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Parameterization of an optical model to refine seagrass habitat requirements in an urbanized coastline

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A B S T R A C T

Accurate simulation of the underwater light climate is a requirement to understanding and predicting the ecological benefits from initiatives to manage land-derived inputs of nutrients and suspended solids to the coast. The goal of this work was to derive an empirical light model to be used in the water quality module of a coupled physical-biogeochemical model, developed to estimate habitat suitability for seagrasses in a metropolitan region (Adelaide, South Australia) affected by wastewater and stormwater discharges. Paired measurements of water quality and the vertical attenuation coefficient (K_d) obtained at six fixed sites between summer and winter were used in a multiple regression analysis to optimize local specific attenuation coefficients for coloured dissolved organic matter (CDOM) (2.623 cm m^{-1}, measured as absorption at 254 nm), chlorophyll *a* (0.071 m^2 mg^{-1}), suspended solids (SS) < 63 μm (0.032 m^2 g^{-1}) and >63 μm (0.001 m^2 g^{-1}). The fitted relation explained 65% of the variability in K_d, with a standard error of 0.10 m^{-1}. This error is small considering a mean K_d value of 0.35 m^{-1}, varying between 0.12 and 0.89 m^{-1}. Fine SS contributed on average 6 times more light attenuation (34%) than coarse SS (6%). CDOM was the second largest contributor to K_d (36%). The contribution of chlorophyll *a* to K_d (13%) was only marginally higher than the background imparted by seawater (12%). In shallow waters (3–5 m), K_d values were >0.4 m^{-1} and heavily influenced by sediment resuspension. In deeper waters (12–15 m), K_d values decreased and were dominated by CDOM. The light climate in the north of the study region was also heavily influenced by CDOM inputs, not only from point sources but also from mangroves and seagrass beds. This analysis suggests the control of fine SS as the most powerful management option to deter further nearshore seagrass losses in the region, and promote seagrass restoration.

1. Introduction

Anthropogenic inputs of nutrients and sediments act to deteriorate the underwater light climate supporting key habitat forming species in coastal regions (Burkholder et al., 2007; de Boer, 2007; van der Heide et al., 2011). While light limitation of phytoplankton occurs at 1% surface irradiance (Sverdrup, 1953; Strickland, 1958), seagrasses have high respiratory demands and require as much as 10–15% surface irradiance for optimal growth (Duarte, 1991; Ralph et al., 2007). High light requirements affect the ability of seagrasses to survive chronic turbidity and epiphyte cover associated with continuous land inputs, as well as to cope with and recover from stochastic disturbance from extreme events such as storms. Changes in seagrass habitat in turn affect sedimentary nutrient fluxes, wave attenuation, water residence time and other buffering and feedback mechanisms that impact on the susceptibility of coastal ecosystems to eutrophication (Hendriks et al., 2008; Eyre et al., 2011; Koftis et al., 2013; Adams et al., 2016).

The development of models to predict the effect of management actions on habitat suitability for seagrasses is a necessary step for cost-effective investment in initiatives for the conservation and restoration of meadows. This study is based on one such a model developed for Adelaide, a 1.3 million metropolitan centre spanning

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70 km along the east coast of Gulf St Vincent, South Australia. The region, considered a hotspot for seagrass diversity in temperate Australia (Short et al., 2007; Bryars and Rowling, 2009; Erftemeijer, 2014), has progressively lost ~60 km^2 of its original seagrass cover since the 1940s (Tanner et al., 2014). These changes have prompted management agencies to set aspirational reduction targets for both nitrogen and suspended solids (SS) to bring loads in the coastal zone down to 1 t N/km^2 and 7 t SS/km^2 (Fox et al., 2007). More recently, the Adelaide Receiving Environment Model (AREM) has been developed to provide further definition of sustainable loads in time and space. AREM is based on the Delft3D modelling suite, and includes four modules that simulate hydrodynamics, waves, water quality and seagrass habitat suitability (Zijl et al., 2014). This paper provides an overview of the development of an empirical light attenuation model for inclusion in the water quality module of AREM.

The empirical light model is based on the relationship between water quality and the vertical attenuation coefficient (K_d). The latter quantifies the exponential decay of light with depth from absorption and scattering of photons by both living (phytoplankton) and non-living particulate matter (detritus and inorganic sediments), coloured dissolved organic matter (CDOM), and seawater molecules. Assuming that the effects of each optically active constituent are additive in nature (Gordon, 1989; Gallegos, 2001), K_d is commonly approximated as:

$$K_d = K_{sw} + K_{CDOM} + K_{Chla} + K_{Det} + K_{IM} \tag{1}$$

where K_{sw}, K_{CDOM}, K_{Chla}, K_{Det} and K_{IM} are the partial attenuation coefficients of pure seawater, CDOM, chlorophyll *a* (Chla, as a proxy for phytoplankton), detritus and inorganic matter, respectively. The partial attenuation coefficient of each constituent i (K_i) in turn is calculated as a function of its specific attenuation coefficient k_i and concentration [I]:

$$K_i = k_i \times [I] \tag{2}$$

In this study, we used an empirical approach to estimate local k values for CDOM, Chla, detritus, and both fine (<63 μm) and coarse inorganic matter. Water quality and light data were obtained at fixed sites between summer and winter and used to estimate and optimize k values for the region using multiple linear regression. The calculated partial attenuation coefficients were then used to identify the main drivers controlling light in the system. This optical model, implemented in the water quality module of AREM, allows the estimation of the intensity of light available in the photosynthetically active radiation (PAR) part of the spectrum (400–700 nm) and the identification of areas where seagrass presence or recovery might be inhibited by light. This information will help target improvements into the precise locations and periods of the year most likely to deliver ecological gains for seagrasses. The study's approach can be extended to address similar problems in other urbanized regions, with the local derived k values being particularly relevant for Southern Australia.

2. Materials and methods

2.1. Study area

The coast of Adelaide (Fig. 1) borders Gulf St Vincent, a large embayment opening to the Southern Ocean. The Gulf is an inverse estuary, with salinity increasing towards its upper reaches as a result of minimal surface runoff and a semi-arid regime where evaporation is in excess of precipitation. The mean annual wave height decreases from south to north following a broadening of the bathymetry towards the upper Gulf (Pattiaratchi et al., 2007). Catchments in the north and south of the Adelaide coast are mainly agricultural with the usual pattern of flow restricted to pulse events around winter (June–September) (Wilkinson et al., 2005b). The urbanized river systems in the central coast flow more frequently due to a large proportion of impervious surfaces in these catchments (Wilkinson et al., 2005a). Tidal currents parallel to the coast dominate the circulation and trap land-based inputs in water depths <10 m (Pattiaratchi et al., 2007). Coastal waters are vertically well mixed, with riverine and wastewater inputs having no significant effect on salinity (Kaempf, 2006). In 2014/15, rivers were the main source of SS (53%) to the coast, while wastewater treatment plants (WWTPs) were the main source of nitrogen (82%) (Jones, 2015; SA Water data). Two rivers discharged more than half of the annual riverine load of 3831 t of SS: the central Torrens River (60%) and the northern Gawler River (10%) (Jones, 2015). The load of 650 t of nitrogen from WWTPs was delivered by the outfalls of Bolivar (68%), Glenelg (22%) and Christies Beach (10%) (Katherine Reid, SA Water, personal communication).

2.2. Sampling

Between November 2014 and September 2015, 97 paired water quality and light data were obtained at fixed sites ranging from shallow to deep waters in two different locations of the Adelaide coast (Fig. 1). Shallow sites were located in fragmented/sparse seagrass whilst the deeper sites were located in dense seagrass meadows. One location at the mouth of Barker Inlet is impacted by effluents from the Bolivar WWTP, the largest in Adelaide (~40 GL $year^{-1}$). The Bolivar transect had 3 sites located at 3 m (B3), 7 m (B7) and 12 m (B12) mean sea level depth. The other location, near Grange Beach, is impacted by inputs from the Torrens River, the single largest source of SS to the metropolitan coast. This transect also had three sites, located at 5 m (G5), 10 m (G10) and 15 m (G15). Sites were surveyed for both light and water quality constituents on the same day, over 18 different days, but a few light data points were missed due to occasional flooding of light meters (B3 n = 17, B7 n = 15, B12 n = 16, G5 n = 17, G10 n = 15, G15 n = 17).

For the measurement of light, two Odyssey PAR 2π cosine light sensors (Dataflow Systems, New Zealand) (Long et al., 2012) were deployed 0.7–1 m apart at each site, with the lowest light meter placed 0.6–0.9 m from the bottom to avoid sediment interference, while approximating the height of the seagrass canopy (Collings et al., 2006; Pedersen et al., 2012). The light meters were attached to a steel frame with arms separated by 180° to preclude shading of the lower light meter. The light meters recorded average light intensity (pulse counts) for each 30 min period and were deployed for periods of time from two weeks to over a month. Calibration against a 2π LI-190R Licor sensor was used to convert data obtained from the Odyssey sensors to μmol photons m^{-2} s^{-1}.

For deeper sites with low light attenuation, K_d could not be determined with the <1 m spacing used between the underwater sensors (see section 2.4). For this reason, two Odyssey light meters were also deployed on land (sites L1 and L2 in Fig. 1), together with a 2π LI-190R Licor sensor. For periods when land-based light data were not available, data were obtained from a site maintained by the Adelaide and Mount Lofty Ranges Natural Resources Management Board at Virginia (site L3).

Copper tape was used around the underwater light sensors to discourage biofouling, and sensors were retrieved and/or cleaned on an approximately fortnightly basis. The data from each sensor was plotted against each other at the beginning and end of deployments to check for any interference from fouling. The good accuracy of K_d calculated from data collected by the underwater sensors in the shallower sites (high light attenuation) when