MONITORING THE MARINE INVASIVE/NON-NATIVE SPECIES OF JETTY BEACH, FSC DALE FORT, PEMBROKESHIRE, WALES.

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Non-native species and invasive species are widely accepted as a growing threat to global biodiversity. Yet monitoring, although vital, is patchy and inconsistent. With human influence regarded as a key contributor in the spread of non-natives, it is important for us to actively participate in long-term monitoring. In this study, the percentage cover of the non-native *Caulacanthus okamurae* (Okamura's pom-pom weed) was monitored in 30 quadrats over 8 months, in order to look for trends in growth with seasonality, sea temperature and substrate. In addition to this, frequencies of other non-natives were also recorded, demonstrating the level of monitoring able to be put into practice by staff at an environmental education centre. No overall increase in the percentage cover of the species was shown, nor a correlation between percentage cover and sea temperature. However, a statistically significant difference was shown in the percentage cover of the seaweed on two different substrates (artificial and natural), which is thought to be attributable to the seaweed's ability to colonise rapidly as an 'aggressive coloniser' on flat surfaces, thriving better on the Victorian jetty at this site, in comparison to a rocky beach. Although short for a study of this kind and failing to quantify the effect of the non-natives in this ecosystem, this study is important in highlighting the continued need for close and consistent monitoring of non-native and invasive species, and in suggesting further questions to be asked about the non-native *Caulacanthus okamurae.*

INTRODUCTION

Invasive species have been widely discussed as a threat to global biodiversity and a key element in global change (Colautti and MacIsaac, 2004; Lee, 2002; Occhipinti-Ambrogi, 2007; Ricciardi, 2016). More than 20 years ago Steffen and Walker (1997) were describing invasive species as the second biggest threat to global biodiversity. However, the actual definition for the term 'invasive species' and the associated terms such as 'non-native' and 'alien' species appears to be a fluid concept, with the terms being used interchangeably, leaving them difficult to define (Colautti and MacIsaac, 2004). Generally, the most accepted definitions are those of the CBD (Centre for Biological Diversity, 2018), which states an alien species to be "a species, subspecies or lower taxon, introduced outside its natural past or present distribution; includes any part, gametes, seeds, eggs, or propagules of such species that might survive and subsequently reproduce" and an alien invasive species to be "an alien species whose introduction and/or spread threatens biological diversity".

In the 2012 Defra study by Roy *et al*., (2012) it was stated that there is a total of 1875 established non-native species in Great Britain, although not all of these species are considered invasive. One example of an invasive species is the harlequin ladybird *Harmonia axyridis,* which was introduced in order to control the growing aphid populations in many different locations (Raak-van den Berg *et al*., 2017). The species was introduced to Europe in 1982, with the first established populations being found in 2001-2002 (Brown *et al*., 2011). Since then the ladybird has established itself in over 38 countries, spreading at rates between 100 and 500 km year^{-1} (Brown *et al*., 2011). Another example of an invasive is the "killer shrimp" *Dikerogammarus villosus* (MacNeil *et al*., 2012), an invasive from the Ponto-Caspian region of eastern Europe. The species was first detected in the UK in 2010. Since then the impact of the killer shrimp has been well studied, showing that this organism preys on many macroinvertebrates (MacNeil *et al*., 2012).

Invasive species like these require close and consistent monitoring as they are prone to fast colonisation, and so can become established very quickly. However, national records and databases can be highly inaccurate often due to misidentifications (Zenetos *et al*., 2017). For example, the list of alien species which was created in 2005 by Pancucci-Papadopoulou *et al*. was updated in 2009 and 2010 (Zenetos *et al*., 2017), and could be considered very much out of date by 2017 due to inclusion of identification errors. In this case the error being mollusc misidentification – out of the 100 potential taxa, around half were misidentifications (Zenetos *et al*, 2017). It is therefore clear that accurate monitoring is of upmost importance. This may be greatly aided by the growing interest in citizen science, which can be a very valuable tool in widespread monitoring, allowing us to gather vital data over a much wider geographical range. However, identification issues may be a problem for inexperienced citizen scientists, so some element of training will be preferable.

The spread of invasive species can often be linked to human influence and rates of invasion are increasing worldwide, especially in aquatic systems (Ricciardi, 2016). A key influencer in the spread of marine invasive species is shipping traffic (Ricciardi, 2016; Bax *et al*., 2003; Meyerson and Mooney, 2007) which makes up more than 80% of the world's trade (Bax *et al*., 2003). This is not a new concept, as ships have always inadvertently transported cargo which may be considered pests – such as diseases, rats and marine organisms. A 1750s wooden sailing ship could carry up to 120 marine organisms in its hull and another 30 organisms associating with the dry ballast and anchor chain. In a modern context, it has been suggested that at any one time 10,000 different species are being transported in ballast tanks alone (Bax *et al*., 2003).

These marine invasive species can have a huge array of influences, such as inducing the collapse of fisheries, transport of viral and bacterial pathogens, and altering their new-found habitats. This can lead to social and economic impacts such as negative effects on human health, and damage to marine-based economies. That said, invasive species can also bring opportunities to marine economies and to scientific knowledge and research. For example, as Craig (2010) argues, the presence of invasive species can increase biodiversity, which in turn leads to increased ecological interactions which may lead to ecological speciation. Furthermore, the study of introduced species may benefit wider scientific understanding of ecology and population dynamics (Allendorf and Lundquist, 2003). However, this knowledge may have little or even no use or relevance in actually managing the invasive species populations and their spread (Allendorf and Lundquist, 2003).

One key example of a marine invasive is the Slipper limpet (*Crepidula fornicata).* This mollusc is native to the east coast of North America and became established in Europe around the late 1870s due to the imports of American oysters (*Crassostrea virginica)* to the south east of Britain. Over a million tonnes of this species now exist in UK waters, and these vast numbers are able to alter habitat, change communities and even form the majority of the biomass present in any one location (McNeill *et al*., 2010). Another example is the invasive seaweed Wireweed (*Sargassum muticum),* native to Japan and introduced to North America by the 1940s and Europe by the 1970s (Pérez-López *et al*., 2014). Schaffelke, Hewitt, and Smith, (2006) considered invasive macroalgae like this a major threat to native species and to the ocean's resources across the world. Thanks to its impressive reproductive ability, *S. muticum* is now almost distributed worldwide. Unfortunately, several studies (Britton-Simmons, 2004, Kraan, 2007) have highlighted the effects it can have on native species, generally this being the most prevalent at the subtidal zone which is thought to be due to its shading effects. Both of these species have become high profile in terms of highlighting the problem of invasives, with events like "invasive species week" being used to promote awareness of this growing problem. The Slipper limpet and Wireweed have both become well established in Wales, and they can be found easily in Pembrokeshire. The 2017 report *Invasive Non-native Species (INNS) – Priority species for action in Wales* detailed 20 invasive species not yet established in Wales but predicted to arrive soon: 11 species as a priority for management, meaning they are found currently in low numbers or isolated populations; and 8 species which will require long-term management, as they are already established (Partnership, 2018). In the case of marine invasive species, examples of these established individuals included the carpet sea squirt (*Didemnum vexillum)* and the Chinese mitten crab (*Eriocheir sinensis)*. Therefore, it is clear that monitoring of marine invasive species in Pembrokeshire can provide valuable data on this global issue. Close and consistent monitoring is the key to controlling the spread of these species.

Staff and students from the FSC's Dale Fort Field Centre frequently visit Jetty Beach (Ordnance Survey 1:50,000 map SM 822 053), recording the species found there, and adding their findings to historical data for the site. Furthermore, Natural Resources Wales (NRW) have also monitored the area for several years, creating a data set which shows records of all the species present year on year. Together, these data give a clear record of when invasive and non-native species were first identified.

The non-native *Caulacanthus okamurae*, was first recorded by NRW on Jetty Beach in 2017 (Partnership, 2018); however, it was described as only present in very small amounts, in addition to other non-native species such as *Botrylloides violaceus*, *Correlia Eumyota* and *Waterispora subquorata* which were all first recorded in 2013 (Partnership, 2018). Although this monitoring is a good starting point, it is clear that further investigation is needed to monitor the non-native populations.

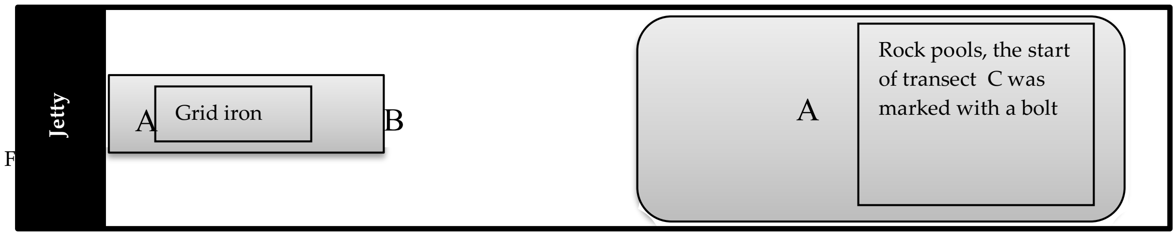
Caulacanthus okamurae is red alga occurring most commonly on the coasts of South Korea and is considered a non-native in the UK (Choi *et al*., 2001). *Caulacanthus okamurae* is also in some cases referred to as *Caulacanthus ustulatus* (Seaweed.ie, 2018), which was first identified as non-native when it was described in Spain, and up to the Bay of Biscay (including south west France); since then, the species was seen to spread to Brittany and the south west of the UK. DNA sequencing was used to show that this particular species had been introduced from the Pacific Ocean (Rueness and Rueness, 2000). Usually fairing best in warmer waters, studies have shown that *Caulacanthus okamurae* individuals grow best over a range of temperatures from 13-27°C; optimal growth occurs at 23°C (Choi and Nam, 2001). This information is used to identify any trends for a relationship between growth and sea temperature on Jetty beach. This species appears to have flourished on Jetty Beach. Here, the seasonality of this seaweed was investigated, by surveying monthly, and comparing its abundance to the average sea temperatures to look for a correlation, whilst also providing valuable records on the spread of invasive and non-native species. In addition, descriptive data on other non-natives were collected to ensure their presence is being monitored.

METHODS

A preliminary search was carried out at Jetty Beach (Ordnance Survey 1:50,000 map SM 822 053) in September 2017 to identify non-natives present. Additionally, a monthly survey of Jetty Beach was carried out, at low tide on spring tides, from November 2017, to May 2018. In addition, three, 10 m transects were established in the middle shore, perpendicular to the sea: the first being against the Victorian jetty, the next from the end of the concrete grid iron, and the final in the rock-pooled area, at an easily distinguishable point. This is illustrated below. At 1 m intervals along each transect a 0.25 m^2 quadrat frame was placed, and the percentage cover of *Caulacanthus okamurae* was recorded. This was repeated every month from October 2017 to May 2018.

Additionally, one 30 m line was established in the lower shore. The sample line was parallel to the sea, at the end of the Jetty steps, at 0.9 m above chart datum. This line was walked monthly and any marine non-natives found on rocks touching the tape were recorded in a descriptive record. This took the form of a written narrative including photographs, notes of position, appearance, colour and texture, plus length of the non-native species and the size of the boulder they were on.

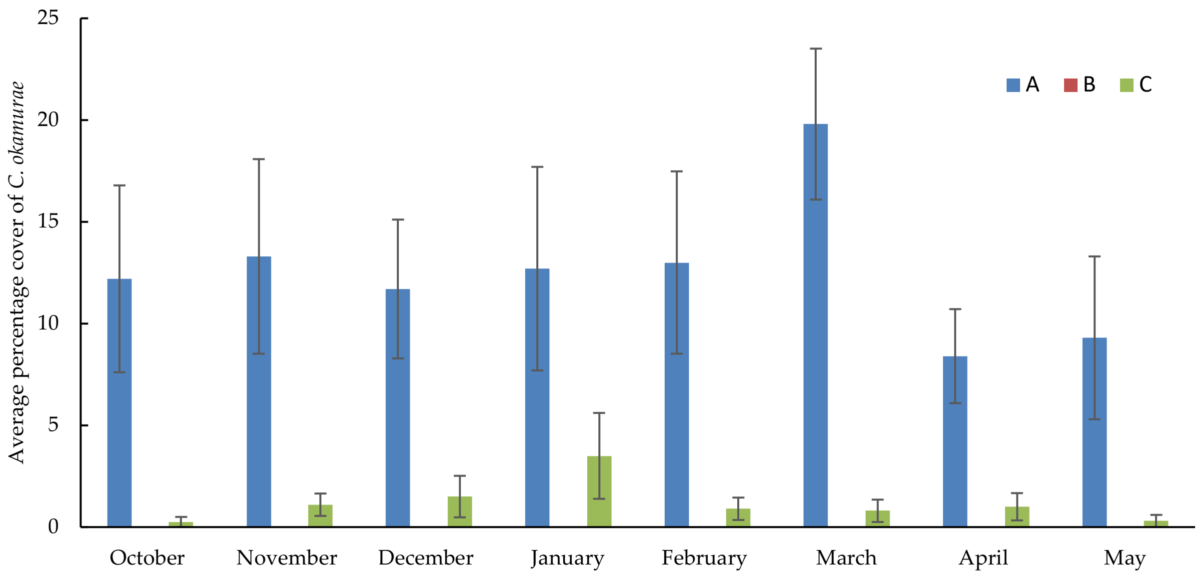
The data was collated on Excel and statistical analysis of the data was carried out in SPSS.



RESULTS

The percentage cover from each of the 10 quadrats was used to calculate a monthly average percentage cover for each of the three transects (Figure 2).

FIGURE 2: The average percentage cover of the non-native seaweed *Caulacanthus okameurae* from 10, 0.25 m^2 quadrats, over three



transects on Jetty Beach, Dale. The data were collected monthly for 8 months, from October 2017 to May 2018. The three transects are orientated perpendicular to the sea, covering a distance of 10 metres. They are ordered A, B and C from left to right of the shore (when facing upshore). No *C. okamurae* was found on transect B. The error bars shown display the standard error of the mean.

A Mann-Whitney U test was used to identify if there was a significant difference in the percentage cover of *Caulacanthus okamurae* for transects A and C, from the October data in comparison to the May data. A statistical test was not performed for transect B as the percentage cover was always measured at 0%. The test gave a p value of 0.912 for transect A, and 0.971 for transect B; therefore the null hypothesis is accepted that there is no significant difference in median between October and May on either transect.

A Mann-Whitney U test was also used to see if there was a statistically significant difference between the median percentage covers of transects A and C. This showed a p value of 0.001, allowing us to reject the null hypothesis and accept the alternative hypothesis: there is a statistically significant difference in the percentage cover of transect A in comparison to transect C.

Mann-Whitney U tests were used as the data were not normally distributed, as identified by a Shapiro-Wilks Test in SPSS.

Alongside recording the percentage cover on the three transects, observational notes on other alien species were recorded from a 30 m line parallel to the sea shore. *Botrylloides violaceus*, *Corrella eumyota* and *Watersipora subtorquata* were consistently recorded in small quantities. There were generally more of these individuals in October and November. A Slipper limpet was found in March. These were the only other non-natives recorded.

Daily sea temperature data gathered from a data logger at Jetty Beach was used to calculate a monthly average sea temperature, these averages are shown in Table 1. This data are plotted against the average monthly percentage cover of *Caulacanthus okameurae* in Figure 3. A Spearman's Rank Correlation test was used to establish if there was a statistically significant relationship between these two variables; a p value of 0.983 was calculated, suggesting no relationship between percentage cover and sea temperature.

TABLE 1. Average monthly sea temperatures from the data logger on Jetty Beach.

Month	Average sea temperature (°C)
September	16.0
October	14.5
November	12.3
December	10.2
January	8.3
February	7.6
March	8.0
April	9.9

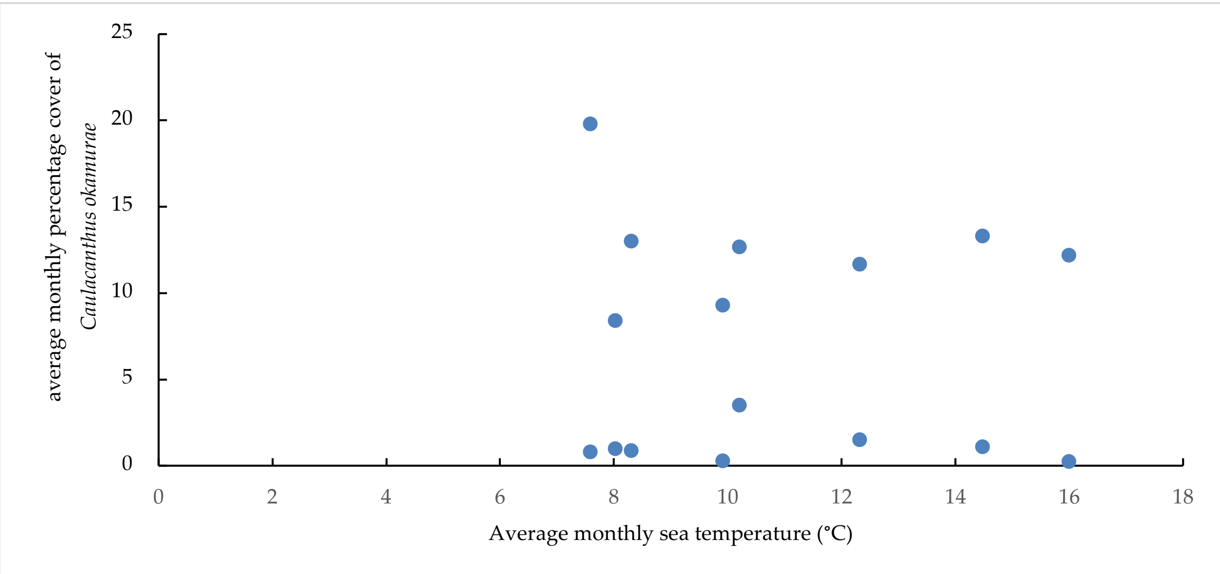


FIGURE 2: The relationship between average monthly sea temperature and percentage cover of the non-native *Caulacanthus okameurae*. Average monthly sea temperature was attained from a data logger on Jetty Beach, Dale (see Table 1). Since the percentage cover was recorded on the first of the month, the percentage cover is compared to the previous month's sea temperature.

A Spearman's Rank Correlation test was used to establish if there was a statistically significant relationship between these two variables; a p value of 0.983 was calculated, suggesting no relationship between percentage cover and sea temperature.

DISCUSSION

By investigating the percentage cover of the non-native *Caulacanthus okameurae* for 8 months, this study aimed to both map any change in the percentage cover of the species on Jetty Beach over time, and to identify any seasonal trends in its growth. No significant difference was shown between the percentage cover of *Caulacanthus okamurea* on either transect A or transect B between the ten quadrats surveyed in both October and May. Although there were fluctuations in percentage cover over the months, overall, the percentage cover has not increased, suggesting that the non-native does not appear to be colonising at an alarming rate for the time being, although continued monitoring is necessary to support this, as the species has previously colonised elsewhere very rapidly. Note also that the data were predominantly collected over the winter period which may have affected growth.

Seasonal variations are considered a major factor in affecting the dynamics of a rocky shore (Ngan and Price, 1980;). Figure 1 shows the trends in average percentage cover each month in order to denote any patterns in seasonality. Ngan and Price (1980) suggested that *Caulacanthus* species follow a pattern of active growth through the winter and then reproduction in the spring; described as the autumn-winter-spring growth period. This study does initially appear to support this trend; however, it would need to be extended for a longer period of time in order to give a definite conclusion in support of this pattern – as an observed decline in only April and May is not clear enough to suggest definite seasonality. Studies of seasonality such as the work of Ngan and Price, (1980) often run for over 20 months, to give an accurate representation, including multiple seasonal cycles. This study was time limited as a necessary consequence of my 'placement' time at FSC Dale Fort.

The sea temperature recorded by the data logger on Jetty Beach allows for comparison to the percentage cover of the *Caulacanthus okamurea;* however, in this study, no relationship was found between percentage cover and sea temperature. It has been extensively described in scientific literature that the temperature of seawater can greatly influence growth and reproduction in seaweeds, which can in turn affect their geographical distribution (Choi *et al*. 2001). Existence at the edge of temperature boundaries can often result in sterile plants (Luning, 1990; Orfanidis *et al*., 1999; Choi *et al*., 2001), but often red seaweeds are able to maintain their species' population by means of vegetative propagation (Rueness, 1997; Choi *et al*., 2001), which may begin to suggest how *Calacanthus okamurae* is able to survive in Wales, despite the low sea temperature. Typically, *Caulacanthus* species (such as *ustulatus* and *okamurae*) inhabit exposed rocky shores in Korea, China and Japan, yet in all cases fertile plants are rarely found in the field. Choi *et al*., (2001) showed that this sterility is largely a consequence of storm damage. As *Caulacanthus okamurae* is mostly found at the upper edge of the inhabitable region of the rocky shore, growth and reproduction are limited more than for other seaweeds (Kang, 1966; Lee and Lee, 1981; Choi *et al*., 2001) as this habitat has reduced available resources (Choi *et al.*, 2001). Traits such as storm resistance and being able to survive on limited resources in hostile areas enable the survival of non-natives making them able to thrive where natives may struggle.

On Jetty Beach, *Caulacanthus okamurae* appeared to show better growth on the vertical side of the artificial structure (the Victorian jetty), as opposed to on the rocks and in the rock pools. A statistically significant difference in the percentage cover was shown in these differing habitats. Artificial structures, including jetties, breakwaters, sea walls and floating pontoons, are now common substrata on coastlines (Bacchiocchi and Airoldi, 2003; Bulleri and Chapman, 2004; Bulleri and Airoldi, 2005) and these structures often support non-native species (Holloway and Keough, 2002; Lambert and Lambert, 2003; Bulleri and Airoldi, 2005). Bulleri and Airoldi (2005) investigated colonisation of artificial structures by the seaweed *Codium;* they describe how artificial structures located in the study area allow the non-native to create corridors for dispersal in areas that would be considered inhospitable. Brzana and Janas (2016) showed that out of 327 non-natives in North America, 71% of them settle on these kinds of hard substrate. This is also supported by the work of Glasby *et al*., (2006), who show that artificial structures provide unusual habitats which are often poorly utilised by native species. The rapid colonisation of artificial structures adds to the argument for the need for close and consistent monitoring of invasive species, as the imminent threat of rising sea levels and more frequent storms is likely to lead to the proliferation of coastal defences. This consequence of global change is one of many factors likely to facilitate the spread of non-natives, which could greatly affect the biodiversity of our coastlines. This is another example of human influences affecting the spread of non-natives.

Monitoring of non-natives is vital, but marine ecosystems have often been overlooked, especially in terms of quantifying the effects the non-natives. Review by Schaffelke, Hewitt and Smith (2006) suggested that only around 6% of exotic seaweeds had been studied to determine their ecological impact. *Caulacanthus* ssp. were shown to be invasive in the USA by Smith *et al.* (2014) leading to significant disruption to the intertidal zone, yet in the UK they are currently only considered non-natives. This competitive advantage is conveyed by the seaweed's ability to create a tuft structure, which has the capacity to creep over and outcompete other species on flat surfaces. This may explain *C. okamurae's* preference for the jetty. The tuft can actually increase the complexity of the habitat with its ability to trap sediments and maintain moisture. Although displacing other occupants, this behaviour may eventually lead to an overall increase in diversity (Smith *et al*., 2014). This effect could become more widespread with the predicted rise in prevalence of artificial coastal defences.

Long-term monitoring of non-natives needs to be quantified with their impact on the surrounding ecosystem. This study would have been improved if other species had also been monitored to establish relationships between them on Jetty beach. It was noticed that *Caulacanthus okamurae* tended to associate with pepper dulse (*Osmundea pinnatifida)*; however, the overall effect on biodiversity was not studied. There is definitely scope for this kind of investigation, especially considering the work of Smith *et al*. (2014), to see if overall the disturbance is affecting biodiversity positively or negatively.

Furthermore, it could be suggested a more accurate measure than observed percentage cover would be increase the value of this data. This could be achieved, for example, by the use of software such as ImageJ, which uses photographs to give more accurate measurement of percentage cover, as opposed to percentage cover judged by a researcher. This being said, in order to evaluate the perceived percentage cover, the survey methodology was repeated by students of the FSC Dale Fort Marine Science Camp 2018 whose data were not statistically significantly different from the researcher for that day, which suggests a reasonable level of accuracy was being attained in the methodology of this study.

Although no significant change in the non-natives of Jetty Beach were shown in this study, this must come with the caveat that monitoring studies are often far longer term – typically a minimum of 20 months. As a result, it cannot be thought that further monitoring is not necessary, as there is clearly a presence of non-natives at this location which could in time have a huge impact on the ecology of this area. Although important in terms of research, and in raising the profile of non-natives in this area, this study can only really be considered a pilot in monitoring on Jetty Beach which would need to be continued and extended to fully monitor the spread of these non-natives, to give indications of seasonal growth patterns in these species and to quantify the impact on the biodiversity in the area.

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REFERENCES

Allendorf, F. and Lundquist, L. (2003). Introduction: Population Biology, Evolution, and Control of Invasive Species. *Conservation Biology*, **17**(1), 24-30.

Bacchiocchi, F. and Airoldi, L. (2003) Distribution and dynamics of epibiota on hard structures for coastal protection. *Estuarine Coastal and Shelf Science*, **56**, 1157–1166

Bax, Geeves, Aguero, M., W., N. Gonzalez, E., and Williamson, A. (2003). Marine invasive alien species: a threat to global biodiversity. *Marine Policy*, (4), 313-323.

Britton-Simmons, K. (2004). Direct and indirect effects of the introduced alga *Sargassum muticum* on benthic, subtidal communities of Washington State, USA. *Marine Ecology Progress Series*, **277**, 61-78.

Brown, P., Estoup, A., Jeffries, D., Lawson Handley, L., Lombaert, E., and Thomas, C. (2011). The global spread of *Harmonia axyridis* (Coleoptera: Coccinellidae): distribution, dispersal and routes of invasion. *BioControl*, **56**(4), 623-641.

Brzana, R. and Janas, U. (2016). Artificial hard substrate as a habitat for hard bottom benthic assemblages in the southern part of the Baltic Sea – a preliminary study. *Oceanological and Hydrobiological Studies*, **45**(1), 121-130.

Bulleri, F and Airoldi, L. (2005). Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile ssp. tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, **42**(6), 1063-1072.

Bulleri, F. and Chapman, M.G. (2004) Intertidal assemblageson artificial and natural habitats in marinas on the north-west coast of Italy. *Marine Biology*, **145**, 381–391.

Centre for Biological Diversity website (2018). *Glossary of Terms*. Available at: https://www.cbd.int/invasive/terms.shtml [Accessed 22 Jan. 2018].

Choi, H. and Nam, K. (2001). Growth, Tetrasporogenesis, and Life History in Culture of *Caulacanthus okamurae* (Gigartinales, Rhodophyta) from Korea. *Botanica Marina*, **44**(4), 315-320.

Choi, H., Nam, K. and Norton, T. (2001). No whirlwind romance: typhoons, temperature and the failure of reproduction in *Caulacanthus okamurae* (Gigartinales, Rhodophyta). *European Journal of Phycology*, **36**(4), 353-358.

Colautti, R. and MacIsaac, H. (2004). A neutral terminology to define 'invasive' species. *Diversity and Distributions*, **10**(2), 135-141.

Craig, M. (2010). Pattern versus process: broadening the view of marine invasive species. *Marine Biology*, **157**(10), 2127-2128.

Glasby, T., Connell, S., Hewitt, C., and Holloway, M. (2006). Nonindigenous biota on artificial structures: could habitat creation facilitate biological invasions? *Marine Biology*, **151**(3), 887-895

Holloway, M.G. and Keough, M.J. (2002) An introduced poly-chaete affects recruitment and larval abundance of sessile invertebrates. *Ecological Applications*, **12**, 1803–1823.

Kang, J.W. (1966). On the geographical distribution of marine algae in Korea. *Bull Pusan Fisheries Coll*, **7**, 1-125

Kraan, S. (2007). *Sargassum muticum* (Yendo) Fensholt in Ireland: an invasive species on the move. *Journal of Applied Phycology*, **20**(5), 825-832.

Lambert, C.C. and Lambert, G. (2003) Persistence and differential distribution of nonindigenous ascidians in harbors of the Southern California Bight. *Marine Ecology Progress Series*, **259**, 145–161

Lee, C. (2002). Evolutionary genetics of invasive species. *Trends in Ecology and Evolution*, **17**(8), 386-391.

Lee, H.B. and Lee, I.K. (1981). Flora of benthic marine algae in Gyeonggi Bay, western coast of Korea. *Korean Journal of Botany*, **24**, 107-138.

Luning, K. (1990). Seaweeds : Their Environment, Biogeography, and Ecophysiology. Wiley, New York.
McNeill, G., Minchin, D., and Nunn, J. (2010). The slipper limpet *Crepidula fornicata* Linnaeus, 1758 becomes established in Ireland. *Aquatic Invasions*, **5**(Supplement 1), S21-S25.
MacNeil, C., Boets, P., Goethals, P., Lock, K. (2012). Potential effects of the invasive 'killer shrimp' (*Dikerogammarus villosus*) on macroinvertebrate assemblages and biomonitoring indices. *Freshwater Biology*, **58**(1), 171-182.
Meyerson, L. and Mooney, H. (2007). Invasive alien species in an era of globalization. *Frontiers in Ecology and the Environment*, **5**(4), 199-208.
Ngan, Y. and Price, I. (1980). Seasonal growth and reproduction of intertidal algae in the townsville region (Queensland, Australia). *Aquatic Botany*, **9**, 117-134.
Occhipinti-Ambrogi, A. (2007). Global change and marine communities: Alien species and climate change. *Marine Pollution Bulletin*, **55**(7-9), 342-352.
Orfanidis, S., Breeman, A.M., and Venekamp, L.A.H. (1999). Ecophysiological adaptations of two Mediterranean red algae in relation to distribution. *European Journal of Phycology*, **34**, 469-476.
Pancucci-Papadopoulou, Corsini-Foka, M., M., Politou, C. and Zenetos, A. (2005). Update of marine alien species in Hellenic waters. *Mediterranean Marine Science*, **6**(2), 147.
Partnership, W. (2018). *Wales Biodiversity Partnership - Ecosystems and Species Expert Groups*. Available at: http://biodiversitywales.org.uk/Ecosystems-Species-Expert-Groups [Accessed 24 Mar. 2018].
Pérez-López, P., Balboa, E., Domínguez, H., Feijoo, G., González-García, S., Moreira, M. (2014). Comparative environmental assessment of valorization strategies of the invasive macroalgae *Sargassum muticum*. *Bioresource Technology*, **161**, 137-148.
Raak-van den Berg, C., Hemerik, L., de Jong, P., van Lenteren, J., and van der Werf, W. (2017). Life history of the harlequin ladybird, *Harmonia axyridis*: a global meta-analysis. *BioControl*, **62**(3), 283-296
Ricciardi, A. (2016). Tracking marine alien species by ship movements. *Proceedings of the National Academy of Sciences*, **113**(20), 5470-5471.
Roy, H, E., Adams, L., Bishop, J., Bacon, J., Beckmann, B., Harrower, A, C., Hill, M, O., Issac, N, J, B., Jukes, A, R., Marchant, J, H., Musgrove, A., Noble, D., Pearman, D., Preston, C, D., Rathod, B., S, L, Rorke., Seeley, B., Sewell, J., Sweet, N., and Walker, K, J. (2012). Non-Native Species in Great Britain: establishment, detection and reporting to inform effective decision making. Available at: http://file:///C:/Users/User/Downloads/2012_-_NNSIP_Final_report.pdf [Accessed 22 Mar. 2018].
Rueness, J. (1997). A culture study of *Caulacanthus ustulatus* (Caulacanthaceae, Gigartinales, Rhodophyta) from Europe and Asia. *Cryptogamie, Algologie*, **18**, 175-185.
Rueness, E., and Rueness, J. (2000). *Caulacanthus ustulatus* (Gigartinales, Rhodophyta) from Brittany (France) is an introduction from the Pacific Ocean. *Cryptogamie Algologie*, **21**(4), 355-363.
Schaffelke, B., Hewitt, C., and Smith, J. (2006). Introduced Macroalgae – a Growing Concern. *Journal of Applied Phycology*, **18**(3-5), 529-541.
Smith, J., Vogt, S., Creedon, F., Lucas, B. and Eernisse, D. (2014). The non-native turf-forming alga *Caulacanthus ustulatus* displaces space-occupants but increases diversity. *Biological Invasions*, **16**(10), 2195-2208.
Steffen, W. and Walker, B. (1997). An Overview of the Implications of Global Change for Natural and Managed Terrestrial Ecosystems. *Conservation Ecology*, **1**(2).
Zenetos, A. Çinar, M., Crocetta, F., Golani, D., Rosso, A., Servello, G., Shenkar, N., Turon, X., and Verlaque, M., (2017). Uncertainties and validation of alien species catalogues: The Mediterranean as an example. *Estuarine, Coastal and Shelf Science*, **191**, 171-187.

APPENDIX

Table 1: The average percentage cover of the non-native seaweed *Caulacanthus okameurae* from 10, 0.25m^2 quadrats, over three transects on Jetty beach, Dale. The data was collected monthly for 8 months, from October 2017 to May 2018. The three transects are orientated perpendicular to the sea, covering a distance of 10 meters. They are ordered A, B and C from left to right of the shore (when facing upshore; see Figure 1).

	Average percentage cover of *C. okameura* (%)		
Month	A	B	C
October	12.2	0.0	0.25
November	13.3	0.0	1.1
December	11.7	0.0	1.5
January	12.7	0.0	3.5
February	13	0.0	0.9
March	19.8	0.0	0.8
April	8.4	0.0	1
May	9.3	0.0	0.3