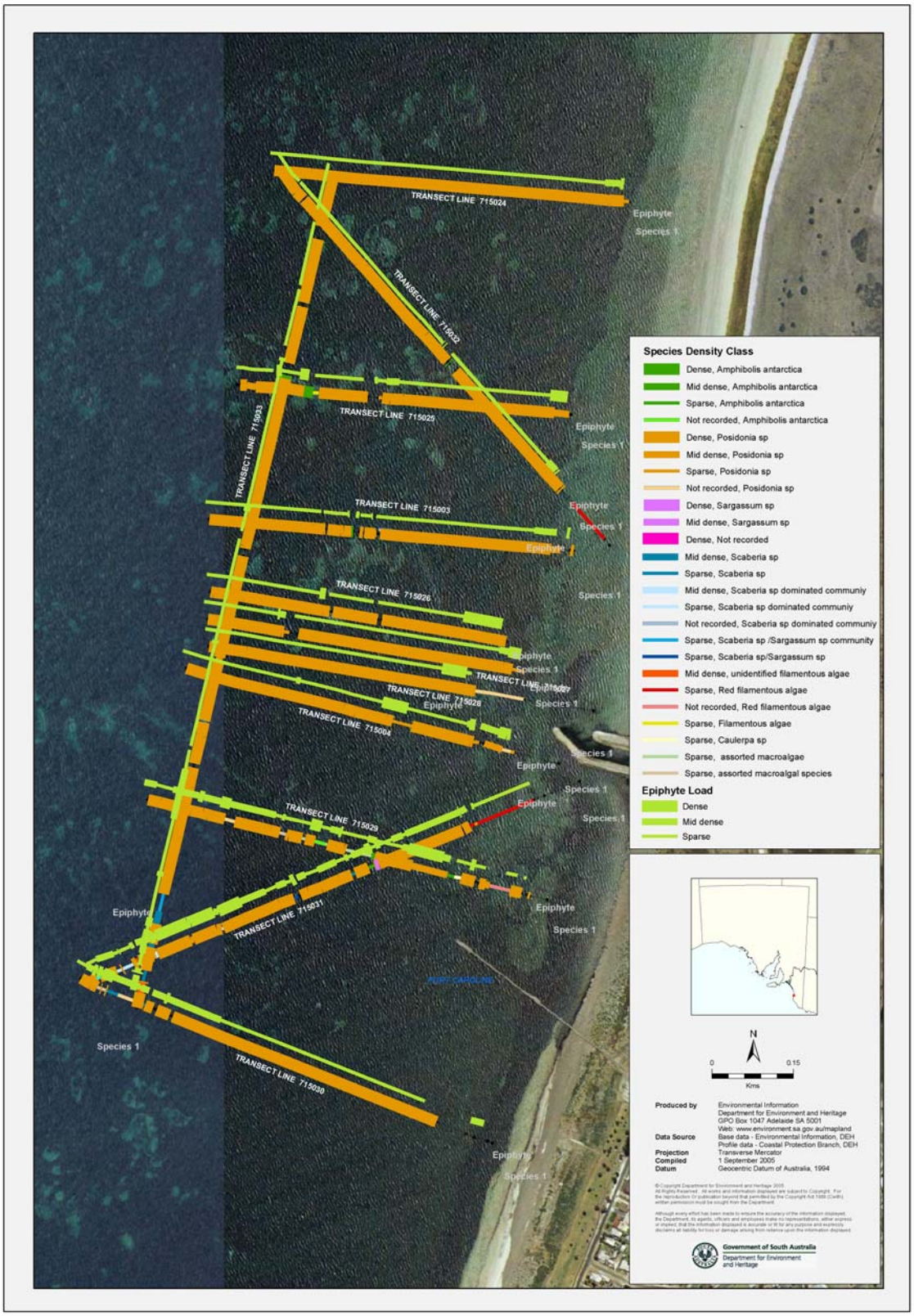

Figure 34. Species density and epiphyte loads along video transects adjacent to the Maria Creek Outlet.

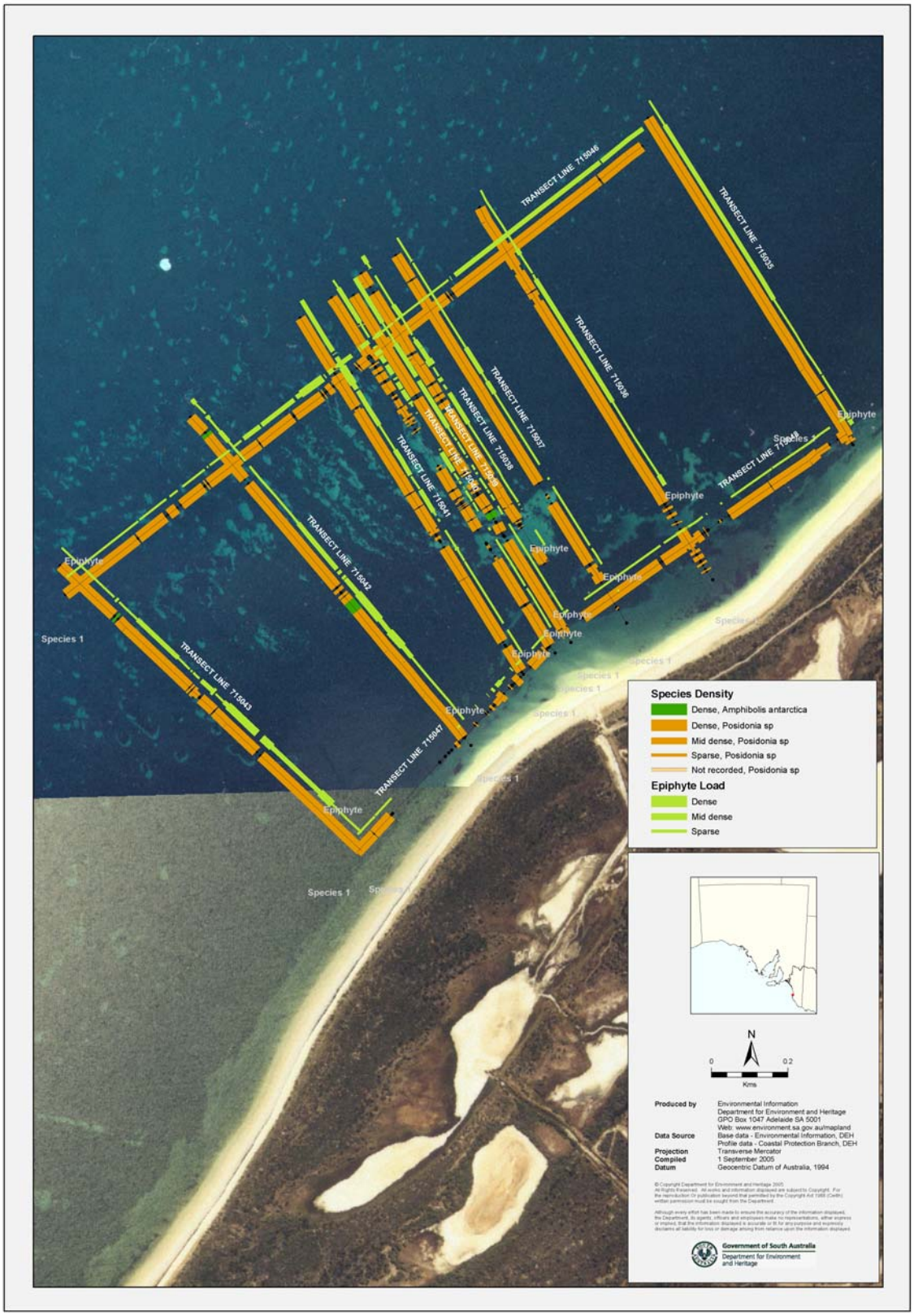

Figure 35. Species density and epiphyte loads along video transects adjacent to the Butchers Gap Drain Outlet.

3.4. Profile Lines

Plots of seafloor height from four profile lines in the Kingston, Butchers Gap and Cape Jaffa regions showed changes over time (Figure 36). The position of the 'blue' line (inshore edge of seagrass) at profile lines 715002 (Cape Jaffa) and 715003 (Kingston) was difficult to determine and as a result we are unable to establish if there have been any changes to the blue line over time. The remaining two profile lines (Profile 715001 – Butchers Gap, and 715004 - Kingston) have varied more over time and although not obvious when compared with previous profile line data (Figure 36), the seagrass line near Butchers Gap (profile 715001) has receded 40 m over the past 5 years, while the seagrass line at Kingston (profile 715004) has receded 84 m over the last 20 years (recorded by field staff while undertaking survey). The recession of seagrasses at these two sites does not appear to have had any significant effect on the seafloor height, suggesting that coastal erosion is not occurring.

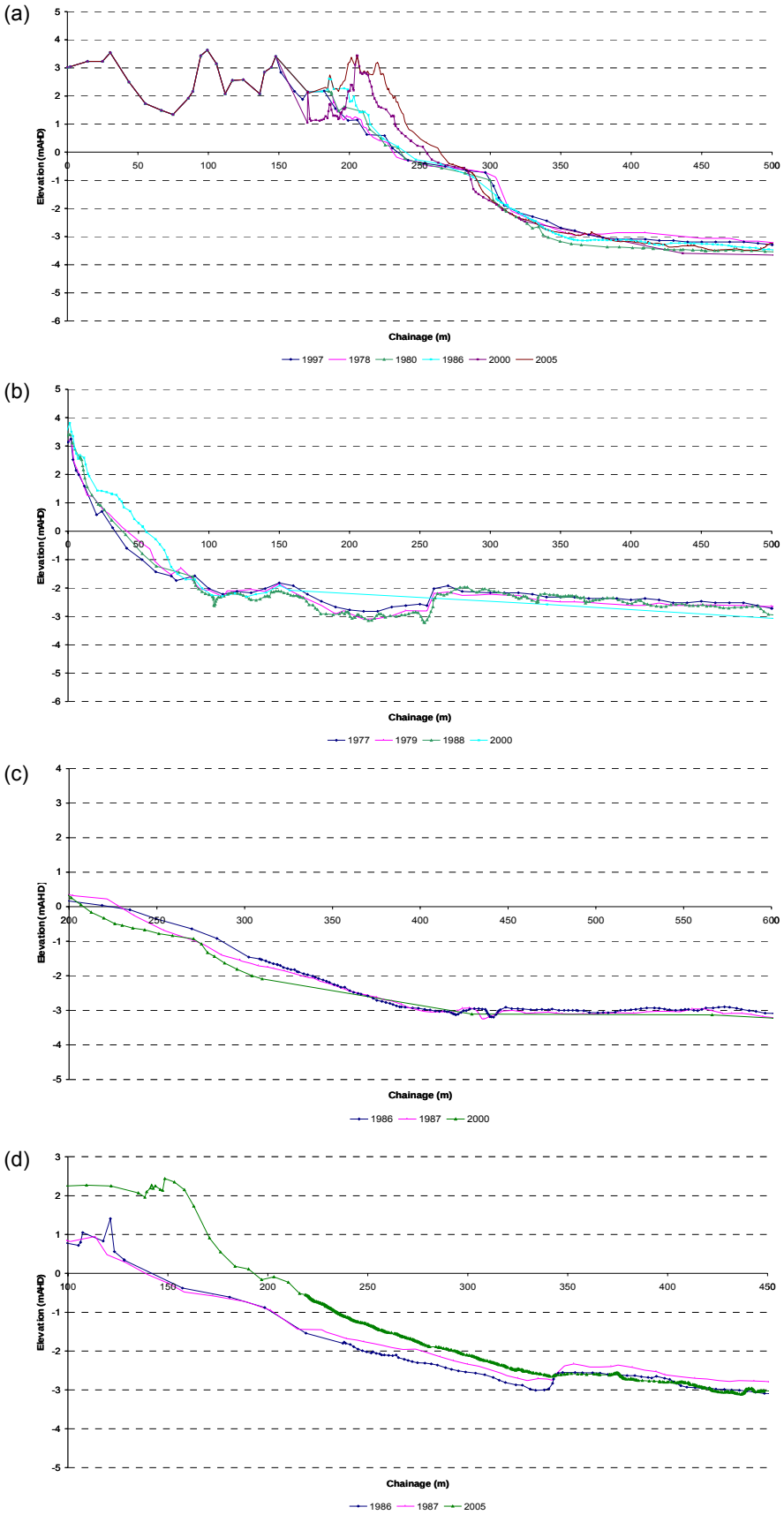

Figure 36. Seabed heights for profile lines (a) 715001 (Butchers Gap), (b) 715002 (Cape Jaffa), (c) 715003 and (d) 715004 (Kingston) for selected years. Data courtesy of the Coastal Protection Branch, Department for Environment and Heritage.

4. CONCLUSION

In comparison to Control sites, seagrass health adjacent to the discharges from Drain M, Blackford Drain and Maria Creek appears to be impacted, as demonstrated by reduced seagrass leaf densities and leaves of reduced stature. The level of impact adjacent to the drains tends to reflect the volume of water discharged and the size of each drain and its associated subsidiaries. Of the four drains investigated as part of the current study, Drain M and its subsidiaries have a combined length of 280 km and discharge the greatest volume of water. Recently, Seddon *et al*. (2003) suggested that approximately 78.8% of the seagrasses within the vicinity of the Drain M discharge have been lost, with highest rates of decline coinciding with the construction of Drain M and major works and enlargement of the drain throughout the 1960s. Seagrass losses appear to be continuing (Seddon *et al.*, 2003) and existing seagrass patches have characteristics of seagrasses commonly found in degraded systems. The next largest drain is the Blackford Drain which has a combined length of 168 km and discharges an annual average of 20,215 ML. According to profile line data, the inshore edge of seagrasses in the vicinity of this drain has receded 84 m over the past 20 years. The Maria Creek Drain and associated drains are substantially smaller than the Blackford Drain, with a combined length of 28.6 km. While seagrasses adjacent to the Maria Creek Drain are impacted in a similar way to those adjacent to the Blackford Drain, a greater recovery of seagrasses adjacent to the Maria Creek Drain between spring and autumn was apparent. The smallest drain assessed as part of the present study was the Butchers Gap Drain, which has a combined length of 15.2 km. Despite differences in seagrass species at Control and Impact sites (*P. sinuosa* versus *P. australis*), the seagrasses adjacent to this drain appeared healthier than those at other sites. Notwithstanding this, profile line data have revealed that the inshore edge of seagrasses adjacent to the Butchers Gap Drain has receded 40 m over the past five years.

In an effort to identify the causes of impacts to seagrasses adjacent to the drains, various water quality parameters were tested. Water within the drains was characterised by reduced salinities, increased turbidity, and high concentrations of nutrients and chlorophyll *a*. Seagrass species are generally relatively tolerant of varied salinities (e.g. Walker, 1985; Hillman *et al*., 1990; Edyvane, 1999), and samples taken adjacent to the drains had similar salinities to those recorded at Control locations, suggesting that the drain water is rapidly assimilated into the marine environment. Seagrasses are far less tolerant of changes in water quality parameters that reduce the amount of light available to them (e.g. turbidity, shading, increased epiphyte loads; Preen *et al.*, 1995; Rollon *et al*., 2003; Ward *et al.*, 2003). In the current study it was found that water within the drains is relatively turbid and when released from the drains, the water can produce plumes of turbid water. The presence and persistence of such plumes is likely to depend on coastal geomorphology, broad-scale hydrodynamics, and local environmental conditions, with plumes likely to persist in smaller enclosed bays (e.g. the northern and southern end of Rivoli Bay), rather than in more open areas (e.g.

Lacepede Bay). While elevated nutrient concentrations were found within the drains, these concentrations did not appear to persist in the marine environment. Furthermore, epiphyte loads adjacent to drain discharges were low, providing additional evidence that the input of nutrients into the marine environment is not primarily responsible for reduced seagrass health adjacent to the drains in Lacepede Bay. Nonetheless, increased epiphyte loads may have been a contributing factor to the seagrass loss in the more enclosed waters of northern Rivoli Bay.

It appears that the drains investigated in the present study are having a negative impact upon nearby seagrass beds and although water quality within the drains is poor, the drain water appears to be rapidly assimilated into the marine environment. Seagrasses are likely to be able to tolerate the individual effects of increased turbidity, reduced salinities and increased nutrients during pulse events better than a combination of these factors for longer periods; this may explain the impacts to seagrasses observed adjacent to the drain discharges. Remaining seagrass beds in northern Rivoli Bay are also likely to be affected by high wave energy and seabed instability resulting from previous seagrass losses in the area (Seddon *et al*., 2003).

One of the major concerns with regard to impacts on seagrasses is that initial seagrass loss can destabilise an area resulting in further losses (as is being observed in Rivoli Bay, Seddon *et al*., 2003). The present study has revealed that the inshore limit of seagrass distribution at Kingston and Butchers Gap has receded, although, with the information available, we are unable to conclude that this is a direct result of the drain discharges or whether it is a more widespread phenomenon occurring in Lacepede Bay. To ensure that seagrasses within the vicinity of drain discharges in Lacepede Bay are not undergoing long-term and possibly irreversible damage, a monitoring program has been designed (see Section 6) and its implementation is highly recommended.

5. SUGGESTIONS FOR FUTURE WORK

- Initiation of a long term monitoring program. Details of a suggested monitoring program are provided in section 6.

- Detailed hydrodynamic modelling is required in Rivoli Bay and Lacepede Bay. In particular, modelling of the fate of drain discharges during different environmental conditions is required.

- Investigation of potential impacts of drain discharges to other ecosystems in the South East. Other drains discharging along the South East coast, not investigated as part of the present study may also be negatively impacting upon nearby ecosystems. Research will enable the identification of such ecosystems (which will include reef habitats), and the extent of an impact, if it exists.

- Research to determine tolerance limits of local seagrasses (particularly *Posidonia* and *Amphibolis* species) to chronic and pulsed exposures to various potential stressors such as elevated nitrogen, elevated turbidity, and reduced salinity .

- Further research is needed to determine the usefulness of various seagrass health parameters at different locations within South Australia. Seagrass characteristics are likely to vary naturally with season and locality and further research on these characteristics will enable scientists to differentiate between natural variations and changes resulting from anthropogenic disturbances. Seagrass characteristics should include epiphyte biomass, leaf density, shoot density, leaf length, leaf width, leaf area index, and productivity.

6. SUGGESTED MONITORING PROGRAM

6.1. Profile Lines

Regular monitoring of profile lines over time is an effective method to determine shifts in the nearshore seagrass line. There has been a seaward shift in the seagrass edge at Butchers Gap and Maria Creek and it is important to determine if this shift is associated with proximity to drain discharges or if it is occurring at a number of places in Lacepede Bay. If there are changes in the seagrass line, regular monitoring of profile lines will also enable us to determine the rate of movements.

Monitoring of profile lines adjacent to the drains (i.e. Profile lines 715015 (Blackford Drain), 715028 (Maria Creek), 715039 (Butchers Gap) and those adjacent Drain M, Rivoli Bay) set up as part of the present study, in addition to those set up in 1977 by DEH, should be conducted on an annual basis. If no further movement in the seagrass line occurs after three years of annual monitoring, then monitoring may be extended to every two or three years.

6.2. Aerial Photography

Mapping from aerial photographs taken over time can be an effective means of assessing changes in nearshore seagrass distribution. While monitoring profile lines over time will pick up any small changes in the seagrasses at individual locations, comparisons of aerial photographs over time provide information on changes over a larger-scale. Aerial photographs in areas of seagrass cover in Lacepede and Rivoli Bays should be undertaken approximately every five years. In addition, historical aerial photographs should be analysed to determine the extent of loss and date in which the loss occurred.

6.3. Monitoring Seagrass Beds and Water Quality

Monitoring of seagrass meadows should be undertaken annually, as seagrass health can give an early warning of impending problems. Seagrass parameters that should be assessed should have previously shown a response to environmental conditions, be widely accepted as being appropriate, be easily obtained, and be non-destructive. Such parameters will include seagrass leaf/shoot density and maximum leaf length. Each monitoring event should assess seagrass parameters during summer and assess seagrasses adjacent to drain discharges and control locations. At the time of seagrass bed monitoring water quality samples should be collected from the same sites and within the drains. Samples should be analysed for turbidity, salinity and various nutrient parameters.

7. REFERENCES

ANZECC (2000) Australian Guidelines for water quality monitoring and reporting. National Water Quality Management Strategy No. 7, October 2000. Prepared by the Australian & New Zealand Environment and Conservation Council and the Agriculture and Resources Management Council of Australia and New Zealand. Australian Water Association, Artarmon.

Bell, J. D., Pollard, D. A. (1989) Ecology of fish assemblages and fisheries associated with seagrasses. In Larkum, A. W. D., McComb, A. J., Shepherd, S. A. (eds.). *Biology of Seagrasses*. Elsevier Science Publishers B. V., New York Pp. 565-609.

Borum, J. (1985) Development of epiphytic communities on eelgrass (*Zostera marina*) along a nutrient gradient in a Danish estuary. *Marine Biology* **87**, 211-218.

Bryars, S. (2003) An inventory of important coastal fisheries habitats in South Australia. Fish Habitat Program, Primary Industry and Resources South Australia, Adelaide.

Bryars, S., Neverauskas, V., Brown, P., Gilliland, J., Gray, L., Halliday, L. (2003) Degraded seagrass meadows and evidence of eutrophication in Western Cove, Kangaroo Island. Fish Habitat Program, Primary Industries and Resources, Adelaide. 25 pp.

Bulthius, D. A. (1983) Effects of in situ light reduction on density and growth of the seagrass *Heterozostera tasmanica* (Martens ex Ashers.) den Hartog in Western Port, Victoria, Australia. *Journal of Experimental Maine Biology and Ecology* **67**, 91-105.

Burkholder, J. M., Mason, K. M., Glasgow, H. B. (1992) Water column nitrate enrichment promotes decline of eelgrass *Zostera marina*: evidence from seasonal mesocosm experiments. *Marine Ecology Progress Series* **81**, 163-178.

Cabello-Pasini, A, Lara-Turrent, C, Zimmerman, RC (2002) Effect of storms on photosynthesis, carbohydrate content and survival of eelgrass populations from a coastal lagoon and the adjacent open ocean. *Aquatic Botany* **74**, 149–164.

Cambridge, M. L., Chiffings, A. W., Brittan, C. Moore, L., McComb, A. J. (1986) The loss of seagrass in Cockburn Sound, Western Australia. II. Possible causes for seagrass decline. *Aquatic Botany* **24**, 269-285.

Campbell, S. J., McKenzie, L. J. (2004) Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. *Estuarine Coast and Shelf Science* **60**, 477-490.

Cheshire, A. C., Wilkinson, C. R. (1990) Modelling the photosynthetic production by sponges on Davies Reef, Great Barrier Reef. *Marine Biology* **109**, 13-18.

Clarke, S. M., Kirkman, H. (1989) Seagrass Dynamics. In: Larkum, A. W. D., McComb, A. J., Shepherd, S. A. (eds.). *Biology of seagrasses. A treatise of the biology of seagrasses with special reference to the Australian region*. Elsevier, Amsterdam. Pp. 304-345.

Connolly, R., Jenkins, G., Loneragan, N. (1999) Seagrass dynamics and fisheries sustainability. In: Butler, A., Jernakoff, P. (eds.). *Seagrasses in Australia*. CSIRO Publishing, Victoria. Pp. 25-64.

D'Arcy, R. D., Austin, S., Cohen, B., Mooney, P., Murray, D. (1984) South East environmental profile study. M. L. Morris (eds.). Assessments Branch, South Australian Department of Environment and Planning.

Delgado, O., Ruiz, J., Pérez, M., Romero, J., Ballesteros, E. (1999) Effects of fish farming on seagrass (*Posidonia oceanica*) in a Mediterranean bay: seagrass decline after organic loading cessation. *Oceanologica Acta* **22**, 109-117.

Dennison, W. C., Orth, R. J., Moore, K. A., Stevenson, J. C., Carter, V., Kollar, S., Bergstrom, P. W., Batiuk, R. A. (1993) Assessing water quality with submerged aquatic vegetation. *Bioscience* **43**, 86-94.

De Silva Samarasinghe, J. R., Lennon, G. W. (1987) Hypersalinity, flushing and transient salt-wedges in a tidal gulf – an inverse estuary. *Estuaries, Coastal and Shelf Science* **24**, 483-498.

Duarte, C. M. (2002) The future of seagrass meadows. *Environmental Conservation* **29**, 192-206.

Duarte, C. M., Chiscano, C. L. (1999) Seagrass biomass and production: a reassessment. *Aquatic Botany* **65**, 159-174.

Edyvane, K. S. (1999) Conserving marine biodiversity in South Australia – Part 2 – Identification of areas of high conservation value in South Australia. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. 281 p.

English, S., Wilkinson, C., Baker, V. (1994) Survey manual for tropical marine resources ASEAN Australian Marine Science Project. Australian Institute of Marine Science, Townsville.

Enríquez, S., Marbà, N., Cebrián, J., Duarte, C. M. (2004) Annual variation in leaf photosynthesis and leaf nutrient content of four Mediterranean seagrasses. *Botanica Marina* **47**, 295-306.

EPA (1998) Changes in seagrass coverage and links to water quality of the Adelaide metropolitan coastline. Environment Protection Agency, Adelaide.

Fitzpatrick, J., Kirkman, H. (1995) Effects of prolonged shading stress on growth and survival of seagrass *Posidonia australis* in Jervis Bay, New South Wales, Australia. *Marine Ecology Progress Series* **127**, 279-289.

Fonseca, M. S., Fisher, J. S. (1986) A comparison of canopy friction and sediment movement between four species of seagrass with reference to their ecology and restoration, *Marine Ecology Progress Series* **29**, 15-22.

Fonseca, M. S., Kenworthy, W. J., Thayer, G. W. (1998). Guidelines for the conservation and restoration of seagrasses in the United States and adjacent waters. NOAA Coastal Ocean Program Decision Analysis Series No. 12. NOAA Coastal Ocean Office, Silver Spring, MD.

Frankovich, T. A., Fourqurean, J. W. (1997) Seagrass epiphyte loads along a nutrient availability gradient, Florida Bay, USA. *Marine Ecology Progress Series* **159**, 37-50.

Gacia, E., Duarte, C. M. (2001) Sediment retention by a Mediterranean *Posidonia oceanica* meadow: The balance between deposition and resuspension. *Estuarine Coastal and Shelf Science* **52**, 505-514.

Guidetti, P., Lorenti, M., Buia, M. C., Mazzella, L. (2002) Temporal dynamics and biomass portioning in three Adriatic seagrass species: *Posidonia oceanica*, *Cymodocea nodosa*, *Zostera marina*. *Marine Ecology* **23**, 51-67.

Hart, D., Clarke, K. (2002) Seagrass change at Beachport, South Australia: 1951 to 1997. Image Data Section, Department for Environment and Heritage, Adelaide. 20 p.

Haynes, D., Müller, J., Carter, S. (2000a) Pesticide and herbicide residues in sediments and seagrass from the Great Barrier Reef World Heritage Area and Queensland's coast. *Marine Pollution Bulletin* **41**, 279-281.

Haynes, D., Ralph, P., Prange, J., Dennison, B. (2000b) The impact of the herbicide Diuron on photosynthesis in three species of tropical seagrass. *Marine Pollution Bulletin* **41**, 288-293.

Hillman, K., Walker, D. I., Larkum, A. W. D., McComb, A. J. (1989) Productivity and nutrient limitation. In: Larkum, A. W. D., McComb, A. J., Shepherd, S. A. (eds.). *Biology of seagrasses. A treatise of the biology of seagrasses with special reference to the Australian region*. Elsevier, Amsterdam. Pp. 635-685.

Hillman, K., Lukatelich, R. J., Bastyan, G., McComb, A. J. (1990) Distribution and biomass of seagrasses and algae, and nutrient pools in water, sediments and plants in Princess Royal Harbour and Oyster Harbour. Technical Series No. 40, Environmental Protection Authority, Western Australia.

Hillman, K., Lukatelich, R. J., Bastyan, G., McComb, A. J. (1991) Water quality and seagrass biomass, productivity and epiphyte load in Princess Royal Harbour, Oyster Harbour and King George Sound. Environmental Protection Authority. Technical Series No. 39.

Holmes, J. W., Waterhouse, J. D. (1983) Hydrology. In: Tyler, M. J., Twidale, C. R.,, Ling, J. K., Holmes, J. W. (eds.). *Natural History of the South East*. Royal Society of South Australia Inc., Adelaide. Pp 49-60.

Howard, F. (1871) Nautical description of Rivoli Bay. No. 23.

Jensen, A., Hoey, P., Kopli, P., Shepherd, M., Till, M., Weinert, M. (1980) The effect of drainage on groundwater behaviour in counties Cardwell and Buckingham and the effect on the Coorong. Report for the Minister of Water Resources 1981-1983. South Australian Department for the Environment.

Keough, M. J., Jenkins, G. P. (2000). Seagrass meadows and their inhabitants. In: Underwood, A. J., Chapman, M. G. (eds.) *Coastal Marine Ecology of Temperate Australia*. University of New South Wales, Australia. Pp. 221-239.

Kirkman, H., Kuo, J. (1990) Pattern and process in southern Western Australian seagrasses. *Aquatic Botany* **37**, 367-382.

Larkum, A. W. D., West, R. J. (1990) Long term changes of seagrass meadows in Botany Bay, Australia. Aquatic Botany 37, 55-70.

Lewis, R. K. (1981) Seasonal upwelling along the south-eastern coastline of South Australia. *Australian Journal of Marine and Freshwater Research* **32**, 843 – 854.

Longstaff, B. J., Dennison, W. C. (1999) Seagrass survival during pulsed turbidity events: the effects of light deprivation on the seagrasses *Halodule pinifolia* and *Halophila ovalis*. *Aquatic Botany* **65**, 105-121.

Macinnis-Ng, C. M. O., Ralph, P. J. (2003a) Short-term response and recovery of *Zostera capricorni* photosynthesis after herbicide exposure. *Aquatic Botany* **76**, 1-15.

Macinnis-Ng, C. M. O., Ralph, P. J. (2003b) *In situ* impact of petrochemicals on the photosynthesis of the seagrass *Zostera capricorni*. *Marine Pollution Bulletin* **46**, 1395-1407.

Macinnis-Ng, C. M. O., Ralph, P. J. (2004) *In situ* impact of multiple pulses of metal herbicide on the seagrass, *Zostera capricorni. Journal of Experimental Marine Biology and Ecology* **302**, 63-83.

Marbà, N., Walker, D. I. (1999) Growth, flowering, and population dynamics of temperate Western Australian seagrasses. *Marine Ecology Progress Series* **184**, 105-118.

Meehan, A. J., West, R. J. (2002) Experimental planting of *Posidonia australis* seagrass in Port Hacking, Australia, to assess the feasibility of restoration. *Marine Pollution Bulletin* **44**, 25-31.

Meehan, A. J., West, R. J. (2004) Seedling development and patch formation of the seagrass *Posidonia australis* in a southwest Australian estuary. *Aquatic Botany* **79**, 1-14.

Miller, D. J., Cheshire, A. C., Hall, S., Havenhand, J. (1998) Assessing the status of temperate reefs in Gulf St Vincent III. Evaluation and description of methodologies. Report to the Environment Protection Agency of South Australia. University of Adelaide, Adelaide.

Moore, K. A., Wetzel, R. L. (2000) Seasonal variations in eelgrass (*Zostera marina* L.) responses to nutrient enrichment and reduced light availability in experimental ecosystems. *Journal of Experimental Marine Biology and Ecology* **244**, 1-28.

Muir, B. G. (1977) Biological survey of the Western Australian wheatbelt. Part 2: Vegetation and Habitat of Bendering Reserve. Records of the Western Australian Museum. Supplement No. 3.

Neverauskas, V. P. (1987) Monitoring seagrass beds around a sewage sludge outfall in South Australia. *Marine Pollution Bulletin* **18**, 158-164.

Neverauskas, V. P. (1988) Response of a *Posidonia* community to prolonged reduction in light. *Aquatic Botany* **31**, 361-366.

Neverauskas, V. P. (1990) Lake Bonney (SE): Impact on the marine environment. Engineering and Water Supply Department, Adelaide.

Nunes, R. A., Lennon, G. W. (1986) Physical property distributions and seasonal trends in Spencer Gulf, South Australia: an inverse estuary. *Australian Journal of Marine and Freshwater Research* **37**, 39-53.

O'Loughlin, E. (2004) Stress response of *Amphibolis antarctica* and *Posidonia sinuosa* to freshwater inputs. Honours thesis, Department of Environmental Biology, The University of Adelaide. 72 pp.

Paling, E., I., McComb, A. J. (2000) Autumn biomass, below-ground productivity, rhizome growth at bed edge and nitrogen content in seagrasses from Western Australia. *Aquatic Botany* **67**, 207-219.

PAN (2004) Pesticide Alert Network (PAN) Pesticide Database – Chemicals: www.pesticideinfo.org.

Peralta, G., Pérez-Lloréns, J. L., Hernández, I., Vergara, J. J. (2002) Effects of light availability on growth, architecture and nutrient content of seagrass *Zostera noltii* Hornem. *Journal of Experimental Marine Biology and Ecology* **269**, 9-26.

Preen, AR, Lee Long, WJ, Coles, RG (1995) Flood and cyclone related loss, and partial recovery, of more than 1000 km^2 of seagrass in Hervey Bay, Queensland, Australia *Aquatic Botany* **52**, 3-17.

Ralph, P. J. (2000) Herbicide toxicity of *Halophila ovalis* assessed by chlorophyll *a* fluorescence. *Aquatic Botany* **66**, 141-152.

Ralph, P. J., Gademann, R., Dennison, W. C. (1998) *In situ* seagrass photosynthesis measured using a submersible, pulse-amplitude modulated fluorometer. *Marine Biology* **132**, 367-373.

Ralph, P. J., Tomasko, D., Moore, K. A., Seddon, S., McGinnis-Ng, C. M. O. (2006) Human impact on seagrass. In. Larkum, A. W. D., Orth, R., Duarte, C. M (eds.). *Treatise on seagrass biology*. Springer, Netherlands. Pp 567-593.

Roberston, E. (1984). Seagrasses. In: Womersley HBS. *The marine benthic flora of southern Australia Part 1*. Government Printer, South Australia. pp.57-122.

Rollon, R. N., van Steveninck, E. D., van Vierssen, W. (2003) Spatio-temporal variation in sexual reproduction of the tropical seagrass *Enhalus acoroides* (L.f.) Royle in Cape Bolinao, NW Philippines. *Aquatic Botany* **76**, 339-354.

Ruiz, J. M., Pérez, M., Romero, J. (2001) Effects of fish farm loadings on seagrass (*Posidonia oceanica*) distribution, growth and photosynthesis. *Marine Pollition Bulletin* **42**, 749-760.

Ruiz, J. M., Romero, J. (2001) Effects of *in situ* experimental shading on the Mediterranean seagrass *Posidonia oceanica*. *Marine Ecology Progress Series* **215**, 107-120.

Ruiz, J. M., Romero, J. (2003) Effects of disturbance caused by coastal constructions on spatial structure, growth dynamics and photosynthesis of the seagrass *Posidonia oceanica*. *Marine Pollution Bulletin* **46**, 1523-1533.

Sand-Jensen, K. (1977) Effect of epiphytes on eelgrass photosynthesis. *Aquatic Botany* **3**, 55-63.

Scarlett, A., Donkin, P., Fileman, T. W., Evans, S. V., Donkin, M. E. (1999) Risk posed by the antifouling agent Irgarol 1051 to the seagrass, *Zostera marina*. *Aquatic Toxicology* **45**, 159-170.

Schrale, G., Cramer, S. R., Lewis, D. C. (1987) Evaluation of drains and weirs in the South East of South Australia. Department of Agriculture, South Australia, Technical Report No. 122.

Schwarzchild, A. C., MacIntyre, W. G., Moore, K. A., Libelo, L. E. (1994) *Zostera marina* L. growth response to atrazine in root-rhizome and whole plant exposure experiments. *Journal of Experimental Marine Biology and Ecology* **183**, 77-89.

Schwebel, D. A. (1983) Quaternary Dune Systems. In: Tyler, M. J., Twidale, C. R.,, Ling, J. K., Holmes, J. W. (eds.). *Natural History of the South East*. Royal Society of South Australia Inc., Adelaide. Pp 15-24.

SEDB (1980) Environmental impact study on the effect of drainage in the South East of South Australia. SEDB 28/76, 13/79. South Eastern Drainage Board, Adelaide.

SEDB (1987) Digging through time; a display illustrating the history of drainage in the South-East of South Australia. South Eastern Drainage Board.

Seddon, S. (2002) Issues for seagrass rehabilitation along the Adelaide metropolitan coast; an overview. In: Seddon, S., Murray-Jones, S. (eds.) *Proceedings of the seagrass restoration workshop for Gulf St Vincent 15-16 May 2001*. Department for Environment and Heritage and SARDI Aquatic Sciences, Adelaide. ISBN 0 7308 5272 5.

Seddon, S., Cheshire, A. C. (2001) Photosynthetic response of *Amphibolis antarctica* and *Posidonia australis* to temperature desiccation using chlorophyll fluorescence. Marine Ecology Progress Series 220, 119-130.

Seddon, S., Murray-Jones, S. (2002). Proceedings of the seagrass restoration workshop for Gulf St Vincent, 15-16 May 2001. Department for Environment & Heritage and SARDI Aquatic Sciences, Adelaide.

Seddon, S., Miller, D., Fotheringham, D., Burgess, S., McKechnie, J. (2003) Beachport seagrass loss and links with Drain M in the Wattle Range Catchment. Prepared for the Coast Protection Board, Department for Environment and Heritage and the Environment Protection Authority. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. Publication no. RD03/0190.

SEWCDB (2003-2006) South Eastern Water Conservation and Drainage Board Management Plan 2003 - 2006. Department of Water, Land and Biodiversity Conservation, Adelaide.

Shepherd, S. A. (1970) Preliminary report upon degradation of sea grass beds at North Glenelg. Department of Fisheries and Fauna Conservation. Adelaide.

Shepherd, S. A., McComb, A. J., Bulthuis, D. A., Neverauskas, V., Steffensen, D. A., West, R. (1989). Decline of seagrasses. In: Larkum, A. W. D., McComb, A. J., Shepherd, S. A. (eds.).

Biology of seagrasses. A treatise of the biology of seagrasses with special reference to the Australian region. Elsevier, Amsterdam. Pp. 346-393.

Short, F. T., Wyllie-Echeverria, S. (1996) Natural and human-induced disturbances of seagrasses. *Environmental Conservation* **23**, 17-27.

Short, F. T., Burdick, D. M., Kaldy, J. E. (1995) Mesocosm experiments quantify the effects of eutrophication on eelgrass, *Zostera marina*. *Limnology and Oceanography* **40**, 740-749.

Silberstein, K., Chiffings, A. W., McComb, A. J. (1986) The loss of seagrass in Cockburn Sound, Western Australia. III. The effect of epiphytes on productivity of *Posidonia australis* Hook. F. *Aquatic Botany* **24**, 355-371.

SKM (2001) Technical Investigations Report. For the Plan Amendment Report Relating to Marine Aquaculture in Kingston/Cape Jaffa. Sinclair Knight Merz, Adeladie.

Smith, N. M., Walker, D. I. (2002) Canopy structure and pollination biology of the seagrasses *Posidonia australis* and *P. sinuosa* (Posidoneaceae). *Aquatic Botany* **74**, 57-70.

Sprigg, R. C. (1979) Stranded and submerged sea beach systems of Southeast South Australia and the Aeolian Desert Cycle. *Sed. Geol*. **22**, 53-96.

Tomasko, D. A., Lapointe, B. E. (1991) Productivity and biomass of *Thalassia testudinum* as related to water column nutrient availability and epiphyte levels: field observations and experimental studies. *Marine Ecology Progress Series* **75**, 9-17.

Turner, M., Carter, D. (1989) Down the drain. The story of events and personalities associated with 125 years of drainage in the South-East of South Australia. ISBN 0724385037. South Eastern Drainage Board, Adelaide.

Tyerman, S. D., Hatcher, A. I., West, R. J., Larkum, A. W. D. (1984) *Posidonia australis* growing in altered salinities: leaf growth, regulation of turgor and the development of osmotic gradients. *Australian Journal of Plant Physiology* **11**, 35-47.

Udy, J. W., Dennison, W. C. (1997) Physiological responses of seagrasses used to identify anthropogenic nutrient inputs. *Marine and Freshwater Research* **48**, 605-614.

Udy, J. W., Dennison, W. C., Lee Long, W. J., McKenzie, L. J. (1999) Response of seagrass to nutrients in the Great Barrier Reef, Australia. *Marine Ecology Progress Series* **185**, 257-271.

Underwood, A.J. 1997. *Experiments in Ecology; Their logical design and interpretation using analysis of variance.* Cambridge University Press. 504 pp.

van Katwijk, M. M., Vergeer, L. H. T., Schmitz, G. H. W., Roelofs, J. G. M. (1997) Ammonium toxicity in eelgrass *Zostera marina*. *Marine Ecology Progress Series* **157**, 159-173.

Walker, D. I. (1985) Correlations between salinity and growth of the seagrass *Amphibolis antarctica* (Labill.) Sonder and Aschers., in Shark Bay, Western Australia, using a new method for measuring production rate. *Aquatic Botany* **23**, 13-26.

Walker, D. I., McComb, A. J. (1988) Seasonal variation in the production, biomass and nutrient status of *Amphibolis antarctica* (Labill.) Sonder ex Aschers. and *Posidonia australis* Hook. F. in Shark Bay, Western Australia. *Aquatic Botany* **31**, 259-275.

Ward, D. H., Morton, A., Tibbitts, T. L., Douglas, D. C., Carrera-Gonzalez, E. (2003) Long-term change in eelgrass distribution at Bahia San Quintin, Baja California, Mexico, using satellite imagery. *Estuaries* **26**, 1529-1539.

Westphalen, G., Collings, G., Wear, R., Fernandes, M., Bryars, S., Cheshire, A. (2005a) A review of seagrass loss on the Adelaide coastline. ACWS Technical Report No. 2 prepared

for the Adelaide Coastal Waters Study Steering Committee. South Australian Research and Development Institute (Aquatic Sciences). Publication No. RD04/0073, Adelaide.

Westphalen, G., O'Loughlin, E., Collings, G., Tanner, J., Eglinton, Y., Bryars, S. (2005b) Responses to reduced salinities of the meadow forming seagrasses *Amphibolis* and *Posidonia* from the Adelaide metropolitan coast. ACWS Technical Report No. 9 prepared for the Adelaide Coastal Waters Study Steering Committee. South Australian Research and Development Institute (Aquatic Sciences). Publication No. RD01/020814, Adelaide.

Wood, N., Lavery, P. (2000) Monitoring seagrass ecosystem health – the role of perception in defining health and indicators. *Ecosystem Health* **6**, 134-148.

APPENDIX 1. Additional water samples

Mark de Jong from the South Eastern Water Conservation and Drainage Board (SEWCDB) collected additional water samples from Drain M (at the Robe-Beachport Road), Blackford Drain (at Princes Highway), Maria Creek (at the Robe-Kingston Road) and Butchers Gap Drain (Cape Jaffa Road Bridge). These samples were processed for a range of nutrients, herbicides and pesticides and were collected at the time of the 'first flush' from each drain. To establish timing of first flush events a number of characteristics were taken into account. These included rainfall, water level and flow rate in the drains, and catchment wetness. Water samples were collected when assigned 'trigger values' were achieved (Table 1a). Observations made by Mark de Jong (SEWCDB) within each of the drains are provided in Tables 1b to 1e below. Observations and collection details are provided in Table 1f.

Table 1a. A description of the characteristics used to determine timing of the 'first flush' of four drains in the South East.

Characteristic	Description and Trigger Value
Rainfall	Rainfall data from the Millicent, Robe, Cape Jaffa, Naracoorte, Padthaway and Coonawarra weather stations was monitored daily on the Bureau of Meteorology website (BOM). A local landholder also provided rainfall information at Salt Creek. TRIGGER: Site visits to each drain occurred when rainfall exceeded 100 mm.
Drain level	The request for sample collection occurred prior to significant rainfall, and each site was visited and low drain levels were noted. Drain level was monitored regularly over time. TRIGGER: When rising drain levels were noted, drain flow was assessed.
Drain flow	Together with drain level, drain flow was monitored regularly. Nil or low flow was noted on baseline visits, and future flow was compared with initial observations. TRIGGER: If obvious flow within the drains was observed (i.e. waves around structures, debris movement with flow), then catchment wetness was assessed.
Catchment wetness	Visual assessments were made to determine catchment wetness. The intention was to look for freestanding water or puddles in paddocks adjacent the drains. This was considered to indicate the advent of surface water influence on drain flow as opposed to subterranean flow or "spring activity" causing drain flow. TRIGGER: If obvious freestanding water was observed in the adjacent areas, and drain level, drain flow and rainfall had also been assessed and met their trigger values the samples were collected.

Table 1b. Site observations made by Mark de Jong at Drain M and in the adjacent catchment.

Date	Rainfall	Drain level	Drain flow	Catchment
17/6/05	June: 95 mm to date at Millicent (in 7 days).	Very low, obvious exposed sandbars.	No flow, floating algae not moving.	Dry, no freestanding water.
21/6/05	June: 131 mm to date at Millicent (in 10 days), 100-125% of June mean.	Level has risen slightly, small sandbars in channel covered in water.	No noted increase in flow but algae no longer present.	Puddles observed in areas remote from drain, no obvious water in nearby catchment.
24/6/05	~160 mm of rain to date.	No discernable rise.	Flowing.	Local catchment is wet approximately 7 km upstream.
5/7/05	Above average June rainfall but little follow-up.	Slight rise.		Local catchment is wet approximately 7 km upstream.
18/7/05	Moderate recent rainfall.	No discernable additional rise.	Flowing.	Local catchment is wet approximately 7 km upstream and Sutherlands Drain and Symon Drain are flowing.
18/7/05	Samples collected.			

Table 1c. Site observations made by Mark de Jong at Blackford Drain at Princes Highway.

Date	Rainfall	Drain level	Drain flow	Catchment
17/6/05	65 mm to date in June at Salt Creek (in 6 days)	Low, only slightly higher than what it was during summer.	Very slight flow – groundwater is suspected	No puddles, dry. Heavy rain inland of catchment has contributed groundwater flow
21/6/05	50 – 100 mm to date in June. 125 - 150% of June mean (in 10 days).	Slightly higher, but very slow – tidal influence.	Very slight flow – groundwater?	Local catchment is dry. Despite recent heavy rains. Interior catchment has also received heavy rain.
24/6/05	80 – 120 mm to date In June. 150% of June mean.	Low	Slight flow.	Dry Catchment.
28/6/05	80-120mm to date In June. 150% of June mean.	Low	Slight flow.	Dry Catchment
5/7/05	No further rain after above average June rainfall	Low, sandbanks remain exposed.	Slight flow.	Dry Catchment
	Insignificant recent rainfall.	Low, sandbanks remain exposed.	Slight flow.	Dry Catchment
13/7/05	Little follow-up rainfall to date.	Low, sandbanks remain exposed.	Slight flow.	Dry Catchment
21/7/05	Little follow-up rainfall to date.	Low, sandbanks remain exposed.	Slight flow.	Dry Catchment
28/7/05	Some moderate recent rainfall.	Low, sandbanks remain exposed.	Slight flow.	Dry Catchment, including upstream catchment
July	Generally ~30% below average rainfall for July across the region		Only very slight flow observed in July, thought to be primarily groundwater driven. Upstream landholders advised of dry conditions.	Catchment is wetting up, some puddles visible in areas south of drain
8/8/05	24 mm of rainfall over the 3rd and 4th, and a further 28 mm fell from the 7th-10th.	Higher, sandbanks covered.	Visible flow under bridge.	Catchment is wetting up, some puddles visible in areas south of drain
11/8/05	Samples collected			

Table 1d. Site observations made by Mark de Jong at the Butchers Gap Drain and in the adjacent catchment.

Date	Rainfall	Drain level	Drain flow	Catchment
21/6/05	50-100 mm to date in June. 125-150% of June mean.	No water. Only 1 small shallow puddle upstream of bridge.	No flow.	Catchment is dry, no puddles in catchment.
24/06/05	~130 mm of rain received to date in June.	No water.	No flow.	Catchment is dry.
5/7/05	Above average June rainfall but little follow-up.	Not assessed.	Not assessed.	Catchment is dry.
22/7/05	Insignificant recent rainfall.	Small puddles exist in drain upstream and downstream of bridge.	No flow.	Catchment is wetting up, some puddles visible in areas south of drain.
July	Generally ~30% below average rainfall for July across the region.		No flow was observed in July.	Catchment is wetting up, some puddles visible in areas south of drain.
8/8/05	24 mm of rain fell over the 3rd and 4th, and a further 28 mm fell from the 7th-10th.	First water visible.	Subtle but visible flow.	Catchment is wetter with puddles visible in catchment
11/8/05	Samples collected.			

Table 1e. Site observations made by Mark de Jong at Maria Creek.

Date	Rainfall	Drain level	Drain flow	Catchment
17/6/05	June-65mm to date at Salt Creek (in 6 days)	Low, only slightly higher than what it was during summer.	No flow.	No puddles, dry. Small drain from airport land into Maria Creek is dry.
21/6/05	50-100mm to date In June125-150% of June mean.	High, as was lake downstream. High tide responsible (often happens).	No flow.	Catchment is dry, no puddles in catchment. Aiport Drain remains dry.
24/6/05	50-100mm to date In June125-150% of June mean.	Low.	No flow, algae remain.	Dry Catchment.
28/6/05	50-100mm to date In June125-150% of June mean.	Low.	No flow.	Dry Catchment and Airport Drain. Catchment at Salt Wells Rd (Kingston Main Drain head waters) is dry.
5/7/05	Above average June rainfall, but little follow-up rainfall to date.	High, as was lake downstream. High tide responsible.	No Flow.	Dry Catchment and Airport Drain.
13/7/05	Little follow-up rainfall to date.	Low.	No flow, algae remain.	Dry Catchment and Airport Drain.
21/7/05	Little follow-up rainfall to date.	High, as was lake downstream. High tide responsible.	No flow.	Dry Catchment and Airport Drain. Catchment at Salt Wells Rd (Kingston Main Drain head waters) is showing signs of wetting up.
28/7/05	Some moderate recent rainfall.	Low.	No flow, algae remain.	Dry Catchment and Airport Drain.
July	Generally ~30% below average rainfall for July across the region.		No surface water flow observed.	Catchment is wetting up, some puddles visible in areas south of drain
8/8/05	24mm of rain fell over the 3rd and 4th, and a further 28mm fell from the 7th-10th.	Water level has risen visibly – not caused by tide.	Visible flow, algae flushed from drain.	Airport drain remains dry but Maria Creek and Kingston Main Drain head waters wet.
11/8/05	Samples collected.			

Table 1f. Observations and details recorded at the time of water sample collections. Recorded by Mark de Jong (SEWCDB).

Site	Observations at the time of collection
Drain M	Samples collected 1445 hrs 18/7/2005. It is estimated that the sample collection was approximately 14 days too late catch the first flush. It should be noted that only the catchment within a short distance of the collection point was wet. The remaining catchment beyond the first 7 km remained dry, and it is considered that the majority of flow in the drain was caused by groundwater, but with significant contributions from the Sutherlands Drain and Symon Main Drain.
Butchers Gap Drain	Samples collected at 1015 hrs 11/8/05 at the Cape Jaffa Road Bridge. There was a reasonably strong flow which was expected after the recent rains, and depth was approximately 650 mm, EC was 2.13mS/cm at 7.9^{o}C (temp corrected). It is considered that the sample collection was undertaken in the first flush, as flow was first noted only a few days before collection. All samples were kept on ice for transport and the nutrient samples were frozen. The glass jars for the herbicide/pesticide samples were placed in the fridge. Unfortunately the samples froze in the fridge (over the weekend) and cracked, losing the samples.
Maria Creek	Samples collected at 1100 hrs 11/8/05 at the East Terrace (Robe-Kingston Road) Bridge. There was a moderate flow after the recent rains, and depth was ~350 mm, EC was 5.53 mS/cm at 9.1^{o}C (temp corrected). It is considered that the sample collection was undertaken in the first flush, as flow was first noted only a few days before collection, and upstream catchment areas were wet for the first time in the season after recent heavy rain. All samples were kept on ice for transport and the nutrient samples were frozen. The glass jars for the herbicide/pesticide samples were placed in the fridge.
Blackford Drain	Samples collected at 1130 hrs 11/8/05 at the Princes Highway Road Bridge. There was a reasonably strong flow which was expected after the recent rains, but depth remained shallow over the bridge foundation, EC was 8.27mS/cm at 10.1^{o}C (temp corrected). It is considered that the sample collection was undertaken in the first flush, as flow was first noted only a few days before collection. All samples were kept on ice for transport and the nutrient samples were frozen. The glass jars for the herbicide/pesticide samples were placed in the fridge. Unfortunately one froze in the fridge (over the weekend) and cracked, losing the sample.

APPENDIX 2. Video Transects

Blackford Drain

A total of 14 video transects were recorded over seagrass habitats offshore from the Blackford Drain (Figure 14). Nine of these were run perpendicular to the shoreline, with one transect run directly offshore of the drain outlet. Another two were carried out either side of the drain at 50m, 150m, 400m and 900m. Four other transects intersected the nine transects in order to cross-check features. These consisted of two radial from the discharge and two parallel to the shore (Table 2a). A further transect was run separately from the core study transects to investigate features that were showing up as bare patches further offshore on aerial photography.

Maria Creek Drain

A total of 12 video transects were recorded over seagrass beds offshore of the Maria Creek Drain, immediately north of Kingston (Figure 15). Transects were run perpendicular to the shoreline. As with the Blackford Drain transects, one was placed directly offshore and then at 50m, 150m, 400m and 900m approximately either side of the drain. The distances were varied from the standard model to fit the two existing profiles in the area to make better use of existing historical data. Three other transects intersected the nine transects in order to cross-check features. These consisted of two radial from the discharge and one parallel to the shore (Table 2b).

Butchers Gap

Nine transects were conducted perpendicular to the shoreline, with three transects intersecting the other nine for cross checking. As with the Blackford Drain transects, one was placed directly offshore and then at 50m, 150m, 400m and 900m approximately either side of the drain. The coastline is slightly curved in this area and necessitated that the transect lines were not parallel. Three other transects intersected the nine transects in order to cross-check features these ran parallel to the shore. The closest to the shore was bent to suit the coastline and was therefore regarded as two transects (Table 2c).

Beachport

Eleven transects were run offshore of Beachport at the Lake George Drain. As with the Blackford Drain transects, one was placed directly offshore and then at 50m, 150m, 400m and 900m approximately either side of the drain. The coastline is concave and necessitated that the transect lines were not parallel. As with Maria Creek Drain the existing profile lines were incorporated into the design to make use of historical data. Two other transects intersected the nine transects in order to cross-check features. These consisted of two radial from the discharge (Table 2d). Extremely poor visibility at Beachport made analysing the videos difficult and as such, the results for this site are not provided. Nonetheless, habitat

mapping of northern Rivoli Bay was undertaken by Seddon *et al.* (2003) and can be found in their report.

Table 2a. The location of video transects in the area surrounding the Blackford Drain in the South East of South Australia.

Site	Line	Start E	Start N	End E	End N
Offshore Sand	715010	396941.988	5928007.704	396297.634	5929526.686
Blackford 900m Nth	715011	398378.971	5928548.418	397389.046	5928698.497
Blackford 400m Nth	715012	398304.024	5928054.067	397314.099	5928204.146
Blackford 150m Nth	715013	398266.551	5927806.891	397276.626	5927956.971
Blackford 50m Nth	715014	398251.561	5927708.021	397261.636	5927858.100
Blackford Drain C/L	715015	398244.067	5927658.586	397254.142	5927808.665
Blackford 50m Sth	715016	398236.572	5927609.151	397246.647	5927759.230
Blackford 150m Sth	715017	398221.583	5927510.281	397231.658	5927660.360
Blackford 400m Sth	715018	398184.109	5927263.105	397194.184	5927413.184
Blackford 900m Sth	715019	398109.162	5926768.754	397119.237	5926918.833
SW QQ Line	715020	398251.561	5927708.021	397119.237	5926918.833
NW QQ Line	715021	398236.572	5927609.151	397389.046	5928698.497
Offshore Line	715022	397168.672	5926911.339	397438.481	5928691.002
Inshore Line	715023	397626.011	5926842.003	397895.820	5928621.667

Table 2b. The location of video transects in the area surrounding Maria Creek in the South East of South Australia.

Site	Line	Start E	Start N	End E	End N
Maria Crk 900m Nth	715024	397774.694	5925258.409	396778.577	5925346.444
Maria Crk 400m Nth	715025	397739.480	5924859.962	396743.363	5924947.997
Maria Crk 150m Nth	715003	397717.471	5924610.933	396671.548	5924703.370
Maria Crk 50m Nth	715026	397662.582	5924402.471	396664.370	5924570.192
Maria Crk	715027	397654.297	5924353.162	396656.085	5924520.883
Maria Crk 50m Sth	715028	397646.012	5924303.853	396647.800	5924471.574
Maria Crk 150m Sth	715004	397627.214	5924171.493	396609.067	5924428.158
Maria Crk 400m Sth	715029	397566.103	5923929.077	396547.956	5924185.742
Maria Crk 900m Sth	715030	397443.882	5923444.245	396425.735	5923850.910
SW QQ Line	715031	397654.297	5924353.162	396425.735	5923850.910
NW QQ Line	715032	397654.297	5924353.162	396778.577	5925346.444
Offshore Line	715033	396518.601	5923813.818	396878.189	5925337.641
Inshore Line	715034	397030.927	5923609.186	397406.392	5925290.959

Table 2c. The location of video transects in the area surrounding Butchers Gap in the South East of South Australia.

Site	Line	Start E	Start N	End E	End N
Butchers 900m Nth	715035	394104.092	5918368.142	393476.343	5919371.406
Butchers 400m Nth	715036	393680.228	5918102.927	393052.479	5919106.191
Butchers 150m Nth	715037	393468.296	5917970.320	392840.546	5918973.584
Butchers 50m Nth	715038	393383.523	5917917.277	392755.774	5918920.541
Butchers Gap Drain	715039	393341.137	5917890.755	392713.387	5918894.019
Butchers 50m Sth	715040	393298.750	5917864.234	392671.001	5918867.498
Butchers 150m Sth	715041	393213.977	5917811.191	392586.228	5918814.455
Butchers 400m Sth	715042	393055.417	5917674.534	392305.004	5918562.314
Butchers 900m Sth	715043	392811.552	5917369.091	391955.374	5918183.273
SW QQ Line	715044	393383.523	5917917.277	391955.374	5918183.273
NW QQ Line	715045	393298.750	5917864.234	393476.343	5919371.406
Offshore Line	715046	391992.004	5918148.440	393503.009	5919328.789
Inshore Line Sth	715047	392722.942	5917453.354	393257.168	5918024.954
Inshore Line Nth	715048	393257.168	5918024.954	393995.610	5918541.518

Table 2d. The location of video transects in the area surrounding Drain M in the South East of South Australia.

Site	Line	Start E	Start N	End E	End N
900m Offset South	710002	412855.540	5850598.226	413455.044	5850622.625
400m Offset South	710007	412981.699	5851405.940	413816.698	5850855.689
150m Offset South	710004	413114.565	5851601.073	413735.657	5851096.865
50m Offset South	710008	413252.909	5851760.985	413847.839	5850957.272
C/L Drain	710009	413277.340	5851812.020	413872.300	5851008.264
50m Offset East	710010	413317.738	5851841.484	413912.700	5851037.730
150m Offset East	710011	413398.845	5851899.992	413993.808	5851096.239
400m Offset East	710012	413858.647	5852035.496	414121.605	5851070.688
900m Offset East	710013	414332.321	5852137.660	414378.903	5851138.746
East QQ Line	710014	413358.060	5851702.973	414358.511	5851576.048
South QQ Line	710015	413358.060	5851702.973	413299.125	5850616.285
Adj to Railway Tce	710003	412846.248	5851176.561	413613.398	5850535.093
Adj to Boat Ramp	710005	412849.386	5850978.981	413680.030	5850422.083

APPENDIX 3. Habitat Maps

Figure 3a. Habitat types along video transects adjacent to the Blackford Drain Outlet, Lacepede Bay.

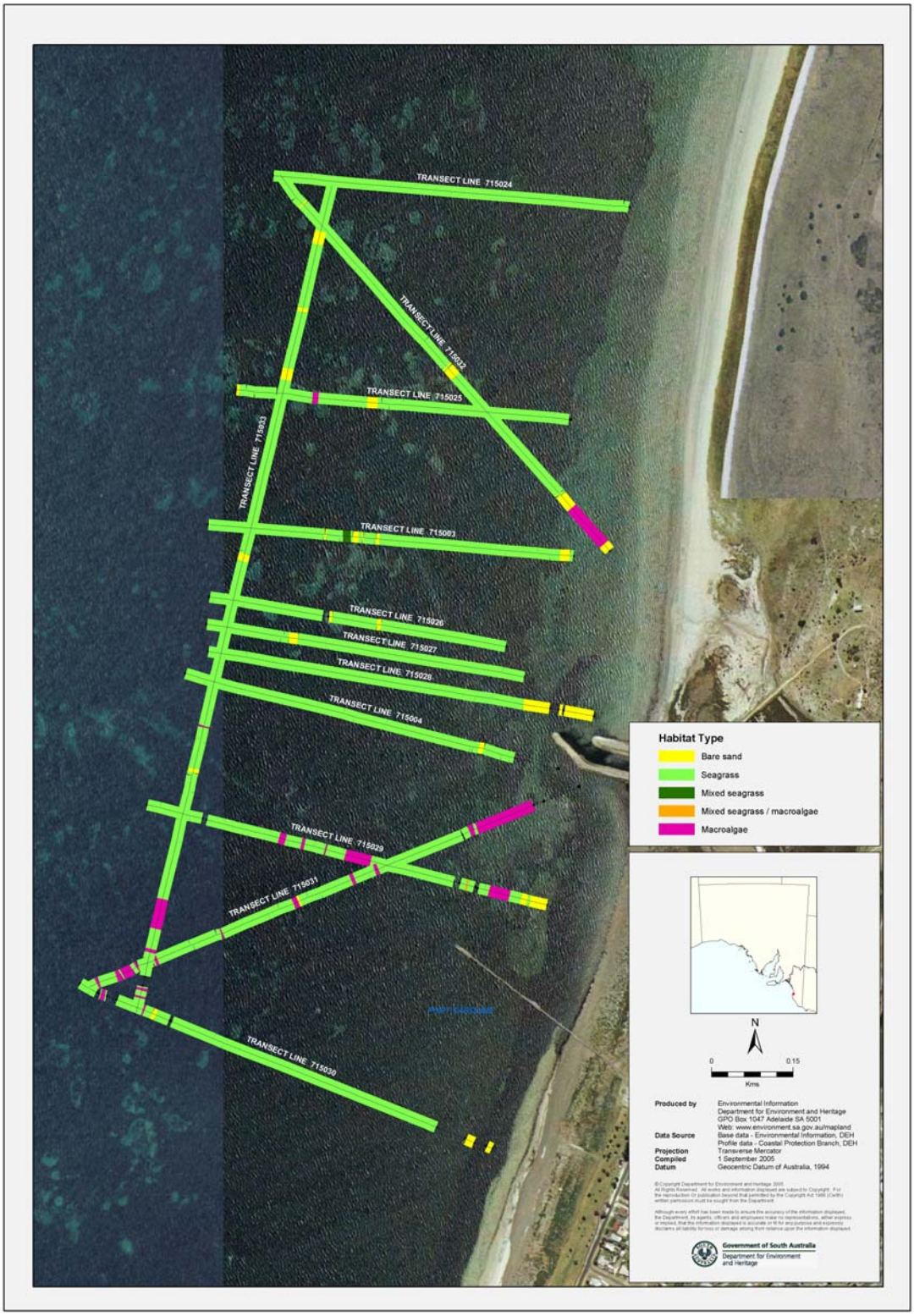

Figure 3b. Habitat types along video transects adjacent to the Maria Creek Outlet, Lacepede Bay.

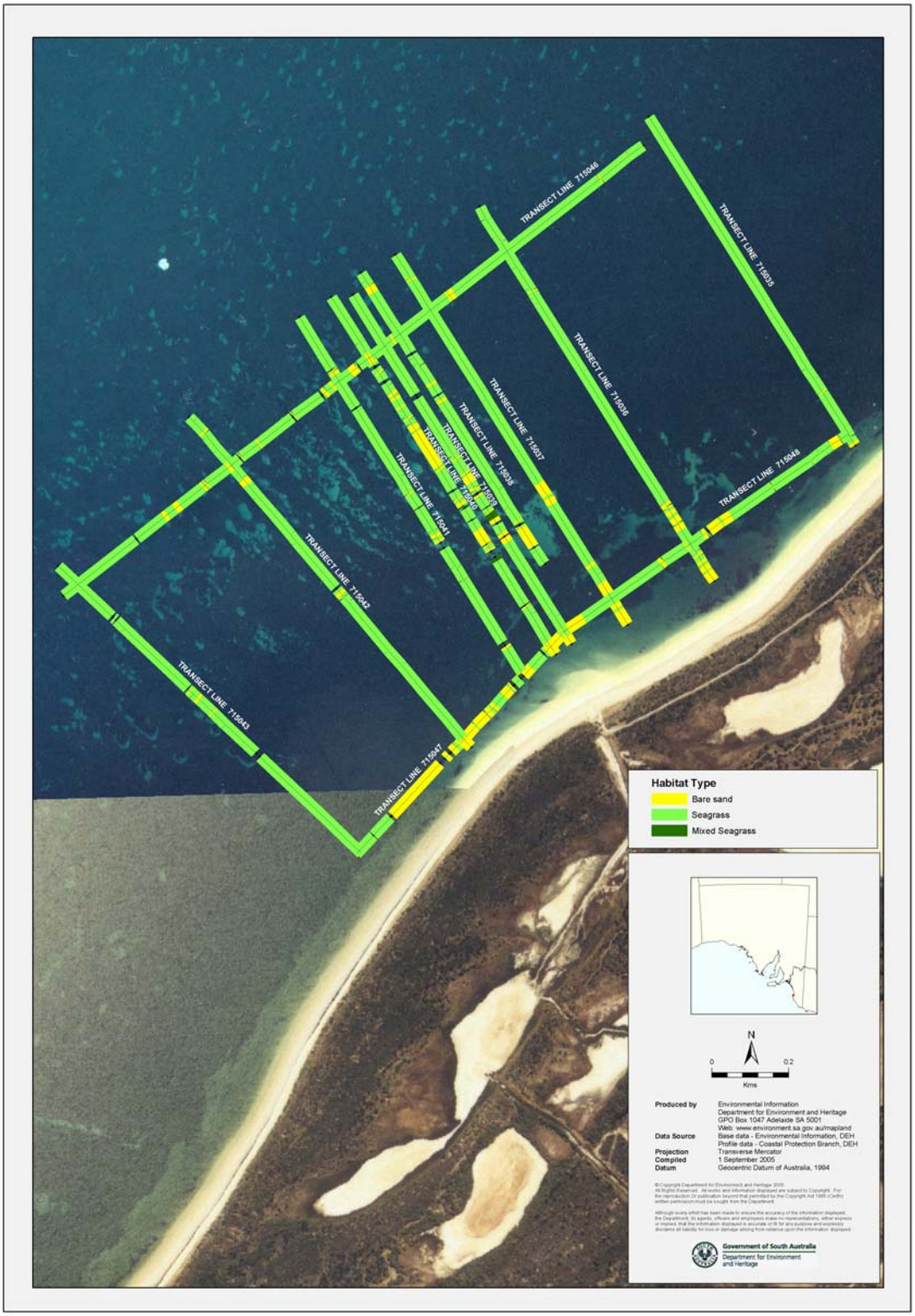

Figure 3c. Habitat types along video transects adjacent to the Butchers Gap Drain Outlet, Lacepede Bay.