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No detrimental effects of desalination waste on temperate fish assemblages

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Water resources are becoming increasingly scarce due to population growth and global changes in weather patterns. Desalination plants that extract freshwater from brackish or seawater are already being used worldwide, with many new plants being developed and built. The waste product from the extraction processes has an elevated salt concentration and can potentially cause substantial impacts to local marine flora and fauna. The present study assesses the impact of saline waste from a 100 GL/year desalination plant on southern Australian temperate fish assemblages, using baited remote underwater video. The study compared four reference sites to the impact site (desalination outfall) and found no evidence that the saline waste was having a detrimental effect on fish assemblages in proximity to the outfall, with species diversity and abundance comparable to those observed at reference sites. However, species diversity and abundance varied across geographical location, protection from fishing pressure, and reef type. Our study is one of the few assessing the ecological impacts of saline waste discharged from a large desalination plant and shows no decrease in fish diversity or abundance, which is the response typically associated with the negative impacts of anthropogenic activities on fish assemblages.

Keywords: baited video, BRUVS, desalination, fish communities, impact assessment, marine protected areas

Introduction

Coastal desalination plants are increasingly being used as a solution to mitigate freshwater shortages (Roberts *et al.*, 2010). Desalination plants exist in many countries but are particularly prevalent in the Middle East and continue to be built in Europe, United States, and Australia to combat water shortages and ensure water security for the future (Roberts *et al.*, 2010). The majority of desalination plants extract freshwater from marine or brackish water using either reverse osmosis (RO) membranes or thermal distillation. The concentrated saline waste that results from these processes is then typically released back into the environment (El Saliby *et al.*, 2009). Previous studies examining the effects of this saline waste on marine life have found decreases in echinoderm, coral, plankton, and fish abundances, along with decreases in infaunal and sessile invertebrate assemblages (Roberts *et al.*, 2010; Clark *et al.*, 2018). These changes are considered to be a result of a variety of factors including increased salinity, temperature, and the presence of chemical compounds associated with the treatment process. There is, however, some contention about the validity of some results due to a lack of details in the methods and/or experimental design rigour (Roberts *et al.*, 2010). Advances in technology have, however, reduced the risks of potential impacts to local flora and fauna from the saline waste, using diffusers at the outfall, which rapidly dilute the waste stream to ambient concentrations (Clark *et al.*, 2018). Past issues with elevated water temperatures around the outfall and the use of toxic chemicals in treatment process have largely been overcome with predominance of RO membrane technology in the desalination process (Lattemann and Höpner, 2008, El Saliby *et al.*, 2009; Roberts *et al.*, 2010). A recent study in southern Australia has found significant increases in fish abundance in the outlet area of a recently constructed desalination plant when the plant was discharging compared to before the plant was built and when the plant was not discharging (Kelaher *et al.*, 2020). Such studies highlight that well-designed and well-situated RO desalination plants may only cause localised changes to marine ecosystems, some of which may be considered positive for some user groups (e.g. fishers).

South Australia has had an operational 100 GL/year capacity desalination plant on Adelaide's metropolitan coast since 2011. The Adelaide Desalination Plant (ADP) uses 24 "duck-billed" diffusers to disperse the saline waste from the treatment process into the local marine environment. A large seawater intake (10 m high × 10 m wide) is also located ∼300 m from the ADP outfall at a water depth of 20 m, with the combined underwater infrastructure acting as an artificial reef in an area in which the substrate is predominately sand (SA Water, 2008). An exclusion zone (no access) exists to protect the ADP infrastructure from potential anchor damage from boats mooring in the region of the intake or outfall structures. This exclusion zone provides local fish assemblages associated with the artificial reef protection from recreational fishing pressures.

This study aims to assess the impact of a saline waste discharge, from a large desalination plant, on local fish assemblages. The study compares fish assemblages living on artificial and natural reefs, protected and unprotected from fishing pressure, and season and annual changes. The hypothesis proposed is that the saline waste from a desalination plant has a greater impact on the diversity and abundance of local fish assemblage than seasonal changes, structural changes, or fishing pressures.

Methods

Sampling sites

Regulatory requirements (EPA, 2020) for the ADP stipulate that there must be ongoing environmental assessments of the ecosystem surrounding the outfall. Fish assemblages associated with the outfall infrastructure are examined every 3 years as part of the monitoring requirement. Previous assessments of fish assemblages within the ADP area showed minimal changes during construction of the desalination plant (Colella *et al.*, 2010), but questions were raised about the selection of reference sites (no impact) when making comparisons with the impact site (ADP outfall), in terms of artificial reef vs. natural communities (Barbara, 2016). As a result, a new sampling design was used for the two most recent assessments in 2015 and 2018 to include more comparable reference sites. Our study examined the fish assemblages on five reefs (four reference sites and the ADP intake/outfall) located in Gulf St Vincent, South Australia (Figure 1). Specifically, we compared the outfall area to four reference areas with two of these areas having artificial reef substrate and two being natural reefs. One of the selected natural reef sites is also protected in a no-take zone (referred to as "protected" hereafter) to compare against the protected outfall area. While it would have been preferable to compare fish assemblages around the ADP outflow to another protected site with an artificial structure, at similar size, and depth, this could not be found in Gulf St Vincent.

The ADP outfall (impact site) comprised of six risers (3 m high × 2 m wide) distributed over 180 m between 17 and 20 m. The risers are protected by large limestone rocks (0.3–0.6 m diameter), with each riser covering ∼16 m^2 and surrounded by sandy substrate and sparse seagrass patches (*Zostera nigricaulis*). Since 2009, the site has been protected by a defined exclusion zone (1.2 km^2; South Australian Government, 2009) prohibiting recreational vessels from entering or anchoring. The ADP has been operational and has been discharging saline waste to the marine environment, with a total volume of 235 GL since 2011. The volume of saline waste discharged during this study varied between 661 and 1287 ML/month at an approximate salinity concentration of 67 g/L (EPA, 2020). Salinity concentrations are recorded every 15 min on the seafloor, 100 m north and south of the outfall, with average salinity values between 36 and 37.5 g/L, when the plant is operational (EPA, 2020). The saline waste plume can be detected up to 400 m away from the outfall (Ayala *et al.*, 2015).

The impact site is located within 60 m of the ADP outfall. The four reference sites are located north and south, at a minimum distance of 6 km from the ADP outfall. The reefs are characterised by substrate (artificial/natural), protection status referring to whether the area is fished or protected by an exclusion zone (e.g. marine reserve), and geographical location (north/south) in relation to the ADP outfall (Table 1). Port Noarlunga Reef (PNR) is a high-profile natural limestone reef that has been a marine reserve since 1971. The reef is situated parallel to the coastline at a depth between 0 and 12 m and is a popular area for recreational activities such as swimming, snorkelling, and diving. Noarlunga Tyre Reef (NTR) is an artificial reef constructed from car tyres and concrete into pyramid-shaped structures, at a depth of 15–18 m. Two wrecks are located alongside these tyre structures, with the surrounding substrate composed of soft sediment. Seacliff Reef (SR) is a lower-profile natural reef, situated in ∼12 m surrounded by seagrass, macroalgae, and soft-sediment habitats. Glenelg Tyre Reef (GTR) is constructed from similar material to the Noarlunga Tyre Reef, at 17–20 m. Noarlunga Tyre Reef, Seacliff Reef, and the Glenelg Tyre Reef are popular recreational fishing sites. Port Noarlunga Reef and Noarlunga Tyre Reef are located south of ADP and Glenelg Tyre Reef and Seacliff Reef are located north.

Baited Remote Underwater Video Stations deployments

Fish assemblages were investigated using Baited Remote Underwater Video Stations (BRUVS). BRUVS are a popular method for assessing fish assemblages (Whitmarsh *et al.*, 2017) and have successfully been used for a wide variety of purposes including assessing marine protected areas, anthropogenic impacts, and spatial variation (Folpp *et al.*, 2013; Kelaher *et al.*, 2014; Whitmarsh *et al.*, 2014; Whitmarsh *et al.*, 2018; Clarke *et al.*, 2019; Whitmarsh *et al.*, 2019). The limitations of this method have previously been documented (Langlois *et al.*, 2010; Harvey *et al.*, 2012; Harvey *et al.*, 2013; Whitmarsh *et al.*, 2017; Whitmarsh *et al.*, 2018) and show that BRUVS is well suited to sample mobile species but may underrepresent small, cryptic

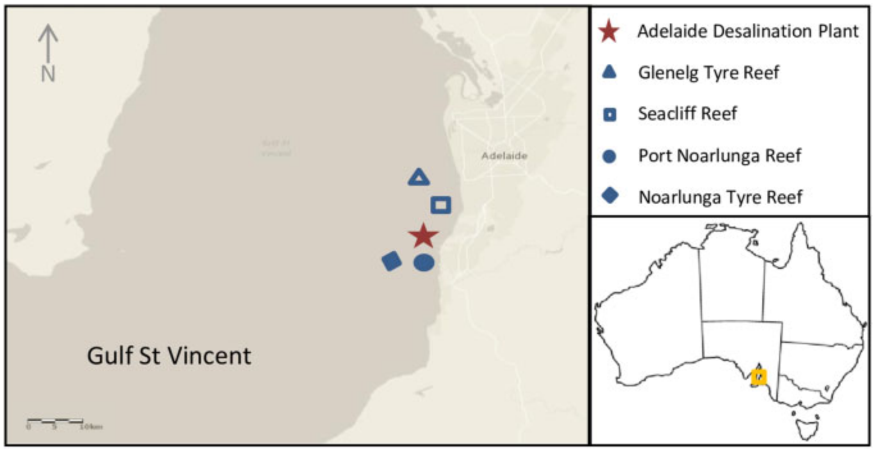

Figure 1. A map of the study sites within Gulf St Vincent, South Australia, showing the Adelaide Desalination Plant (red star) and the reference sites (blue). Northern sites are indicated by unfilled symbols, while southern sites are filled.

Table 1. The survey sites and their attributes.

Site	Acronym	Use	Status	Reef substrate	Location
Adelaide Desalination Pant	ADP	Impact	No-take	Artificial	South
Glenelg Tyre Reef	GTR	Reference	Fished	Artificial	North
Noarlunga Tyre Reef	NTR	Reference	Fished	Artificial	South
Port Noarlunga Reef	PNR	Reference	No-take	Natural	South
Seacliff Reef	SCR	Reference	Fished	Natural	North

species. BRUVS were set up with high-definition GoPro Hero 3+ or 4 Silver video cameras on steel frames. These cameras were selected due to their relative low cost, ability to record in high definition, long battery life, wide-angle viewing, and image quality in low light conditions. Single, horizontal set-ups were used rather than a stereo as fish lengths were not measured in this study. The units were baited with 500 g of minced sardines (*Sardinops sagax*). Six replicate deployments were performed at each site. Units were set to continuous recording and deployed for a minimum of 1 h before retrieval.

Video processing

Videos were analysed using the specialised SeaGIS *EventMeasure* software (SeaGIS Pty Ltd, Bacchus Marsh, VIC, Australia; www.seagis.com.au/event.html). On each replicate, taxa were identified to species where possible (Kuiter, 1996; Gomon *et al.*, 2008) and counted using the relative abundance measure, *MaxN*. *MaxN* is the maximum number of individual fish (for each species or taxon) observed in a single frame throughout the deployment duration. *MaxN* is thus a conservative estimate of abundance, particularly where large fish numbers are present or there is a large turnover of individuals during deployment (Priede *et al.*, 1994; Ellis and DeMartini, 1995; Willis, 2001). Most species were easily recognisable but, if taxa were not able to be reliably identified to species level, then they were grouped into genus or family, e.g. two trevally species could not be differentiated and thus were grouped as *Pseudocaranx* spp.

Data analysis

Statistical analyses were conducted using PRIMER v7 (Clarke and Gorley, 2015) with PERMANOVA+ (Anderson *et al.*, 2008). We analysed 120 replicate deployments across both years, with 60 deployments during each spring and autumn. Univariate analysis assessed differences between year (random, two levels), season (fixed factor, two levels), and site (random factor, five levels) for total abundance and species diversity using a Euclidean distance matrix.

Multivariate data were transformed using dispersion weighting by site to account for the variable schooling nature of some fish species (Clarke *et al.*, 2006) and tested using the above design on a Bray–Curtis resemblance matrix. Pairwise tests were used to further investigate differences between significant factors. Similarity percentage (SIMPER) analyses were used to determine the similarity between groups and which species were driving any observed differences. Distance to centroids for each pair of sites was calculated and then averaged for those comparing ADP to other sites and then for the rest of the site comparisons that did not involve the ADP. Bootstrap averages (run 100 times) were calculated and used to construct a metric multi-dimensional scaling (MDS) ordination plot showing differences among sites. Canonical analysis of principal coordinates (CAP) was also used to test for differences between sites.

CAP was also used to assess the influence of the factors: geographic location (north vs. south), type (artificial vs. natural), and protection (protected vs. unprotected). Allocation success rates were then compared to assess the discriminatory power of each factor.

Results

Seventy-eight species and 10 776 individual animals were observed across the two sampling years, five sites, and two seasons. A broad range of species were observed including 65 teleosts, 5 chondrichthyans, 2 cephalopods, 5 decapod crustaceans, and 1 dolphin, hereafter referred to as 'fish assemblage". There were a similar number of species observed in 2015 (60) and 2018 (70; Pseudo-$F = 0.02$, $p = 0.894$). Significant differences were, however, observed between seasons in 2018 ($t = 4.44$, $p = 0.001$) but not in 2015 ($t = 0.74$, $p = 0.475$). The number of species was also different among sites, with ADP being significantly lower than GTR and NTR, and GTR also being significantly higher than PNR and SCR ($p < 0.026$). All other sites had a similar number of species ($p > 0.05$; Figure 2a). Total abundances were more variable with generally lower abundances in 2018 [mean *MaxN* per replicate ($\pm$ SE) = 60 ± 5] compared to 2015 (117 ± 12; Figure 2b) with significant differences between all factors ($p < 0.08$) except season ($p = 0.344$).

Fish assemblages were significantly different between all factors and their interactions (Table 2). Assemblages were least different between season within year, with four non-significant outcomes including both seasons for each year for Seacliff Reef (Table 3). The desalination site was the only site with consistent fish assemblages across seasons, which only occurred in 2015 (Table 3). Fish assemblages varied extensively among sites except for Seacliff Reef and Glenelg Tyre Reef in autumn 2018 and for ADP and Noarlunga Tyre Reef in autumn 2015 and spring 2018 (Table 4). Dissimilarity values among sites supported the paired comparison and were often higher for spring than autumn, ranging between 69 and 93% (Table 4). The most similar sites to the ADP (i.e. lowest dissimilarity) were the other two artificial reefs, Noarlunga and Glenelg Tyre Reefs (Table 4).

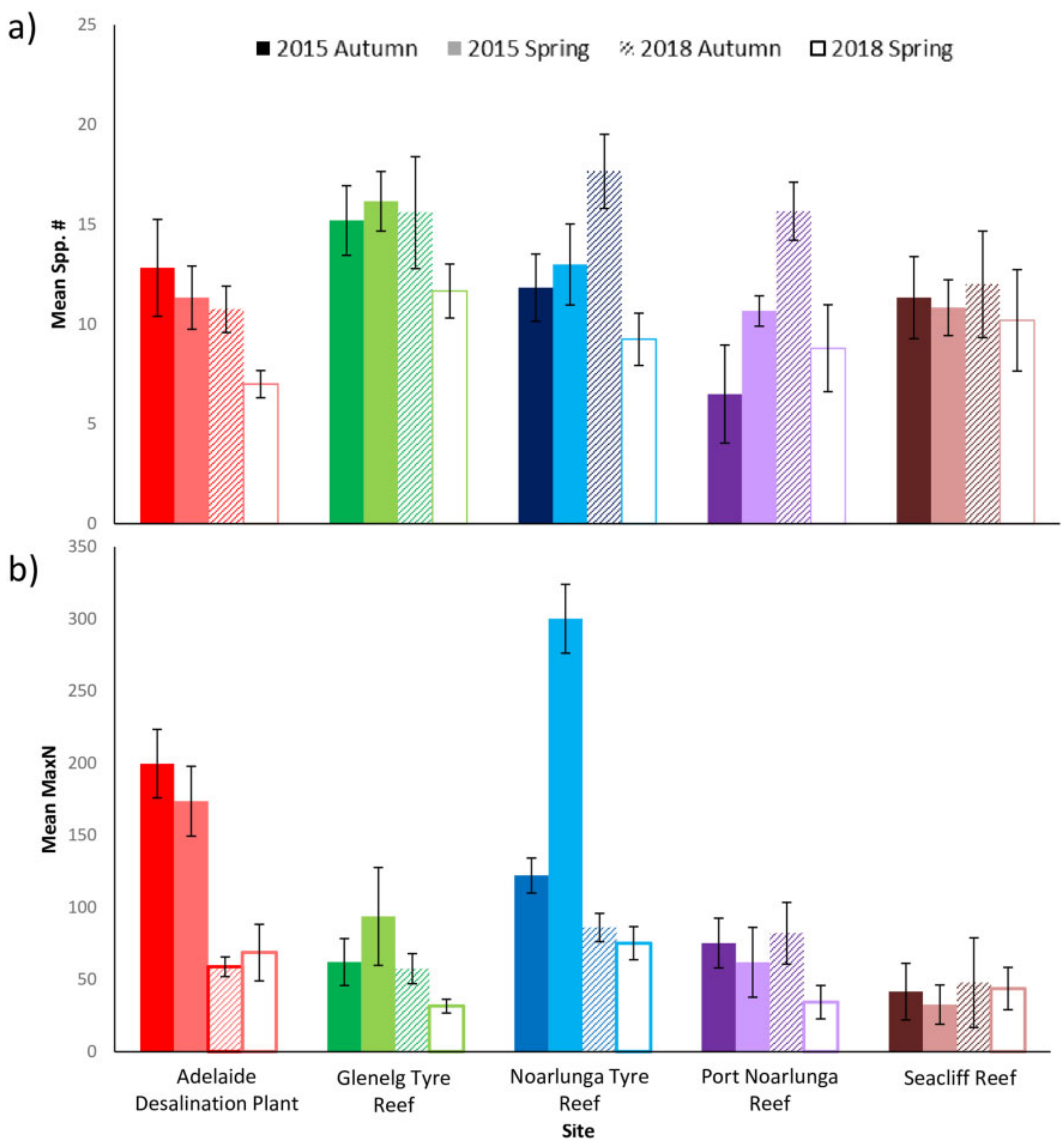

Figure 2. (a) Mean number of species and (b) mean abundance (*MaxN*) $\pm$ SE of total individuals per site for both spring and autumn surveys in each year. $N = 120$.

The average distance to centroids for pairs of sites was slightly lower (i.e. pairs are more similar) for pairs involving the ADP (38.6) compared to pairs not involving the ADP (44.5). This can also be visualised using a bootstrapped averages MDS plot, where the centroid and 95% confidence ellipse for the ADP lies nearby to the other sites (Figure 3). Port Noarlunga Reef (natural substrate) appears to be the most dissimilar site.

Port Noarlunga Reef was also more distinct than other sites using CAP analysis (Figure 4), with the species contributing to similarity within sites falling into three broad groups. Species contributing to the similarity within Port Noarlunga Reef included brownspotted wrasse *Notolabrus parilus*, sea sweep *Scorpis aequipinnis*, and horseshoe leatherjackets *Meuschenia hippocrepis*, while pink snapper *Chyrsophrys auratus*, trevally *Pseudocaranx* spp., and Degen's leatherjackets *Thamnaconus degeni* (Figure 4) contributed to within site similarity at the ADP and Noarlunga Tyre Reef. The species contributing to similarity at Glenelg Tyre Reef and Seacliff Reef were smaller and included blackspotted wrasse *Australabrus maculatus*, rough leatherjackets *Scobinichthys granulatus*, silverbelly *Parequula melbournensis*, and bluespotted goatfish *Upeneichthys vlamingii* (Figure 4).

Geographic location, reef type, and protection status all affected fish assemblages (trace and delta $p = 0.001$; Figure 5). Geographic location was the factor that best explained the differences in fish assemblages as it had the lowest misclassification error of 3.2%, while type of reef had a higher misclassification error of 7.3%, and protection had the highest misclassification error of 18.7%.

Table 2. PERMANOVA results from the main test of the factors year, season, and site for the multivariate assemblage data. MS = Mean squares

Factor	df	MS	Pseudo-*F*	*p*(perm)
Year	1	18 656	3.11	0.054
Season	1	14 966	1.39	0.265
Site	4	21 868	3.63	**0.005**
Year × season	1	9 047	1.52	0.243
Year × site	4	6 018	2.77	**0.001**
Season × site	4	6 013	1.01	0.493
Year × season × site	4	5 961	2.74	**0.001**
Res	103	2 175	–	–

Values in bold indicate significant differences. Unique permutations ranged from 997 to 999 per pairwise test.

Table 3. Pairwise PERMANOVA results from the interaction of year, season, and site showing the comparison between years and seasons for each site.

	2015 vs. 2018				Autumn vs. spring			
	Autumn		Spring		2015		2018	
Sites	*t*	*p*(perm)	*t*	*p*(perm)	*t*	*p*(perm)	*t*	*p*(perm)
ADP	2.18	**0.003**	1.87	**0.003**	1.28	0.126	1.72	**0.004**
GTR	1.38	0.058	2.38	**0.004**	1.82	**0.02**	1.95	**0.005**
NTR	1.90	**0.007**	2.53	**0.004**	2.21	**0.004**	2.38	**0.004**
PNR	2.36	**0.007**	1.71	0.051	2.37	**0.008**	1.87	**0.025**
SCR	1.36	0.073	1.12	0.275	1.43	**0.014**	1.36	**0.034**

Values in bold indicate significant differences. Unique permutations ranged from 126 to 977 per pairwise test.

Discussion

Our study shows that the artificial habitat and protection provided by the ADP outfall were the only factors influencing fish assemblages, while the discharged saline waste did not negatively affect fish communities. Fish assemblages associated with the ADP showed no reduction in species richness or abundance despite the discharge of saline waste. The ADP site also had the strongest similarity to other artificial sites (NTR and GTR), thus rejecting our hypothesis that the desalination plant has a greater impact on fish assemblage than seasonal changes, structural changes, or localised fishing pressures.

At the ADP, fish abundance and diversity was high and comparable to the reference sites, indicating that the desalination discharge has no discernible detrimental effect. The artificial structure of the diffuser is providing additional habitat for fishes which are also protected through the exclusion zone. Both of these aspects have been shown to increase fish abundance and diversity (Wilhelmsson *et al.*, 2006; Lester *et al.*, 2009). Prior to the plant, soft sediment was the dominant substrate, with a small 1.5 km^2 low profile reef inshore, without any high-profile reef structures (SA Water, 2008). The addition of an artificial reef from the infrastructure installed and the diffusers provides shelter, food resources, and niche habitats that likely drove the increase in species diversity and abundance. This resulted in a

Table 4. Pairwise PERMANOVA results from the interaction of year, season, and site showing the comparison between sites for each season and year, along with the dissimilarity values from SIMPER.

	2015						2018					
	Autumn			Spring			Autumn			Spring		
Pairs	*t*	*p*(perm)	Dissim %	*t*	*p*(perm)	Dissim %	*t*	*p*(perm)	Dissim %	*t*	*p*(perm)	Dissim %
ADP vs. GTR	1.95	**0.005**	76	2.45	**0.003**	76	1.80	**0.001**	80	2.55	**0.001**	91
ADP vs. NTR	1.26	0.125	69	2.63	**0.005**	69	1.66	**0.013**	75	1.14	0.234	76
ADP vs. PNR	1.70	**0.022**	84	3.32	**0.002**	89	2.61	**0.002**	86	1.78	**0.008**	88
ADP vs. SCR	2.05	**0.008**	84	2.40	**0.006**	86	1.70	**0.009**	75	1.80	**0.007**	92
GTR vs. NTR	2.04	**0.006**	76	3.09	**0.004**	82	1.56	**0.006**	73	2.88	**0.002**	89
GTR vs. PNR	2.28	**0.004**	91	3.15	**0.003**	91	2.43	**0.003**	83	2.91	**0.004**	93
GTR vs. SCR	1.64	**0.012**	71	1.66	**0.006**	76	1.33	0.101	69	1.61	**0.013**	74
NTR vs. PNR	1.49	**0.032**	78	3.83	**0.002**	91	2.27	**0.001**	77	2.04	**0.005**	86
NTR vs. SCR	1.95	**0.005**	81	2.47	**0.001**	83	2.20	**0.004**	81	2.15	**0.002**	93
PNR vs. SCR	2.05	**0.002**	91	2.59	**0.004**	91	2.75	**0.003**	85	1.88	**0.005**	92

Values in bold indicate significant differences. Unique permutations ranged from 126 to 978 per pairwise test.

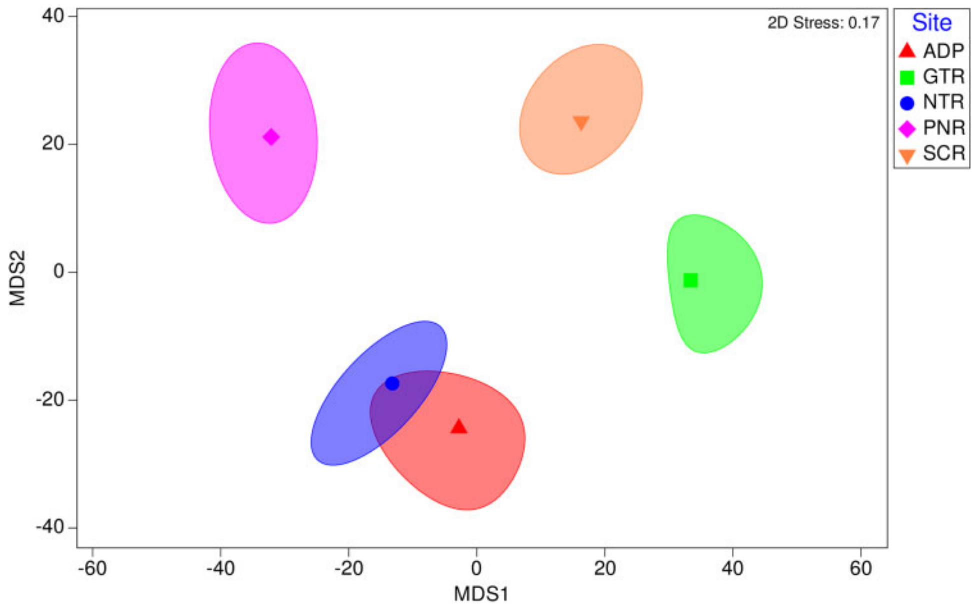

Figure 3. A metric MDS plot calculated using bootstrapped averages and showing the resulting centroid and 95% confidence ellipses for each site ($n = 100$ bootstraps).

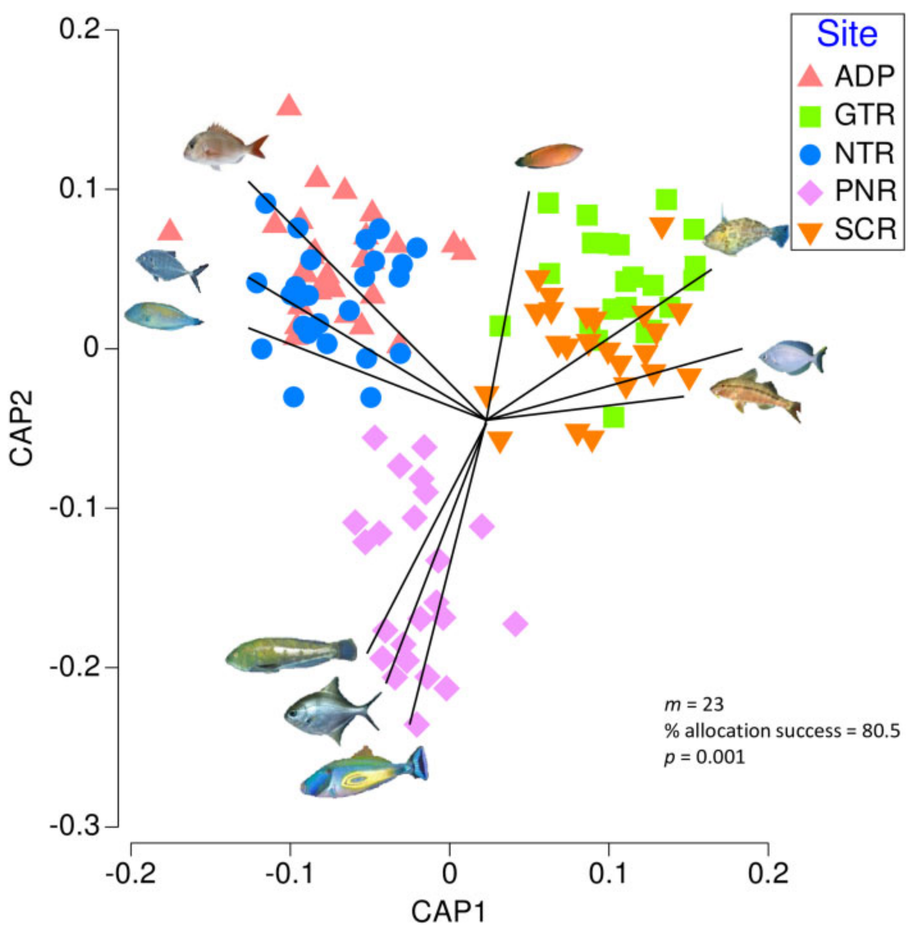

Figure 4. CAP ordination plot displaying the results from SIMPER analysis for site. Fish pictogra represent species in the top 3 contributors to similarity within a site.

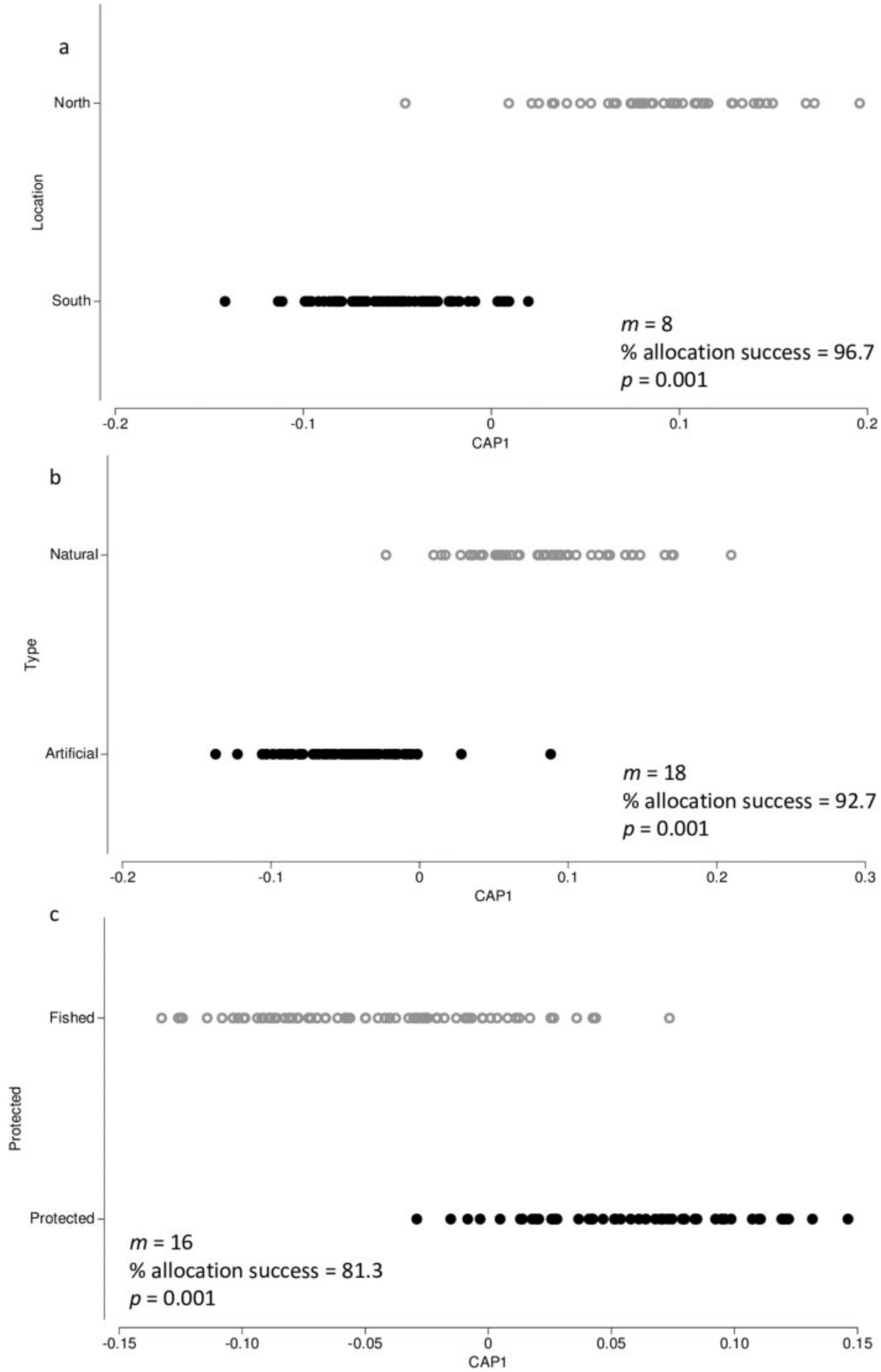

Figure 5. CAP ordination plots showing discriminant factors for the first Principal Coordinate axis (out of *m* axes) for the factor of (a) geographic location, (b) type, and (c) protection.

change in fish community from species typical of soft sediments, e.g. silverbellies *P. melbournensis* and gobies Gobiidae spp. (SA Water, 2008), to common reef species, e.g. *C. auratus* and leather-jackets in the Monacanthidae family (present study). Pre-plant surveys also found low species richness in the areas now occupied by the intake and diffuser, with 6–20 species recorded prior to the plant being built (SA Water, 2008) compared to 40 species in the present study.

The ADP exclusion zone also provides protection to resident fish species (Willis, 2001; McLaren *et al.*, 2015). It is likely to be particularly beneficial to species with small activity spaces and those often targeted by fishers such as *C. auratus*, which was the

3rd most abundant species at the ADP. The ADP exclusion zone is well-enforced through continuous monitoring by security cameras reducing illegal fishing, with studies showing well-enforced protected areas are more likely to lead to higher fish abundance and diversity than those poorly enforced (Edgar *et al.*, 2014; Kelaher *et al.*, 2015). It is also possible that other activities contributed to the differences in fish assemblages, e.g. scuba-diving or vessels anchoring, which differed among sites.

Seasonal variations in fish diversity and abundance occurred at all sites and are common for temperate fish assemblages (Lehodey *et al.*, 2006; Olsson *et al.*, 2012). The limited seasonal variation at the ADP in 2015 is due to consistent abundances of key species such as snapper *C. auratus* and Degen's leatherjacket *Thamnaconus degeni*, which had fluctuating abundances at other sites (e.g. Glenelg, Port Noarlunga Reef, and Noarlunga Tyre Reef). The reasons for these consistent abundances are currently unknown but might be related to favourable environmental conditions and food availability throughout both seasons. The highly enforced nature of the exclusion zone also likely protects resident fish more consistently than at other protected sites with inconsistent levels of enforcement. The artificial reef at the ADP is also relatively young (∼ 3 years old in 2015) and may still show a transitioning community.

Assemblages observed at the study sites appeared to show significant clustering by geographic location, i.e. southern sites (Port Noarlunga Reef, Noarlunga Tyre Reef, and ADP) vs. northern sites (Seacliff Reef and Glenelg Tyre Reef). The major species driving this difference between locations was the Degen's leatherjacket *Thamnaconus degeni*, which was highly abundant at the southern sites and comparatively absent from the northern sites. The reasons for this difference are currently unknown but are unlikely to be due to changes in environmental conditions as the sites are all reasonably close together and experience similar conditions.

Fish assemblages were also affected by the sampled reef type due to differences in reef age, structure, profile, and habitat characteristics (e.g. depth and exposure). Seacliff Reef is a natural site with low-relief reef that had lower fish abundance and diversity than other sites. This is consistent with previous studies showing increased abundance (Wilhelmsson *et al.*, 2006) and diversity (Gratwicke and Speight, 2005) at sites with high-relief reef.

New surfaces of artificial reef available for colonisation can also result in increased diversity of species (Arena *et al.*, 2007; Folpp *et al.*, 2013). Artificial reefs can provide more complex refuges due to increased surfaces available compared to low-relief natural reefs which may be more embedded in soft strata (Perkol-Finkel *et al.*, 2006). Over time artificial reefs may become buried by sediments and their assemblages reflect those of natural reefs in the area (Burt *et al.*, 2011).

Assessment of fish assemblages at multiple locations with desalination effluents would be necessary to determine if the findings from this study can be generalised or are site-specific. The areas around the desalination plant and more broadly in Gulf St Vincent also lack deep, high-profile rocky reef areas and protected artificial reefs to enable the preferred cross-factor experimental design. Changes in sampling design between studies has also made it difficult to compare results to a standard Before-After-Control-Impact sampling design (Green, 1979; Underwood, 1994). Thus, we are unable to quantitatively compare our data to previous years for a direct assessment of changes over time. As we are only able to directly compare our results between impact and reference locations after the installation of the desalination plant, it was challenging to tease out the effects of the ADP outflow from natural spatio-temporal variation across our sites. We have attempted to alleviate this effect by using multiple reference locations (Underwood, 1994). Future monitoring planned for this area should continue with the current sampling design to enable the assessment of changes over time.

Whether our findings are consistent to the effects of other desalinisation plants on fish assemblages is difficult to assess. Desalination plants that have been in operation for a long period or those situated in different environmental conditions are unlikely to provide good comparisons due to differences in habitat types and environmental conditions. There are also very few studies published assessing the ecological impacts of desalination, with ∼48% of studies published being reviews and less than 1% assessing impacts on ecosystem (Roberts *et al.*, 2010). In addition, studies investigating the effects of desalination-plant effluents are often not publicly available. However, some studies have recently assessed the effects of the Sydney Desalination Plant on sessile invertebrates (Clark *et al.*, 2018) and fish assemblages (Kelaher *et al.*, 2020). These studies found higher abundances and diversity of fish and decreased diversity of sessile invertebrates, and suggest that this was likely due to the turbulent mixing of discharge attracting demersal and pelagic species to the area, disturbing the benthos, rather than caused by the minor salinity change (∼ 1 g/L) (Clark *et al.*, 2018; Kelaher *et al.*, 2020). Our study showed similar results with small salinity changes in the surrounding area (<1.3 g/L; Ayala *et al.*, 2015) and comparable fish abundance and diversity to reference locations. These promising results demonstrate that fish assemblages are not negatively affected by well-designed desalination plants.

Conclusion

Overall, temperate fish assemblages varied among years and seasons, and differences due to the ADP outfall were smaller than those due to reef type, geographical location, and protection from fishing. The increased fish abundance and diversity at the outfall site was not the typical response associated with the expected negative impacts of anthropogenic activities on fish assemblages, suggesting that the saline waste discharge from the APD has no detrimental impact on fish assemblages associated with the desalination outfall area.

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This article is dedicated to the legacy of Peter G. Fairweather, who sadly passed away days after its acceptance. Peter's contributions to the field of marine science cannot be overstated. He was an influential quantitative ecologist who played a leading role in the development of marine protected areas and who shaped the minds of generations of students.

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Data availability statement

Data are stored in an online repository named GlobalArchive (http://globalarchive.org).

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