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Removal treatments alter the recruitment dynamics of a global marine invader - Implications for management feasibility

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ABSTRACT

Frameworks designed to prioritise the management of invasive non-native species (INNS) must consider many factors, including their impacts on native biodiversity, ecosystem services, and human health. Management feasibility should also be foremost in any prioritisation process, but is often overlooked, particularly in the marine environment. The Asian kelp, *Undaria pinnatifida*, is one of the most cosmopolitan marine INNS worldwide and recognised as a priority species for monitoring in the UK and elsewhere. Here, experimental monthly removals of *Undaria* (from $0.2\,m^2$ patches of floating pontoon) were conducted at two marinas to investigate their influence on recruitment dynamics and the potential implications for management feasibility. Over the 18-month experiment there was no consistent reduction in *Undaria* recruitment following removals. Cleaning of pontoon surfaces (i.e. removal of all biota) led to significant short-term reductions in recruitment but caused a temporal shift in normal recruitment patterns. Non-selective removal (i.e. all macroalgae) generally promoted recruitment, while selective removal (i.e. *Undaria* only) had some limited success in reducing overall recruitment. The varied results indicate that the feasibility of limiting *Undaria* is likely to be very low at sites with established populations and high propagule pressure. However, where there are new incursions, a mixture of cleaning of invaded surfaces prior to normal periods of peak recruitment followed by selective removal may have some potential in limiting *Undaria* populations within these sites. Multi-factorial experimental manipulations such as this are useful tools for gathering quantitative evidence to support the prioritisation of management measures for marine INNS.

1. Introduction

Invasive non-native species (INNS) can cause significant environmental impacts to the native communities to which they are introduced (Simberloff et al., 2013; Early et al., 2016). There is also major economic cost associated with their management, control and remediation (Pimentel et al., 2005; Williams et al., 2010). Consequently, there is increasing pressure to control the introduction, spread and proliferation of INNS. New legislative tools, such as those adopted in the EU (EU, 2014) and USA (Federal Register, 2016), aim to improve prevention via greater biosecurity, containment and eradication of INNS. Rapid response eradication is generally accepted as the best management option once a new species is detected and biosecurity measures have clearly failed (Beric and MacIsaac, 2015; Early et al., 2016). But when an INNS becomes widespread, available management options are often limited, and can be highly costly, time-consuming and ineffective, especially in highly connected marine environments (Bax et al., 2003; Simberloff et al., 2013; Early et al., 2016; Courtois et al., 2018). As environmental managers have finite resources with which to tackle an ever-increasing number of INNS, management prioritisation procedures are clearly needed (Bonanno, 2016; McGeoch et al., 2016; Seebens et al., 2017; Courtois et al., 2018).

In order to design a prioritisation framework, many factors must be considered, including ecological and economic impacts, the provision of ecosystem services and effects on human health (McGeoch et al., 2016; Epstein, 2017). Many of these factors can be highly subjective and are hard to define and quantify. Therefore more attention has recently been given to the important and less subjective issue of management feasibility (Molnar et al., 2008; Panetta and Novak, 2015; Booy et al., 2017; Corbin et al., 2017). Understanding the likely effectiveness, practicality, risk, cost, impact and timeframe of management options should be fundamental to any prioritisation process.

Evaluating the feasibility of management actions for INNS in terrestrial ecosystems is aided by the historic nature of introductions, the quantity of research and the pre-existence of numerous management programmes (Kettenring and Adams, 2011; Veitch et al., 2011; Panetta

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and Novak, 2015; Corbin et al., 2017). In contrast, the management of INNS in marine ecosystems is comparatively new and understudied, although some control and eradication programmes have been implemented (Bax et al., 2003; Williams and Grosholz, 2008; Beric and MacIsaac, 2015). The inherent connectivity of marine environments can promote the spread of INNS and re-entry to cleared areas (Ruiz et al., 1997; Bax et al., 2003), while their relative inaccessibility renders monitoring efforts and management actions far more difficult (Ruiz et al., 1997; Bax et al., 2003; Thresher and Kuris, 2004; Booy et al., 2017). Large-scale management of marine INNS is, therefore, highly costly. Thus, small-scale eradication or control experiments, or trials, can be an important step in determining management feasibility and prioritisation (Lovell et al., 2006; Williams and Grosholz, 2008).

The kelp, *Undaria pinnatifid*a, is one of the most cosmopolitan marine INNS worldwide (Epstein and Smale, 2017b). Native to the north-west Pacific rocky coastlines of Japan, Korea, Russia and China (Saito, 1975), *Undaria pinnatifida* (hereafter referred to as *Undaria*) can now be found in many parts of the north-east and south-west Atlantic, south-west and east Pacific, and the Tasman Sea (Epstein and Smale, 2017b; South et al., 2017). As an INNS *Undaria* is generally more widespread and abundant on artificial rather than natural substrates (Floc'h et al., 1996; Fletcher and Farrell, 1999; Cremades et al., 2006; Russell et al., 2008; Veiga et al., 2014; Kaplanis et al., 2016). Both marinas and aquaculture sites are strongly linked to introduction vectors and would therefore be expected to have high propagule pressure. They also contain large expanses of artificial substrates on pontoons or buoys which are held at a constant shallow depth, providing ideal conditions for the establishment and proliferation of *Undaria* populations (Fletcher and Farrell, 1999; Cremades et al., 2006; Grulois et al., 2011; Minchin and Nunn, 2014; James and Shears, 2016a, 2016b). *Undaria* has also invaded natural habitats across its non-native range, predominantly on sheltered to moderately wave-exposed rocky reefs (Hewitt et al., 2005; Russell et al., 2008; Dellatorre et al., 2014; Minchin and Nunn, 2014; Epstein and Smale, 2017a). In many cases the introduction of *Undaria* into natural habitats has been linked to spill-over from source populations in nearby artificial habitats, however in some cases incursions may also occur directly into natural substrates (Floc'h et al., 1996; Fletcher and Farrell, 1999; Russell et al., 2008; Grulois et al., 2011; James and Shears, 2016b; Epstein and Smale, 2017a).

Undaria is one of the most cosmopolitan marine INNS, and is considered of major importance for conservation management; yet there has been little targeted control of this species in most of its non-native range (Epstein and Smale, 2017b). Where management has been implemented, there has been some success in limiting or excluding *Undaria* in isolated environments; however, most management attempts have led to reintroduction and wider-scale spread, with localised reductions in population density being rapidly reversed following cessation of management actions (Wotton et al., 2004; Hewitt et al., 2005; Thompson and Schiel, 2012; Forrest and Hopkins, 2013; Crockett et al., 2017).

Undaria sporophytes recruit from microscopic gametophytes that may grow vegetatively in the understory for up to 2 years (Pang and Wu, 1996; Thornber et al., 2004; Choi et al., 2005). In its native range, *Undaria* has a strictly annual life cycle with recruitment restricted to the winter months (Saito, 1975; Koh and Shin, 1990). In many parts of its non-native range the thermal cues for its strict annual life cycle are lost due to the temperate environmental conditions (James et al., 2015). In these locations recruitment may occur year-round or in multiple pulses per year, however a degree of annularity generally remains (Thornber et al., 2004; Cremades et al., 2006; Casas et al., 2008; Primo et al., 2010; James and Shears, 2016a, 2016b). Although temperature is considered the key factor influencing *Undaria* recruitment patterns (Saito, 1975; Floc'h et al., 1991; Gao et al., 2013; James and Shears, 2016a; Murphy et al., 2017), recruitment may be influenced by a variety of other factors including light, temperature, salinity, depth, exposure, nutrients and competition (Russell et al., 2008; Gao et al., 2013; Watanabe et al., 2014; Epstein and Smale, 2017b; South et al., 2017). More knowledge is needed on the recruitment dynamics of *Undaria* and the effect of removal treatments in order to better design management measures and understand the factors affecting the probability of management success.

Undaria was first recorded in the UK in 1994, attached to floating marina pontoons in Port Hamble (Fletcher and Manfredi, 1995). While the majority of records originate from southern England, it has also been recorded on the east and west coasts of England, north and south west Wales, on the east coast of Northern Ireland and the Republic of Ireland, and in Scotland at Queensferry (Epstein and Smale, 2017a). There is currently no known targeted management of *Undaria* occurring in the UK (Epstein and Smale, 2017b), although it does appear on a list of priority species for monitoring and surveillance of marine INNS as part of obligations to the Marine Strategy Framework Directive (Stebbing et al., 2015). It is highly likely that as *Undaria* continues its spread and proliferation around the UK (Minchin and Nunn, 2014; Epstein and Smale, 2017a), there will be further pressure to contain or restrict the species from proliferating in certain areas. Due to their association with introduction vectors, and their possible association with spread to natural habitats, marina and harbour environments are perhaps the best candidates for implementing management actions to limit proliferation and control the spread of *Undaria* populations in the UK (Epstein and Smale, 2017a).

Four different removal treatments were applied to patches of marina pontoon during an 18-month manipulative experiment, to investigate their effects on *Undaria* recruitment patterns and elucidate the potential for control or removal of *Undaria* from marinas. There are various potential methods to control marine INNS, including biocontrol, genetic modification, biocides, herbicides and environmental remediation, however as with most plant invasions, the most commonly employed and widely accepted methods are selective physical removal or full clearance of invaded substrates (Bax et al., 2001; Thresher and Kuris, 2004; Anderson, 2007; Kettenring and Adams, 2011). The treatments in this experiment were selected to incorporate different aspects of potential physical removal methods – those which target the macroscopic INNS only, those which incorporate a more substrate-wide exclusion method, and those which target both the macroscopic and microscopic sources of INNS (Critchley et al., 1986; Wotton et al., 2004; Glasby et al., 2005; Coutts and Forrest, 2007; Forrest and Hopkins, 2013).Treatments were maintained at two marinas in Plymouth, UK, to: 1) examine how different physical and temporal removal methods effect recruitment patterns; 2) identify dissimilarities in recruitment patterns and the influence of removal methods between marinas from the same locality; 3) discern which removal method may be most efficient at reducing or excluding *Undaria*; and 4) consider the feasibility of managing *Undaria* within marina environments.

2. Methods

2.1. Site selection

Plymouth Sound is an enclosed embayment fringed by intense coastal development and large port facilities (Knights et al., 2016, Fig. 1). *Undaria* was first recorded in Plymouth Sound in 2003 within one of the waterfront marinas (NBN, 2017), and can now be found at all marinas and on much of the natural rocky-reef within the Sound at varying density and standing biomass (Epstein and Smale, 2017a). The current study was conducted at two marinas (Fig. 1), which were selected based on: 1) permission to access the facilities all-year round; 2) similar pontoon constructions; 3) large areas of pontoon which would not be disturbed by vessels or maintenance staff; 4) well established *Undaria* populations (*Undaria* was first recorded at the two chosen marinas in 2004 and 2010) (NBN, 2017). All manipulations were carried out on the vertical side of concrete-based floating pontoons, with