Estuarine, Coastal and Shelf Science 167 (2015) 390–403

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Coastal nitrogen plumes and their relationship with seagrass distribution

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ARTICLE INFO

Article history:
Received 9 April 2015
Received in revised form
4 September 2015
Accepted 14 October 2015
Available online 24 October 2015

Keywords:
Seagrass
Seaweed
Nitrogen isotopes
Sewage
Wastewater
Outfalls

Regional terms:
Australia
Gulf St Vincent
Adelaide

ABSTRACT

Urbanised coastlines are affected by cumulative impacts from a variety of anthropogenic stressors, but spatial information on the distribution of these stressors at the local scale is scarce, hindering the ability of managers to prioritise mitigation options. This work investigated the spatial footprint of land-based nitrogen discharges to a metropolitan coastline and assessed the potential role of this stressor alone on seagrass dynamics at the scale of the ecosystem. The macroalga *Caulocystis cephalornithos* was used as a time-integrative sampler of nitrogen in the water column over 202 sites monitored across an area of ~800 km^2. The stable isotopic signature of nitrogen in tissues (δ^{15}N) was used to map plumes of anthropogenic origin. The surface area of these plumes was found to be proportional to nitrogen loads from land. The largest plume was associated with discharges from an industrialised estuary and a wastewater treatment plant, where a monthly nitrogen load in excess of 110 tonnes affected an area >80 km^2. The location and size of the plumes changed with seasons as a result of wind forcing and rainfall/wastewater reuse. The location of the plumes was compared to published seagrass distribution obtained from video transects. Dense seagrass meadows only occurred in areas unimpacted by plumes throughout the year, mostly in shallow (<5 m) regions for *Amphibolis antarctica*, and deeper (5–10 m) for *Posidonia* sp., possibly as a result of this species higher tolerance of low light conditions. This higher tolerance might also explain why *Posidonia* sp. is observed to preferentially recolonise areas of previous loss in the region. While a decrease in the spatial footprint of nutrient plumes has created conditions for natural seagrass recolonisation in some areas, it did not halt seagrass loss in others, suggesting the influence of additional stressors such as wave dynamics and light attenuation due to turbid/coloured stormwater.

1. Introduction

The management of urbanised coastlines involves the identification of local and global stressors, and the prioritization of mitigating options available to local managers to increase system resilience against cumulative impacts (Brown et al. 2014). Spatial information regarding the distribution of stressors is however scarce in marine systems when compared to terrestrial environments. A point in case is the management of seagrass ecosystems, which have been progressively declining around the globe as a result of a suite of anthropogenic stressors, which are difficult to separate and rank in impact (Grech and Coles, 2010). Global decline has been estimated at approximately 30% of the original cover (Waycott et al. 2009), and 14% of all seagrass species are now considered at risk of extinction (Short et al. 2011).

Coastal seagrass meadows adjacent to large population centres have been severely affected, with loss generally linked to wastewater and stormwater discharges resulting in changes in the underwater light climate (e.g. Walker and McComb, 1992; Short and Wyllie-Echeverria, 1996). These changes stem from additional

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http://dx.doi.org/10.1016/j.ecss.2015.10.016
0272-7714/

inputs of coloured dissolved organic matter and suspended sediments, and more nutrients promoting phytoplankton and epiphyte competition for light (Kemp et al. 2004). In Adelaide (population 1.2 million), South Australia, anthropogenic pressure has been implicated in the loss of approximately 40% of seagrass cover (or ~60 km^2) (Tanner et al. 2014). These meadows are considered a hotspot for seagrass diversity, comprising at least 9 of the 18 species currently recorded for the temperate southern coastlines of Australia (Short et al. 2007; Bryars and Rowling, 2009; Erftemeijer, 2014).

Similar to trends observed in Europe and North America (e.g. Orth et al. 2006; Airoldi and Beck, 2007; Garrido et al. 2013; Gurbisz and Kemp, 2014; Marba et al. 2014), land-derived inputs to the Adelaide coast peaked in the late 70s/early 80s, when the rate of seagrass loss also accelerated (Environment Protection Authority, 1998; Wilkinson et al. 2005). At the time, point inputs to the coast amounted to 3550 tonnes of nitrogen and 25,700 tonnes of sediments per year (Wilkinson et al. 2005; McDowell and Pfennig, 2013). The nitrogen discharges derived mainly (65% of total) from wastewater treatment plants (WWTPs), but there were also inputs from an industrial outfall in the estuary of the Port River (28%) and stormwater (7%). Sediment inputs were more evenly distributed between industrial (39%), stormwater (36%), and wastewater (25%) sources. More than half of the sediment discharge from WWTPs derived from two outfalls discharging sewage sludge directly to the coast. By 2011, the closure of these sludge outfalls, investment in infrastructure and operation of WWTPs, wastewater reuse, and catchment management achieved a reduction in inputs of 44% of peak values for nitrogen and 40% for sediments. Continuing improvements and the closure of the industrial outfall in 2013 achieved further reductions of 75% for nitrogen and 80% for sediments.

Despite considerable effort to improve water quality along the Adelaide coast for the past three decades, signs of system recovery have been slow to appear. They only started to emerge during a drought period in the late 2000s (van Dijk et al. 2013) with small localised areas of natural seagrass recolonization (Bryars and Neverauskas, 2004), and some success in assisted seagrass restoration programs (Tanner et al. 2014). Several data gaps hamper the progression of an adaptive management framework for the system, including understanding the time lag between environmental improvement and seagrass recovery, conditions leading to loss versus conditions necessary for recovery, and spatial data on the distribution of local stressors at appropriate scales.

The aim of this work was to investigate the role of land-derived nutrient plumes as a local stressor to seagrass meadows in the Adelaide region. We used the nitrogen isotopic signature of translocated macroalgae to map nutrient plumes in an area of ~800 km^2 and compared the spatial footprint of these plumes to seagrass distribution, loss and recovery. The use of macroalgae to map nutrient plumes ensures that only bioavailable inputs are considered, with the stable isotopic composition of the assimilated nitrogen indicating the source of nutrients (Costanzo et al. 2005; Savage, 2005; Derse et al. 2007; Dillon and Chanton, 2008; Garcia-Sanz et al. 2011). Enrichment in the heavy isotope (^{15}N) acts as a marker for wastewater and industrial effluents (Kendall et al. 2007; Fernandes et al. 2009; Fertig et al. 2009; Mancinelli and Vizzini, 2015), which accounted for 94% of point sources of nitrogen entering the system when the study was performed in 2011. The technique to map land-derived nutrient inputs using macroalgal isotopic signatures had been refined and used previously in Adelaide to trace wastewater plumes during the dry summer season (Fernandes et al. 2012). Here, we investigated the impact of seasonal weather patterns on the location of these plumes, in particular the change in wind direction from the south to the north between the dry season (December–April) and the wet season (July–September) (Wilkinson et al. 2005; Pattiaratchi et al. 2007). We then used this information to test whether the location and size of the plumes had an effect on seagrass dynamics at the scale of the ecosystem.

2. Methods

2.1. Seawater and effluent collection and analysis

Seawater and effluent samples were collected to determine the concentration and isotopic signature of nitrogen in inputs and the receiving environment. Nitrogen isotopic abundances are reported as $\delta^{15}N$, which corresponds to the deviation (in ‰) of the isotopic composition of the sample from the isotopic composition of nitrogen in air (Peterson and Fry, 1987):

$$\delta^{15}N = \left(\frac{R_{sample}}{R_{air}} - 1\right) \times 10^3 \quad (1)$$

where $R = {}^{15}N/{}^{14}N$.

Seawater samples were collected from the surface (<1 m) at 4 coastal sites adjacent to wastewater outfalls or the Port River mouth (sites closest to each point source in Fig. 1). Effluent samples were collected from each WWTP, and the industrial outfall discharging in the Port River (Fig. 1). Samples were collected in the first week of April, and between July and August 2011.

Samples were transported on ice and filtered upon arrival at the laboratory through pre-combusted (400 °C for 4 h) glass fibre filters (MFS GF-75, 0.7 μm, 47 mm diameter). Sub-samples were kept for nutrient analysis and stored frozen. The concentrations of total ammonium (NH_4^+), nitrate/nitrite (NO_x) and total nitrogen were determined by flow injection analysis (FIA) in a Lachat QuickChem QC8500 Automated Ion Analyser. Analytical precision was typically better than 5% based on certified reference seawater supplied by the Queensland Health Forensic and Scientific Services (Brisbane, Australia).

Measurement of the nitrogen isotopic composition of ammonium ($\delta^{15}NH_4$) was based on the diffusion method by Holmes et al. (1998). Briefly, filtered samples were transferred to HDPE incubation bottles. NaCl (50 g L^{-1}) was added to effluent samples with low salinity (i.e. those from WWTPs), and MgO (3 g L^{-1}) and a diffusion trap were added to all samples. The volumes used were 4 L for seawater, and 10–1000 mL for effluents, depending on expected ammonium concentrations. The diffusion trap consisted of an acidified Whatman GF/D filter inside a porous Teflon filter pack. Samples, procedural blanks and standards matching the expected concentration range of samples, were incubated for 2 weeks in a shaker table placed inside an incubator at 40 °C.

After incubation, diffusion traps were removed from the incubation bottles, dried in a desiccator with silica gel and an open container of concentrated sulphuric acid, and stored in individual vials with sealing caps. The GF/D filters inside the diffusion traps were analysed for nitrogen and its stable isotopes using a Carlo Erba CE1110 CHN-S analyser coupled to a Fisons Isochrom CF-IRMS (Continuous-Flow Isotope Ratio Mass Spectrometer). The runs were calibrated against laboratory standard materials, including gelatin and alanine. Analytical precision was typically 0.2‰. Reproducibility is based on replicate incubations of standards and was typically better than 0.3‰. The larger incubation volumes used for the analysis of ammonium in seawater samples (4 L) and in the Glenelg effluents in the wet season (1 L) are typically associated with lower recoveries, and therefore we attempted to correct the $\delta^{15}NH_4$ of these samples for isotopic fractionation calculated from standards (Holmes et al. 1998).