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Intertidal seagrass in Ireland: Pressures, WFD status and an assessment of trace element contamination in intertidal habitats using *Zostera noltei*

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A B S T R A C T

Intertidal seagrass has been selected as a Biological Quality Element for the assessment of ecological status under the Water Framework Directive. In Ireland, two species of seagrass, *Zostera marina* and *Z. noltei* occur in intertidal habitats. This study presents the first comprehensive assessment of the distribution and Water Framework Directive status of intertidal seagrass in the Republic of Ireland and Northern Ireland. Most of the areas assessed, using the Water Framework Directive-compliant assessment tool, have a status of HIGH or GOOD. Only two areas were found to have a status less than GOOD and in both, the cause for the decline was smothering by opportunistic foliose green macroalgae. Linear regression showed a relationship between pressure index scores and Ecological Quality Ratio, showing the relevance of the index as a metric of anthropogenic pressure. Trace element concentrations were examined in *Z. noltei* tissues and Trace Element Pollution Index values were calculated. The relationship between Trace Element contamination and Water Framework Directive status was examined but the results showed little correlation. However, a relationship between the pressure index and trace element contamination was obtained. This assessment provides the most comprehensive overview of intertidal seagrass beds in Ireland and establishes a strong baseline for ongoing monitoring and assessment under the Water Framework Directive. The data provide key information on the pressures affecting these valuable habitats which will assist in the development of measures to improve and protect our transitional and coastal waters.

1. Introduction

Seagrasses are represented by as many as 65 species in 4 families and 11 genera (Algaebase, 2017) and occur widely in all coastal areas of the globe, except in the Antarctic (Hemminga and Duarte, 2000). In Ireland and the United Kingdom, seagrasses generally comprise species of eelgrass (*Zostera* spp.). Several species of tassleweed (*Ruppia* spp.; Potamogetonaceae) also occur in the North Atlantic, but are not generally considered as true seagrasses.

It is widely accepted that seagrasses are critical to many coastal marine ecosystems (Klumpp et al., 1989), housing diverse communities in the intertidal (Li and Huang, 2012), while facilitating foraging (Riosmena-Rodriguez et al., 2010), shelter, spawning and rearing (Hemminga and Duarte, 2000) for a plethora of aquatic species. Importantly, seagrass ecosystems also stabilize sediment, are important in carbon and nutrient cycling and support grazing and detrital food webs (Lewis and Devereux, 2009). However, their proximity to the coastline renders them vulnerable to a variety of anthropogenic threats (Ambo-Rappe, 2014) including direct pressure from physical disturbance (Tu Do et al., 2012; Mauro et al., 2013), indirect pressures such as nutrient enrichment (Fertig et al., 2013; Hauxwell et al., 2001), the alteration of sediment transport (Mauro et al., 2013) and encroachment of invasive species (Grilo et al., 2012). It is predicted that within a few decades more than half of the world's rapidly growing human population will live within 150 km of the coast (Cohen, 2003) where anthropogenic pressures on seagrasses will inevitably intensify.

A consequence of this anthropogenic pressure include the input of trace elements (TEs). Unlike pesticides, TEs occur naturally in the environment some of which are essential micro-nutrients (Avelar et al., 2013). However, if present and available above a certain threshold

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(which varies between species) TEs have the potential to be toxic to organisms (Rainbow, 1995) for example Cu in eelgrass (Nielsen et al., 2017; Zhao et al., 2016). Anthropogenic influences can increase the bioavailability of TEs through diffuse and point source pollution (Llagostera et al., 2011). Some TEs can be toxic and persistent in the marine environment (Bargagli, 1998) (e.g. Hg, Cd, Pb) and thus, accumulate in marine organisms over time and enter the food chain (Nixon and Fulweiler, 2009).

Seagrass beds have experienced considerable global decline due to anthropogenic influences and disease over the last 200 years (Duarte et al., 2008). These declines have also been recorded in Ireland with significant declines during the 1930s due to the wasting disease caused by the heterokont *Labyrinthula zosterae* D. Porter & Muehlstein, which decimated the Atlantic seagrass of Europe and North America. The greatest local decline was primarily documented by Lynn (1936) and the wasting disease still occurs in North America and Europe, but has not had such a catastrophic affect in recent years (Ralph and Short, 2002). The reasons for disease outbreaks are not fully understood (Giesen et al., 1990a; Giesen et al., 1990b), but it is possible that *Zostera* plants only succumb when stressed by other environmental factors such as genetic isolation, increases in water temperature, extreme precipitation, or pollution (Short et al., 1988).

As the only truly marine angiosperms, seagrasses have been selected as a Biological Quality Element (BQE) under the European Union Water Framework Directive (WFD) (Foden, 2007; EC, 2000). Marine monitoring programmes have been developed across EU member states to assess the taxonomic composition and abundance of seagrass beds primarily focusing on the intertidal beds (Foden and Brazier, 2007; García-Marín et al., 2013; Neto et al., 2013). Intertidal seagrass assessment was also included in the EU WFD intercalibration process, aimed at ensuring that different member states assess biological quality elements in comparable ways and that different assessment systems are considering the biological condition at similar levels of anthropogenic disturbance (EC, 2008; EC, 2013), although for seagrasses this process is ongoing.

As part of Ireland's WFD monitoring programme (EPA, 2006), an intertidal-seagrass monitoring programme was initiated in 2007. While the minimum requirement under the WFD is for a single survey in each 6-year River Basin Management Plan (RBMP) cycle, where possible, more frequent surveys have been undertaken to help establish a better understanding of the baseline distribution and annual variability of Ireland's intertidal seagrass stocks (Foden and Brazier, 2007). Additionally, surveys have been undertaken in areas outside those selected for routine WFD monitoring to allow for a comprehensive gathering of baseline information which may be useful for other purposes such as biodiversity assessment or requirements under other EU directives.

Several trans-national strategies aimed at determining ecological quality of the aquatic environment have prompted the use of biotic indices (Oliva et al., 2012; Royo et al., 2011), including the WFD (EC, 2000). Seagrasses have widely been suggested as good biomonitors of TE pollution (Conti et al., 2015; Lafabrie et al., 2007; Luy et al., 2012; Ahmad et al., 2015; Cambrolle et al., 2008), due to their metal bioaccumulation capacity through above and below ground tissues (Llagostera et al., 2011) and, their ability to reflect ambient metal concentrations in the water column and sediment by examination of leaves and roots respectively (Bouchon et al., 2016; Lafabrie et al., 2007).

The criteria for a good biomonitoring programme have been outlined in Rainbow (1995); chosen biota should be easily identified, sedentary, long-lived, abundant and resistant to handling stress, tolerant to variations in physico-chemical parameters, available throughout the year, and cosmopolitan in their distribution to allow for more relevant international comparison of metal contamination in biomonitors (Rainbow, 2006; Rainbow and Phillips, 1993). Seagrasses generally satisfy these criteria and as such have been demonstrated as effective biomonitors of TE pollution in aquatic environments (Bonanno et al., 2017; Tranchina et al., 2005; Sanchiz et al., 2000).

Biomonitors have been suggested as being preferable over typical measures of TE contamination i.e. environmental matrices of water and sediment (Richir and Gobert, 2014b); however, a recent study has promoted a framework of integrated chemical and biological effects monitoring and assessment (Vethaak et al., 2017). Biomonitors reflect chronic impacts by providing more than a snapshot of environmental contamination (Linton and Warner, 2003). They are less susceptible to contamination during processing than traditional matrices (Rainbow and Phillips, 1993) and, importantly, biomonitors give a more relevant measure of the bioavailability of TEs in ecosystems (Conti et al., 2010; Rainbow, 2006; Rainbow, 1995). Bioavailability is a much more appropriate measure than total metal concentration which is given from typical environmental matrices as it reflects the quantities which are of ecotoxicological relevance (Rainbow and Phillips, 1993). Importantly, seagrasses have been shown to be indicative of pollution at higher trophic levels (Govers et al., 2014) and as such can act as first level indicators of TE contamination in the food web (Avelar et al., 2013; Prange and Dennison, 2000) giving early warning of metal contamination in higher trophic levels (Bonanno and DI Martino, 2016; Ferrat et al., 2003).

The main goals of the present study were: (i) to offer the first comprehensive assessment of the WFD status of the Ireland's intertidal seagrass communities; (ii) to carry out initial assessments of anthropogenic pressures acting on these beds and the likely effects of these interactions, (iii) assess TE contamination in coastal ecosystems, and the impact that this indirect anthropogenic pressure may be having on WFD status. The information provided here forms part of WFD assessments of transitional and coastal habitats and will assist in the future implementation of the WFD.

2. Methods

2.1. Site locations and sampling

Intertidal seagrass bed locations (Fig. 1) are summarised in Table S1 (supplementary material).

The assessment tools for intertidal seagrass were developed through the United Kingdom and Ireland (UK-IE) Marine Plant Task Team (MPTT) and have been described by various authors (Foden et al., 2010; Foden, 2007; Uktag, 2014). Data was gathered in-situ and *Z. noltei* Hornemann [also widely known as *Z. noltii*] samples (above and below substrate tissues) were collected either on foot, for small beds where sediment type allowed, or using a light hovercraft. The hovercraft was used in areas where the sediment was too soft to allow safe access or where the seagrass beds were too large to map safely during a single tidal cycle. The hovercraft is a non-invasive way to assess sensitive habitats and any shallow tracks it leaves are removed on the following tide (Abele and Brown, 1977).

Monitoring surveys were undertaken in July-September during the period of peak growth of the seagrass and to ensure minimum disturbance to birds. The arrival of overwintering geese towards the end of September can also lead to a reduction on bed extent due to grazing (Wyer et al., 1977; Vermaat and Verhagen, 1996).

2.2. WFD classification

Ecological classification was calculated according to Foden and Brazier (2007). The Ecological Quality Ratio (EQR) is calculated using three metrics; spatial extent, percentage cover and taxonomic composition with boundary conditions detailed in Table S2 (Supplementary material). Following the WFD intercalibration process, the good-moderate boundary was adjusted to 0.63 and high-good to 0.8 for coastal BQEs and for transitional waters, the good-moderate boundary was adjusted to 0.7 and high-good to 0.83 (EC, 2013). At HIGH ecological status, the taxonomic composition corresponds totally or nearly totally to undisturbed conditions