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Dropping plates to pick up aliens: towards a standardised approach for monitoring alien fouling species

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Biological invasions pose a major threat to biodiversity and significant investment is required to prevent the introduction of alien species, contain introduced populations and mitigate associated impacts. The implementation of standardised long-term monitoring programmes enables the early identification of new alien species, the tracking of spread, the assessment of the effectiveness of management interventions, and an understanding of temporal and spatial trends. Globally, fouling species are known to cause economic losses through accumulation on vessel hulls and port infrastructure and via the disruption of aquaculture activities. Additionally, fouling taxa can have ecological impacts in recipient systems, most often driven by their dominance in biological interactions. In South Africa, this group accounts for the majority of marine alien taxa. Accordingly, this study tested an approach for monitoring fouling biota using open and caged PVC settlement plates deployed in marinas. After 16 weeks, plates were removed, biota identified, and relative abundance captured as percentage cover. This method proved to be effective, logistically simple and cost-efficient. Twenty-four taxa were recorded, of which 58% were invasive species. It is recommended that this monitoring approach be implemented around the South African coast as a first step to providing key information to inform management and research.

Keywords: early detection, invasive species, long-term monitoring, management, marine bioinvasions, South African marinas, surveillance

Introduction

Established alien species are an important driver of global ecological change, as well as an economic, social and cultural problem (Simberloff et al. 2013; Capinha et al. 2015). From the loss of native species following invasions by novel taxa, to traditional communities experiencing the loss of important cultural resources, the issue of biological invasions is a predicament for conservationists, policymakers and society at large (Bellard et al. 2016; Rogers et al. 2017). As such, significant financial investments are required to support the prevention of invasions and the remediation of impacted systems (Vilà et al. 2010; Pyšek et al. 2020). The loss of ecosystem services and the resultant economic impacts that can be associated with invasions are especially alarming for a continent such as Africa, where high levels of poverty and direct dependence on ecosystem services for livelihoods are common (Shackleton et al. 2019).

Considering the impacts on natural ecosystems, the human interests often associated with alien species, and the increased number of emergent invasive alien species forecast for the future (Seebens et al. 2018), the Convention on Biological Diversity (CBD) and the associated Aichi Target 9 require countries to identify alien species and their pathways of introduction, control and eradicate priority species, and manage pathways to prevent the introduction and establishment of alien species (CBD 2018). As the management of alien species has a higher chance of success and lower cost when implemented in the early stages of invasion (Lodge et al. 2016; Reaser et al. 2020), the early detection of and rapid response to bioinvasions (EDRR) is crucial to meeting states' commitments under Aichi Target 9.

A primary requirement for achieving EDRR is the implementation of long-term, standardised surveillance programmes, especially in high-risk areas (Pyšek et al. 2020). Such surveillance can be achieved through monitoring by using a set of consistent methods to record predefined information about species occurrence, distribution and impact, through time and across different geographic scales. Additionally, the prioritisation of species, sites and associated pathways as targets for management is critical to achieving effective policy implementation and favourable management outcomes (McGeoch et al. 2016), and the early detection of introduced species supports such prioritisation.

Long-term monitoring programmes are important not only because of their surveillance value but also because they enable comparisons through time. This can give information on introduction history and spread, while simultaneously assessing the effectiveness of policy interventions and biosecurity measures (Essl et al. 2020). The adoption of long-term and large-scale spatial surveillance is especially important considering the many other agents of global change (e.g. climate change and pollution) and their potential link to the introduction of alien species (Pyšek et al. 2020; Robinson et al. 2020a). Accordingly, the need for accurate information on the distribution of alien species is critical at local, regional and global scales (Latombe et al. 2017). Although a coordinated approach for monitoring biological invasions at these spatial scales is urgent, few countries have national surveillance and monitoring programmes in place (Latombe et al. 2017). The absence of such programmes in many regions likely reflects imbalances among countries in terms of the human and economic resources required to sustain such activities. Evans and Blackburn (2020) demonstrated the implications of such disparities in terms of understanding the impact of alien birds among 247 regions. Notably, they found that the availability of data was related primarily to economic status. Similarly, Measey et al. (2020) demonstrated that, for alien amphibians, the ability to detect high-level impacts was inherently linked to study costs. Similarly, there is a direct relationship between economic development and nations' governance response to bioinvasions, a situation that is particularly challenging in Africa and many Asian countries (Early et al. 2016; Turbelin et al. 2017). This highlights that the development of low-cost and easy-to-use monitoring methods is crucial for fostering broadscale monitoring by countries with constrained resources and for meeting the goal of global monitoring of biological invasions (Latombe et al. 2017).

More than 2 100 marine species are currently listed on the World Register of Introduced Marine Species (WRiMS, http://www.marinespecies.org/introduced/) as having been introduced outside of their native ranges (Ahyong et al. 2021). Fouling organisms are those that settle on submerged hard substrata. Consequently, these taxa are commonly associated with seagoing vessels and harbour infrastructure. In South Africa, 95 marine alien species are known to occur, of which 42% are alien fouling organisms (Robinson et al. 2020b). Notably, shipping is responsible for 91% of marine introductions to South Africa (Robinson et al. 2020b), resulting in harbours and marinas being both recipients and sources of alien fouling taxa. Although fouling biota occur predominantly in anthropogenic environments (Ruiz et al. 2011), some are able to spread to natural habitats (Mack et al. 2000). Here, they can threaten marine biodiversity and have negative consequences for industries that are reliant on marine systems (e.g. aquaculture and tourism) (Vilà et al. 2010). The sea vase ascidians *Ciona intestinalis* and *C. robusta*, for example, are widely distributed alien taxa and frequently become dominant in recipient communities (Blum et al. 2007; Rius et al. 2011). The most conspicuous negative impacts of alien fouling biota to industry are those related to the aquaculture sector. The presence of fouling on mariculture farms, for example, can inhibit shellfish growth and increase maintenance costs (Rius et al. 2011; Lins and Rocha 2020). It has been estimated that the management of biofouling costs accounts for 5–10% of the production expenses incurred by the aquaculture industry and results in global costs of up to $3 billion (Fitridge et al. 2012). However, many of the indirect costs remain unassessed (Fitridge et al. 2012) and it is likely that the monetary impact is greater. As such, biofouling is an important obstacle to efficient and sustainable mariculture production (Bannister et al. 2019).

Despite the 40 alien fouling species already known in South African waters, there are several impactful species that are not yet present, but that could be introduced. These include the carpet sea squirt *Didemnum vexillum* and the Asian date mussel *Arcuatula senhousia*. *Didemnum vexillum* is native to the northwestern Pacific (Lambert 2009) and has an extensive invasive range in temperate regions (Stefaniak 2017). Within its invasive range this colonial ascidian becomes spatially dominant and is known to exclude native taxa (Valentine et al. 2007) and negatively affect mariculture operations (Lambert 2009; Ordóñez et al. 2015). In turn, the Asian date mussel is associated with harmful ecological and economic impacts in coastal areas of North America, Oceania and the Mediterranean, and has recently been recorded in West Africa (Bachelet et al. 2009; Lourenço et al. 2018). The vulnerability of South Africa to the introduction of fouling taxa was highlighted by Lins et al. (2018), who, when modelling the global risk of ascidian invasions, identified the southeastern Atlantic, including South Africa's west coast, as the region in the Southern Hemisphere most at risk to new ascidian invasions.

Considering the high economic, ecological and social impacts associated with fouling invasions and the predominance of hull fouling as a vector of marine alien taxa, there is great value in establishing a countrywide monitoring programme that could aid in early detection of such species and support rapid management responses. Additionally, current data on the occurrence of alien fouling species in South Africa are fragmented, frequently outdated and often unstandardised. This hampers spatial and temporal comparisons that could contribute to a better understanding of distribution patterns and abundance of these species. In recognition of this need and the high costs associated with some monitoring techniques (e.g. eDNA [Rius et al. 2015] and remotely operated vehicles [Peters et al. 2019a]) this study aimed to test a standardised and simple method using settlement plates to detect alien fouling species and track their distribution and abundance. Additionally, this method was contrasted against the use of scrape samples to assess efficacy and cost-efficiency. This information is used to propose a national monitoring programme for fouling species.

Materials and methods

Study sites

To evaluate the proposed monitoring approach in areas with distinct biotic and abiotic characteristics, four marinas spanning the temperature gradient around the South African coast and representative of the three dominant ecoregions (as defined by Sink et al. 2012) were chosen as study sites (Figure 1). Yachtport SA marina in Saldanha Bay (33°01'34" S, 17°57'39" E) was the most westerly site, falling within the cool-temperate Benguela ecoregion. False Bay Yacht Club (34°11'33" S, 18°26'06" E) was the most westerly of the two sites within the warmer Agulhas

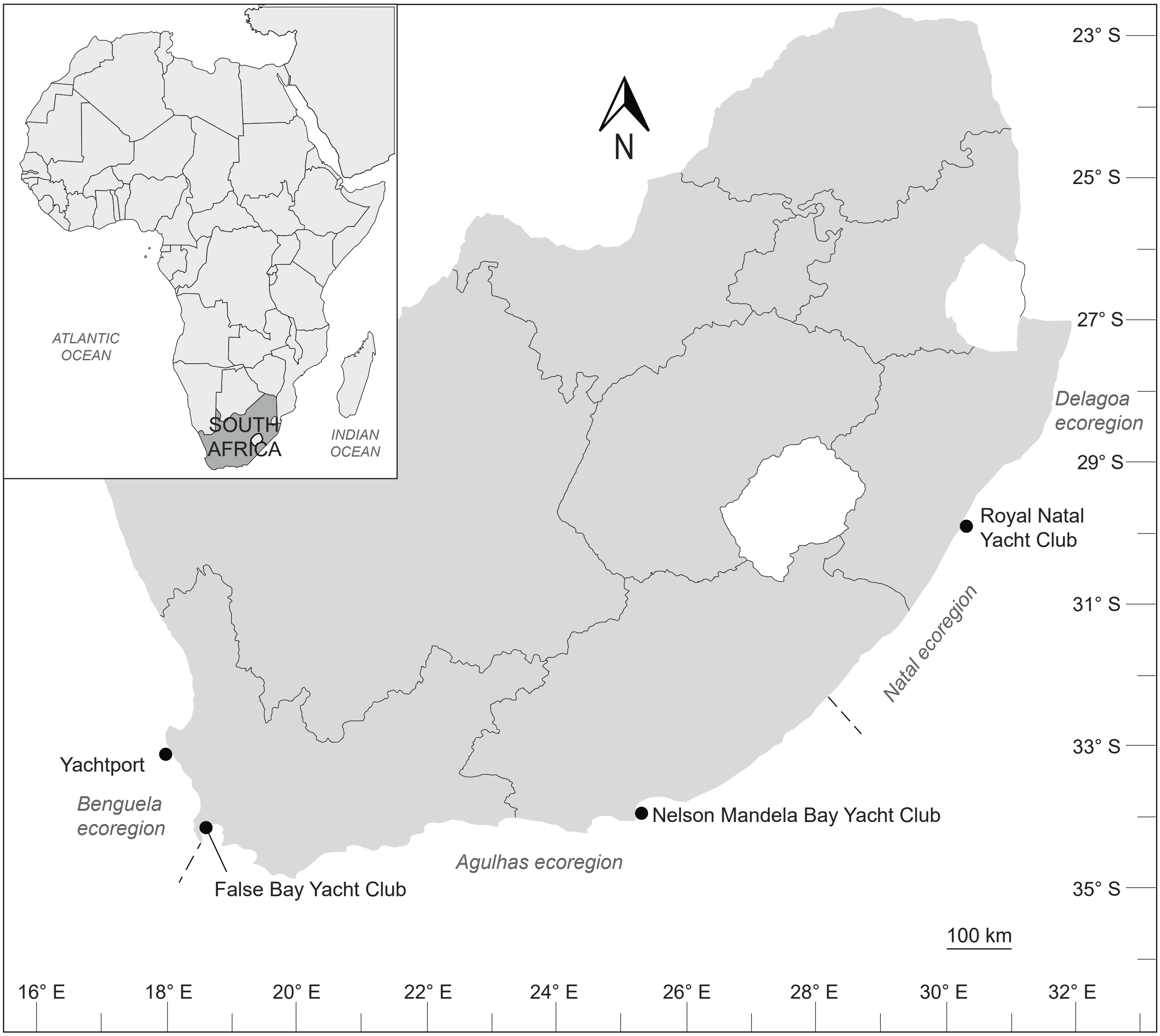

Figure 1: Map of the four study sites along the South African coast where settlement plate arrays were deployed to monitor marine fouling biota

ecoregion, followed by Nelson Mandela Bay Yacht Club (33°58′02″ S, 25°38′01″ E) in Port Elizabeth (now Gqeberha). Lastly, Royal Natal Yacht Club (29°51′47″ S, 31°01′14″ E) in Durban represented the subtropical Natal ecoregion. The marinas are all located within important ports and are thus potentially exposed to propagules from both international and local shipping traffic as well as recreational yachts.

Design of monitoring arrays

To achieve a standardised sampling method that could be used at sites offering different infrastructure and to provide an unoccupied habitat for fouling settlement, PVC settlement plates of 100 cm^2 were used. Prior to deployment, plates were sanded to provide a rough surface to facilitate attachment by sessile taxa. Plates were also soaked in water for 10 days, to pre-leach chemicals that might affect settlement. Two array designs were trialled (Figure 2). Both arrays had the same basic design and consisted of two settlement plates fixed perpendicular to a rope and separated by 1 m. As arrays were suspended from floating jetties, the top plate was always at 1-m depth while the bottom plate was at 2-m depth. All arrays were weighted to stabilise the plates and prevent the ropes from posing a safety risk to vessels. The first design saw arrays deployed as described above, while in the second design plates were placed in cages made of 5-mm plastic mesh. These cages prevented any influence of macropredators on recruitment.

At each site, five arrays of each design were installed randomly, though a minimum distance of 2 m was maintained between arrays. Preliminary analyses (i.e. as described below but including depth as a predictor) identified

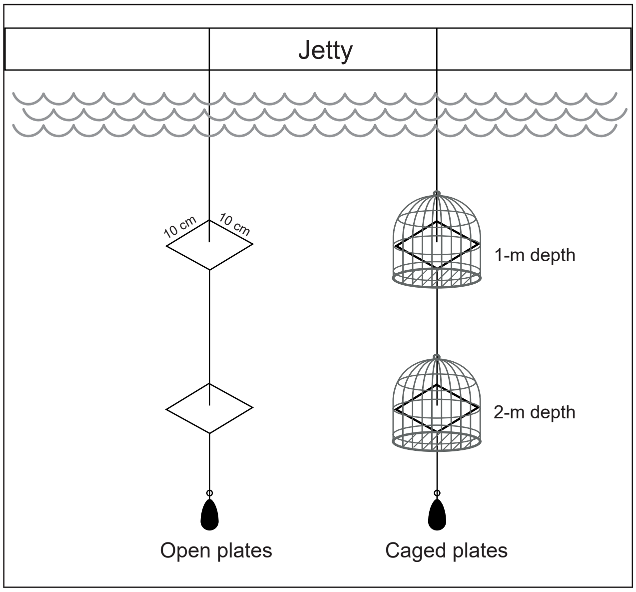

Figure 2: Schematic illustration of the setup used to monitor fouling biota using open plates (accessible to predators) and caged plates (inaccessible to predators). Note: not drawn to scale

no effect of depth on any of the measures considered ($p > 0.05$ in all cases), resulting in 10 open and 10 caged plates per site. Arrays were installed at all sites over a period of 7 days, ensuring a deployment that was as close to simultaneous as was practically possible. After 16 weeks, the arrays were removed from the water. This period was chosen as twice-weekly inspections of the plates revealed that by this time plates supported 70–100% cover and individuals that had settled were predominantly large enough to facilitate visual identification.

Upon removal from the water, the underside of each plate was photographed and specimens whose identification was uncertain and/or required microscopic analysis for confirmation were preserved and returned to the laboratory. The specimens were identified using the works of Montagu (1814), Darwin (1854), Oka (1927), Monniot et al. (2001), Smith and Gordon (2011), Rocha et al. (2012) and Simon et al. (2019). We were unable to identify two serpulid worms to species level, but they were treated as morpho-species in the community analysis as they were morphologically distinct. Taxa were classified as native, alien or invasive, following Robinson et al. (2016). The photographic images from each plate were used to quantify species abundance as percentage cover using ImageJ2 software (Rueden et al. 2017).

Statistical analysis

To assess the effectiveness of the two array designs (open and caged) in detecting species at the different sites, the number of species recorded were compared between designs and among sites using a generalised linear model underpinned by a Poisson distribution. Species accumulation curves were plotted using the UGE index (Ugland et al. 2003) to assess the adequacy of the number of plates used to detect species per array design in each site. All univariate statistics were conducted in R 4.0 (R Core Team 2020).

To assess whether the suite of species detected differed between designs and among sites, community structure was compared using a PERMANOVA. Percentage cover was square-root-transformed to decrease the influence of the dominant species (Clarke and Warwick 2001). For the multivariate model, array design was nested in site owing to the *a priori* recognition of ecological differences among ecoregions and both factors were treated as fixed factors. To identify which species were responsible for the differences identified by the PERMANOVA, a SIMPER analysis was used. The Primer software package (version 6.1.16) was used for all multivariate analyses.

Cost of monitoring

The costs associated with monitoring were calculated as monetary costs for constructing the arrays and the staff time associated with carrying out the work. While the material costs of constructing five arrays of each design were easily calculated in monetary terms (South African Rand [R] at 2020 levels, where R1 = US$0.073), the process of deployment and removal of arrays, photography of plates, species identification and quantification of abundance using the photographs (as described above) was quantified in terms of time (hours) and converted to monetary terms assuming an hourly rate of R150. This is equivalent to salary rates advertised by the South African Department of Forestry, Fisheries and the Environment (DFFE) in 2020 for graduates with a National Diploma or Bachelor's degree and 2–3 years of experience. Nonetheless, it is anticipated that this role would be filled by existing salaried employees of DFFE, rather than a new appointment. The benefit of this approach is that the individuals concerned could be trained at the start of the programme and their skills would be retained within the department. This is preferable over short-term appointments which would require constant retraining of staff and potentially introduce a source of variability into the resulting data. Importantly, the people involved with identification of taxa would require specialised training and would need to work closely with taxonomic experts, especially when confirmation of species identification is needed. The costs associated with this aspect were based on a 2-day workshop led by taxonomic experts. Such workshops would be held every second year to train new staff and share updated taxonomic information with those already experienced in the programme. Additionally, costs were allocated for covering the time of experts needing to confirm difficult species identifications. The cost of genetic confirmation of one new species identification per year was included, but there is great uncertainty around this figure. This is because these costs vary considerably among taxa. Importantly, if the programme works closely with academics in the field, these costs could be shared if an agreement is set up to facilitate the publishing of these results. An additional cost that could not be accounted for was the cost of travel. This is because it would vary depending on the site and where the officials are based. However, as major ports occur near the offices of DFFE these costs are not expected to be high. To assess the cost of a long-term monitoring programme instead of a once-off event, the total cost was calculated over a 4-year period assuming three deployments/retrievals per annum. All costs were calculated for one site so that the cost implications could easily be scaled up depending on the number of sites to be included in a national programme.

Comparisons with an alternate method

To assess the efficacy and cost-efficiency of the proposed method, the number of species detected, and the associated costs were compared with the use of scrape samples. The scrape-sample data are fully described in Peters et al. (2017) but represent 225 cm^2 quadrats that were scraped from harbour infrastructure by divers and then sorted in the laboratory. These samples were collected in the same locations as the current study. As in the current study, 10 scrape samples were collected at each site. Additionally, both studies were conducted in austral winter, although Peters et al. (2017) did sample a few harbours in spring.

Results

Despite 10 plates (i.e. five arrays) of each design being deployed at each site, a few caged plates were lost in all instances. The numbers of lost plates ranged from one in Port Elizabeth to four in Durban. Notably no uncaged plates were lost.

A total of 24 taxa were recorded across all sites and designs, of which 58% were invasive species (Table 1). Although two taxa could not be identified to species level, there was no reason to suspect that they might be alien. The highest number of species was recorded in False Bay (20 species) and the fewest were recorded in Durban (13 species). The number of species detected differed significantly among sites (GLM, $\chi^2 = 58.368$, $p < 0.001$) and between array designs ($\chi^2 = 13.416$, $p < 0.01$) (Figure 3), with no interaction between these factors ($\chi^2 = 1.661$, $p = 0.64$). Pairwise comparisons revealed that the only sites that did not differ in the number of species recorded were Port Elizabeth and Durban ($p = 0.75$), both of which supported fewer species than the sites to the west. In all sites, caged plates supported more species than open plates ($p < 0.01$ in all cases). Notably, some species were restricted to only one array design. The polychaetes *Ficopomatus enigmaticus* and *Sabella spallanzanii*, together with the barnacle *Balanus trigonus*, the ascidian *Microcosmus squamiger* and the sponge *Hymeniacidon perlevis*, occurred uniquely on caged plates. In contrast, the bryozoans *Bugulina flabellata* and *Watersipora subtorquata*, the barnacle *Amphibalanus amphitrite* and the ascidian *Clavelina lepadiformis* were recorded only on open plates. The number of settlement

Table 1: Species recorded on caged (C) and open (O) settlement plates in Saldanha Bay (SB), False Bay (FB), Port Elizabeth (PE) and Durban (DBN), South Africa. Status of species assigned following Robinson et al. (2016)

Taxa	Status	SB	FB	PE	DBN
PORIFERA					
Hymeniacidon perlevis	Native		C	C	
ANNELIDA					
Polychaeta					
Neodexiospira brasiliensis	Invasive	C	C	C, O	C
Ficopomatus enigmaticus	Invasive				C
Sabella spallanzanii	Native				C
Serpulidae 1	–	C, O	C	C	O
Serpulidae 2	–	C, O	C	C, O	C, O
Spirorbis sp.	Native	C	C, O	C, O	C, O
CRUSTACEA					
Cirripedia					
Amphibalanus amphitrite	Native	O	O	O	O
Balanus trigonus	Invasive	C	C		
Notomegabalanus algicola	Native	C, O	O		
BRYOZOA					
Bugula neritina	Invasive	C, O	C, O	C, O	C, O
Bugulina flabellata	Invasive	O	O	O	
Watersipora subtorquata	Invasive	O	O		O
CHORDATA					
Ascidiacea					
Ascidiella aspersa	Invasive	C, O	C	C	
Asterocarpa humilis	Invasive	C, O	C, O	C	
Botrylloides magnicoecus	Native	C, O	C, O		
Botryllus gregalis	Native	C, O	C, O	C	
Botryllus schlosseri	Invasive	C, O	C, O	C, O	
Diplosoma listerianum	Invasive	C, O	C, O	C, O	C, O
Ciona robusta	Invasive	C, O	C, O	C, O	C
Clavelina lepadiformis	Invasive	O	O	O	
Corella eumyota	Native	C, O	C, O	C	
Microcosmus squamiger	Invasive				C
Styela plicata	Invasive			C, O	C, O

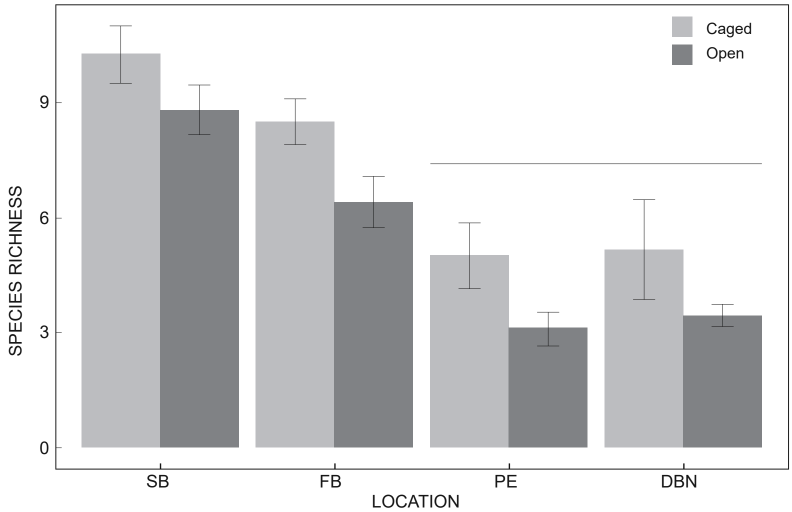

Figure 3: Mean number of species detected on caged and open settlement plates at each site: Saldanha Bay (SB), False Bay (FB), Port Elizabeth (PE), and Durban (DBN), South Africa. A line connects sites in which the number of species recorded did not differ significantly ($p > 0.05$). Error bars represent SD. At all sites the number of species detected differed significantly between array designs ($p < 0.05$)

plates used in each site was deemed sufficient as species accumulation curves for both array designs tended towards horizontal asymptotes for all sites (Figure 4).

Community composition differed significantly between sites (PERMANOVA, $F = 11.934$, $p < 0.01$) and array designs (PERMANOVA, $F = 5.868$, $p = 0.001$). *Post hoc* analyses demonstrated that, at all sites, composition differed significantly between array designs ($p < 0.05$ in all cases) (Figure 5). The SIMPER analysis identified that the significant differences between array designs within the sites were driven primarily by invasive ascidians including *Diplosoma listerianum*, *Ciona robusta*, *Ascidiella aspersa* and *Botryllus schlosseri*, as well as invasive bryozoans from the genera *Bugulina* and *Bugula* (Table 2). Notably, while all these taxa occurred on both array designs, the colonial ascidians *C. robusta* and *A. aspersa* were most prolific on caged plates, whereas the other taxa were dominant on open plates.

Comparisons of the number of alien species detected by this study and scrape samples collected by Peters et al. 2017 revealed a large degree of overlap (Table 3). The only site at which this was not observed was Port Elizabeth, where the present study recorded 10 species whereas only three species were noted in scrape samples, with no overlap in species composition.

The costs associated with monitoring for fouling species are presented in Table 4. Despite general costs (e.g. training workshops) being shared by both methods, the costs specific to the use of settlement plates and scrape samples differed markedly. This resulted in the overall costs of using plates for monitoring being less than half that of scrape samples. The vast differences between the costs of these methods stem primarily from the high costs of using divers to collect scrape samples and the fact that these samples take almost double the time to sort once back in the laboratory. When the annual costs of each method are converted to the costs of a single deployment at four sites (i.e. equivalent to the detection effort in this study and in Peters et al. [2017], at R88 600 and R224 133 for plates and scrape samples, respectively), the comparative costs per alien species detected are R6 329 for plates and R12 452 for scrape samples. This reflects the cost-efficient nature of using plate arrays for monitoring.

While the costs of using settlement plates reflect deployment of arrays in marinas or harbours, their installation is also possible on floating structures such as buoys or aquaculture rafts. However, a boat would be needed to access these structures, and the additional time required for installation will depend on the distance of such structures from the shore. To minimise vessel costs it is suggested that any monitoring using buoys or aquaculture infrastructure could be dovetailed with existing maintenance of these structures. This would require cooperation with farmers and/or ports authorities. While acknowledging uncertainties around several of the costs presented in Table 4, it does allow estimates of costs associated with a national monitoring programme. For example, a 4-year programme that includes all eight national marine ports of entry (i.e. Richards Bay, Durban, Ngqura, East London, Port Elizabeth, Mossel Bay, Cape Town and Saldanha Bay) will incur an approximate cost of R2.126 million. If monitoring was extended to include marinas in Port Owen, Hout Bay, Simon's Town, St Francis, Port Alfred and Richards Bay this would rise, but not proportionally as the staff, camera and microscope used for Richards Bay harbour could service the marina too. Similarly, this could be applied to Simon's Town and Hout Bay marinas with respect to staff and equipment servicing Table Bay, as well as Port Owen Marina and the Port of Saldanha Bay. Considering this, the total cost of 4-year programme monitoring at all eight ports of entry and the above-mentioned marinas would be R2.870 million.

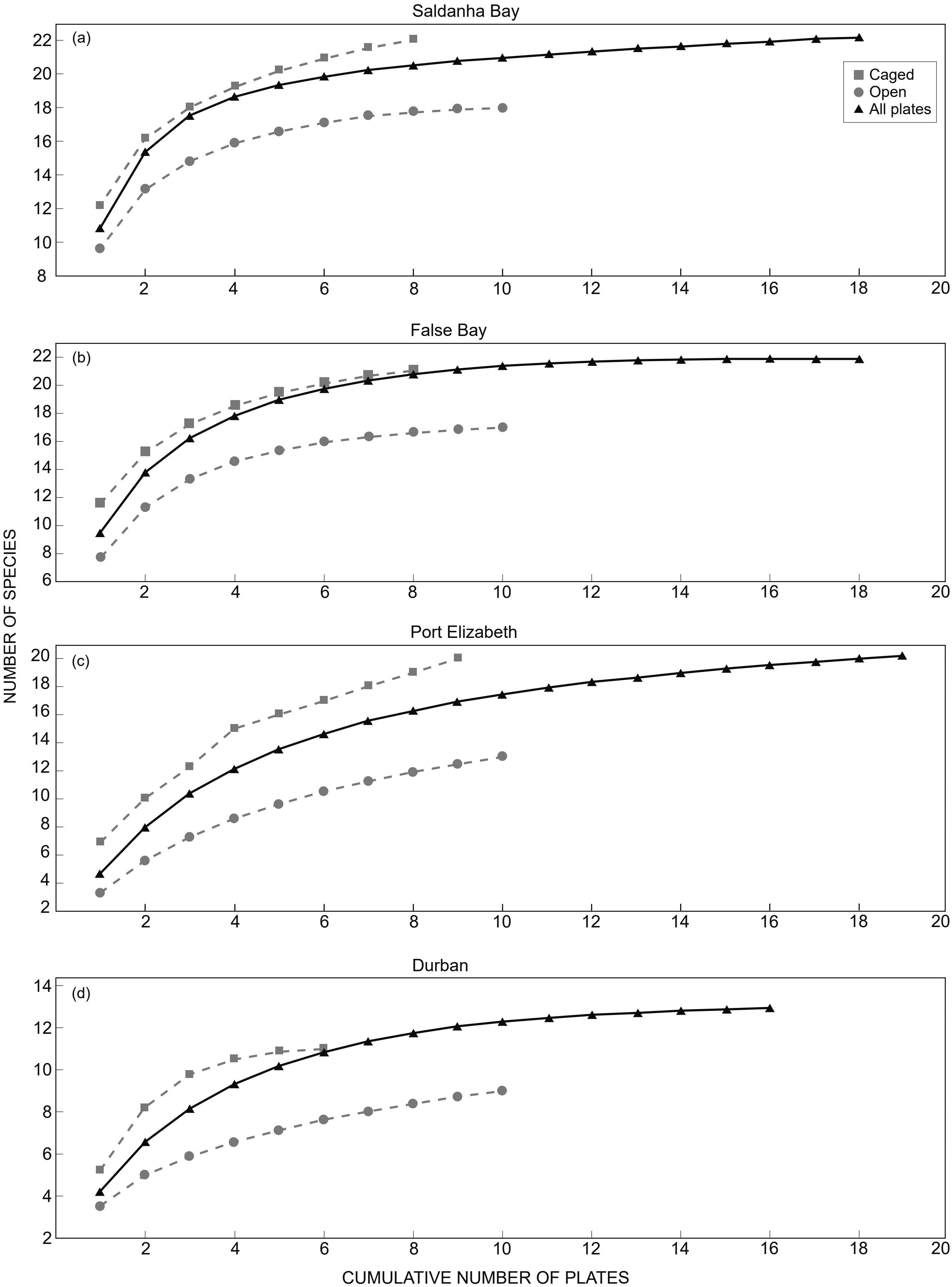

Figure 4: Species accumulation curves for each array design (caged and open settlement plates), as well as the two arrays combined, in (a) Saldanha Bay, (b) False Bay, (c) Port Elizabeth, and (d) Durban, South Africa. The curves were plotted using the UGE index calculated in Primer 6

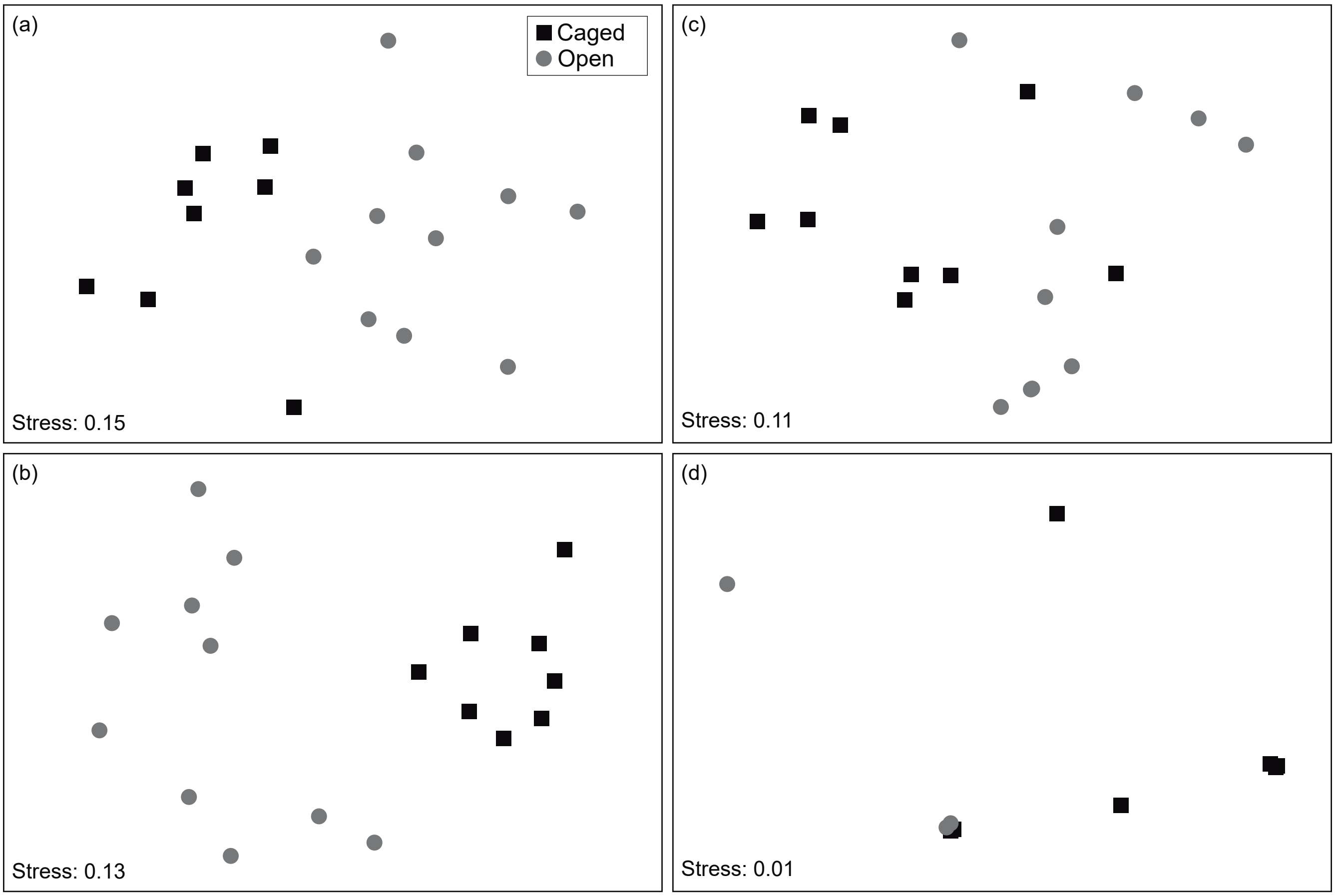

Figure 5: Two-dimensional multidimensional scaling plots based on standardised root-transformed percentage cover of fouling communities in (a) Saldanha Bay, (b) False Bay, (c) Port Elizabeth, and (d) Durban, South Africa. The different symbols represent open and closed array designs

Discussion

An important element of the effective management of biological invasions is the ability to detect alien species soon after their arrival—before high abundance is reached and spread occurs. Such early detection followed by rapid response increases the probability of successful eradication or control and decreases management costs, while concurrently minimising potential impact (Lodge et al. 2016). As such, and in acknowledgment of the need for low-cost solutions to this challenge, this study presents a standardised monitoring approach that would facilitate early detection of new incursions and track the spread of alien fouling species. The concurrent use of two different array designs was found to promote the simultaneous detection of a variety of fouling species, and thus offers a way of maximising detection efficiency while minimising costs and effort. This approach was found to be logistically simple and affordable, yet effective at multiple sites around the coast.

Logistical considerations for implementing a monitoring programme in South Africa

For a long-term programme to monitor alien fouling species to be effective, it is vital that the method used is effective at detecting the target species. The use of open and caged settlement plates in this study resulted in two very different suites of species being recorded, with communities that developed on the two array designs showing over 60% dissimilarity at all sites, and above 80% dissimilarity at Port Elizabeth. It is notable that colonial ascidians and bryozoans tended to be dominant on open plates, while in contrast solitary ascidians were most abundant on caged plates. This outcome could be driven by an interplay between predatory effects and light exposure (Loureiro et al. 2021). This ultimately results in shade-seeking species and those susceptible to predation occurring predominantly on caged plates, while those taxa that are resistant to predation and not photonegative are most dominant on open plates. Although no mobile taxa were recorded in this study, it is worth noting that while they would be excluded from caged plates, such groups would be able to access open plates. While some species were recorded on both array types, others were restricted to one array design (e.g. the invasive bryozoans *Bugulina flabellata* and *Watersipora subtorquata* occurred on open plates only, while the ascidian *Microcosmus squamiger* was recorded only on caged plates). These results highlight the need to simultaneously deploy arrays of both designs to maximise the number

Table 2: SIMPER analysis of community composition on caged and open settlement plates in Saldanha Bay, False Bay, Port Elizabeth and Durban, South Africa. The five species most responsible for the dissimilarity between array designs are listed for each site

Species	Average abundance		Contribution	Cumulative
	Caged	Open	%	%
Saldanha Bay (average dissimilarity between designs: 61.31%)				
Ciona robusta	6.06	3.34	11.62	11.62
Botryllus schlosseri	2.06	4.18	10.13	21.75
Bugulina flabellata	0	3.33	9.03	30.78
Ascidiella aspersa	3.26	0.89	8.38	39.17
Botrylloides magnicoecus	0.52	2.98	8.10	47.26
False Bay (average dissimilarity between designs: 64.52%)				
Diplosoma listerianum	1.54	6.16	14.28	14.28
Ascidiella aspersa	3.57	0	10.65	24.93
Corella eumyota	5.43	3.10	10.54	35.46
Ciona robusta	4.55	1.57	9.79	45.25
Bugulina flabellata	0	3.11	9.32	54.57
Port Elizabeth (average dissimilarity between designs: 81.14%)				
Diplosoma listerianum	2.10	4.36	19.43	19.43
Ciona robusta	3.88	0.22	19.29	38.72
Bugula neritina	0.96	1.88	11.21	49.93
Ascidiella aspersa	2.40	0	10.71	60.64
Botryllus schlosseri	0.94	1.21	7.37	68.01
Durban (average dissimilarity between designs: 64.81%)				
Diplosoma listerianum	4.06	7.74	25.33	25.33
Bugula neritina	2.68	0.14	11.71	37.03
Styela plicata	2.39	0.24	9.37	46.40
Microcosmus squamiger	2.39	0	8.88	55.28
Ficopomatus enigmaticus	1.56	0	8.52	63.80

Table 3: The comparative number of alien fouling species detected on settlement plates in this study and in scrape samples by Peters et al. (2017) in Saldanha Bay (SB), False Bay (FB), Port Elizabeth (PE) and Durban (DBN), South Africa. 'Unique' species refers to those detected by one method but not the other

Site	No. of alien species detected		No. of unique alien species detected		Percentage of shared alien species
	Plates	Scrape	Plates	Scrape	
SB	10	10	1	1	90
FB	11	15	0	4	73
PE	10	3	10	3	0
DBN	7	7	2	2	71
Total	14	18	10	5	78

and variety of species recorded. This approach could be extended in the future, by testing the effectiveness of deploying additional materials (e.g. wood or rope) along with settlement plates (Holmes and Callaway 2020). Although 10 plates were found to be effective at detecting alien fouling taxa, it is notable that caged plates were lost at all sites. This may be reflective of all ports having been exposed to significant winter storms during the deployment period. To account for potential loss during winter the deployment of additional caged plates could be considered.

While this study did record 14 alien species, this is not reflective of all fouling species known from South Africa. Of the species that were not detected, the only ones that would be expected to be detected from photographs and are known from the sites that were sampled are the brachiopod *Discinisca tenuis* (known from Saldanha Bay only), the sponge *Suberites ficus* (Saldanha Bay), the mussels *Mytilus galloprovincialis* (Saldanha Bay, False Bay and Port Elizabeth) and *Semimytilus algosus* (Saldanha Bay, False Bay), and the ascidian *Ascidia sydneiensis* (False Bay and Port Elizabeth). While the reasons for these species not being detected remain unclear, their absence could reflect the once-off nature of this study. Continuous deployments, as suggested for a monitoring programme, would offer a greater probability of detecting species as seasonality would be negated. It is worth noting that all except one species (i.e. *A. sydneiensis*) reported by Peters et al. (2017) and not detected in the present study were small-sized taxa that would require a microscope to see (e.g. the amphipod *Jassa slatteryi*).

Cost-effectiveness is a key consideration for any management intervention and reflects the benefit gained for

Table 4: Costs associated with the use of settlement plate and scrape samples for detecting marine fouling species. Costs were estimated in terms of time required from personnel (at a rate of R150 h^{-1}), material costs and costs that would be outsourced (i.e. work undertaken by personnel not employed by the Department of Forestry, Fisheries and the Environment). Annual costs account for three deployments per year. # = The costs of a taxonomy workshop would be incurred only every two years. *The cost of travelling was excluded as this will vary depending on the site in question. $ = The cost of preserving species that are detected by plates but are not easily identifiable would likely not be repeated within a 4-year programme; in contrast, this will be an annual cost when using scrape samples. @ = As a microscope and camera will only need to be bought once, these costs were divided by four to calculate an annual cost, although the contribution to annual costs would decrease the longer the programme runs. +The grand total for a 4-year monitoring programme includes general costs plus the costs specific to the use of either plates or scrape samples. Note: US$ value was calculated at an exchange rate of R1 = US$0.073

	Staff costs		Material costs	Outsourced costs	Total cost per year	Grand total per year	Grand total per 4-year programme
Source of cost	Hours	R/$	R/$	R/$	R/$	R/$	R/$
General costs							
Taxonomy workshop (2 days)	18#	2 700/196#		10 000/727#	6 350/465#		
Travel time	*	*	*		*		
Microscope			20 000/1 453@		5 000/363@		
Expert taxonomist				2500/182	2 500/182		
DNA confirmation of species identification				20 000/1 453	20 000/1 453		
						33 850/2 464	
Plates							
Construction of caged arrays (5 units)	10	1 500/109	250/18		5 250/384		
Construction of open arrays (5 units)	5	750/55	150/11		2 700/189		
Array installation (10 units)	1	150/11			450/33		
Array removal (10 units)	1	150/11			450/33		
Vials and alcohol for preservation of specimens			600/44$		150/11$		
Camera			8 000/596@		2 000/149@		
Photography of plates on removal (20 units)	8	1200/87			3 600/263		
Image analysis (species identification and assessment of abundance)	40	6 000/436			18 000/1 317	32 600/2 379	265 800/19 372+
Scrape samples							
Dive boat				15 000/1 098	45 000/3 294		
Dive team (incl. equipment)				18 000/1 317	54 000/3 952		
Collection equipment			300/22		900/66		
Vials and alcohol for preservation of specimens			600/44		600/44		
Sample sorting and species identification	75	11 250/823			33 750/2 468		
						134 250/9 824	672 400/ 49 152+

the cost expended. Comparisons of the cost of deploying plates vs taking scrape samples showed that the use of plates is less than half as expensive as scrape samples. This pattern was also reflected in the cost incurred per alien species detected, with the method proposed in this study found to be the most cost-efficient by far.

Although the choice of monitoring method is important, there are additional aspects that need to be considered prior to the implementation of monitoring for marine fouling biota. The first of these is the location of monitoring sites. As hull fouling is an important vector of alien fouling species in South Africa and worldwide (Peters et al. 2019b; Bailey et al. 2020), ports, harbours and marinas form primary invasion sites (e.g. Peters and Robinson 2017). As such, a national monitoring programme should aim to start with the inclusion of such locations. Previous work has highlighted that the Port of Durban is particularly at risk of shipping-related invasions due to connectivity and environmental matching with Asian ports (Faulkner et al. 2017). Similarly, these factors have been highlighted as likely causes of the numerous Chilean species that have recently been recorded in Saldanha Bay, the most-invaded of South Africa's ports (Peters and Robinson 2018). While it would be desirable to institute monitoring of all ports and marinas, this is unlikely to be immediately possible. Thus, it is recommended that key locations be chosen as part of a preliminary monitoring programme. Priority should be given to locations with high traffic volume from regions with similar climatic conditions and those that are visited by multiple vessel types (e.g. the ports of Durban and Saldanha), as these would be the most likely points of new introductions. Additionally, monitoring within mariculture facilities would be an important consideration for a fully-fledged monitoring programme. This is because mariculture operations can be a source of introductions if part of the production process includes the importation of alien species (Mineur et al. 2014), but also because marine farming can be vulnerable to negative impacts associated with fouling taxa (e.g. Lins and Rocha 2020). Thus, farms can benefit from proactively detecting and managing alien species before they have economic consequences. Notably, because coastal mariculture facilities use floating infrastructure, the arrays described in this study could easily be deployed from these structures to provide data comparable with those from harbours. While the choice of location is important, so is the placement of arrays within chosen locations. For example, different quays in ports service different types of vessels, and ports can have various basins. Although this study considered deployment of five arrays in relatively close proximity within each location to increase the probability of detecting newly introduced species at low density, groups of five arrays could be also deployed in distinct areas within harbours. Should specific areas within a harbour service vessels from regions of concern (i.e. those that show climatic matching with the harbour under consideration) then arrays could be dedicated to monitor that harbour area. Ultimately, the greater the coverage within a harbour, the greater the chance of detecting rare, novel species.

The second aspect that needs to be considered before the implementation of a monitoring programme is the timing of deployments. Since it is impossible to predict when a new species will arrive, arrays should always be deployed to maximise the probability of early detection. This could be achieved by placing new arrays into the water as old arrays are retrieved. A deployment period of 16 weeks, as used in this study, would result in three deployments annually. While this period yielded 70–100% coverage of plates (note: <100% cover is desirable to prevent competition for space from excluding any species), it should be noted that this study took place between March and June and the effect of seasonality remains unconsidered. Thus, although fouling species are expected to settle continuously irrespective of peak settlement times for particular species (e.g. *Ciona robusta* that settles in greatest density in winter [Robinson et al. 2017]), overall settlement may be higher in some seasons than others. Therefore, it is suggested that, during a preliminary monitoring programme, seasonality should be assessed, and the length of deployment be adjusted accordingly.

A vital aspect to any monitoring programme is the ability to correctly identify the species that are detected. This can be particularly challenging when faced with alien species that are new to the region. To ensure the integrity of the monitoring programme and maximise the value of the data collected, it will be vital to secure formalised collaborations with regional taxonomists that will: (i) support the training of implementing personnel; (ii) offer support in identifying challenging specimens; and (iii) provide confirmation of taxa that have not been reported from South Africa before. While essential, this could prove a challenging step in establishing a rigorous monitoring programme because marine taxonomic skills are scarce in South Africa (Griffiths et al. 2010). While species identification can largely be done using photographs of the plates, it is important that any specimens for which identification is uncertain be returned to the laboratory. These specimens should be preserved in 97% alcohol to facilitate genetic confirmation of species identification if required. Importantly, genetic approaches should be used alongside traditional morphometrics to confirm all new alien records. This is especially important for taxa known to form part of species complexes that are difficult to distinguish based on morphometric characteristics alone (e.g. Rocha et al. 2019; Simon et al. 2019). This aspect of a monitoring programme has big cost implications as it requires input from highly skilled individuals and the use of expensive genetic techniques. With careful consideration and signing of formal collaboration agreements, these costs could, however, be reduced as close working relationships between academics, and the programme could see costs paid in kind. For example, taxonomists might not charge for their time if they are given permission to publish findings related to the identifications that they confirm.

Once a monitoring programme is well established, there is the potential to magnify its value through the inclusion of data collected by citizen scientists. This has the benefit of increasing the volume of data that can be captured, while promoting awareness and involvement by the public (Lehtiniemi et al. 2020). This could be achieved by engaging yacht owners who could supplement data collection by placing arrays next to their boats, retrieving and photographing them after 16 weeks, and then deploying new arrays. While this would require the provision of arrays and limit identification to photographs, it would expand the coverage of monitoring and alert authorities to sites that require a more focused approach. The implementation of such a citizen science programme could be facilitated through the development of a dedicated mobile application that would enable easy upload of photographs. It is anticipated that the yachting fraternity would be receptive to such a programme as they often display a high potential for ocean stewardship (Lusby and Anderson 2008).

The need to embed a monitoring programme into a comprehensive system of early detection and rapid response

To fully realise the benefit of early detection of new incursions, it is vital that a monitoring programme be embedded within a comprehensive EDRR system (Figure 6). The logical starting point for such a system is a monitoring programme that encompasses the components of monitoring, detection and identification. While such a programme is outlined in the current study, its efficiency could be increased by the use of watch lists. Watch lists offer a cost-efficient approach for identifying species with an invasion history that are currently absent from the region of interest, but for which vectors exist that could facilitate introduction (Faulkner et al. 2014). As such, watch lists provide the information required to focus monitoring efforts and elevate awareness of alien taxa that are likely to arrive. A targeted approach has shown great value in relation to detection of alien fouling taxa elsewhere (e.g. Minchin et al. 2016). While a watch list has been developed for predatory crabs in the South African context (Swart and Robinson 2019), this is a currently underutilised management tool.

After field data are collected and species are identified, it is vital that sound data management is applied. Ideally, these publicly funded data should be stored in an open-access database to support research and management and facilitate state reporting on the status of alien taxa (e.g. van Wilgen and Wilson 2018). These data would not only serve a surveillance purpose but would enable tracking of spread and the assessment of efficacy of policy and management interventions, and in the long-term could help to shed light on how other drivers of global environmental change (e.g. climate change) influence bioinvasions (Essl et al. 2020). In a South African context, the immediate benefit drawn from these data would be the ability to meet domestic legal imperatives under the National Environmental Management: Biodiversity Act (RSA 2004) and international obligations under Aichi Target 9 of the CBD. Importantly, because timeous administrative actions are a key requirement for rapid management responses (Burgiel 2020), detection of a new species should trigger reporting through clearly defined channels to ensure that authorities tasked with risk assessment and feasibility screening are informed immediately.

While a well-developed and transparent framework for assessing risk associated with alien taxa is available in South Africa (i.e. Risk Analysis for Alien Taxa [RAAT]: Kumschick et al. [2020]), the processes of feasibility screening are less formalised. This aspect of the decision-making process requires an understanding of the biological, logistical, financial and legal constraints that managing the taxa under question may face (Anderson 2005; Mabin et al. 2020). As a result of the complexity of these considerations and the time-constraints associated with the process, it is essential that a transdisciplinary team with specialised skills be preassembled and ready to screen taxa as they are recorded. It has been suggested that risk analysis and feasibility screening occur concurrently to facilitate decisions within a matter of days (Reaser et al. 2020)—although the challenges associated with this time-frame should not be underestimated.

Following the assessment of response options, one of four response measures could be implemented; these are: (i) documentation (i.e. no actions in the field but all documentation gathered is made publicly available); (ii) further analysis (i.e. insufficient information was gathered during risk analysis and feasibility screening to inform a management decision and thus those processes must continue); (iii) control (i.e. prevention of alien species' spread); and (iv) eradication (i.e. complete removal of the species) (Reaser et al. 2020). As implementation of these measures is time-sensitive, clearly defined roles and responsibilities, and timely flows of financial resources, are key to ensuring their success (Mabin et al. 2020).

Conclusions

The value of a South African monitoring programme for alien fouling biota is clear, and the results of this study suggest that this can be achieved in an effective, cost-efficient manner. Importantly, this approach provides an evidence-based solution for financially constrained countries wishing to play an active role in understanding and managing the threat of marine alien species. As the need to strengthen regional biosecurity becomes more evident (Faulkner et al. 2020), broadscale adoption of effective and financially viable monitoring programmes is going to be key to achieving this goal. This is particularly important in Africa where capacity for addressing invasions remains severely constrained (Egoh et al. 2020).

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References

Ahyong S, Costello MJ, Galil BS, Gollasch S, Hutchings P, Katsanevakis S et al. 2021. World Register of Introduced Marine

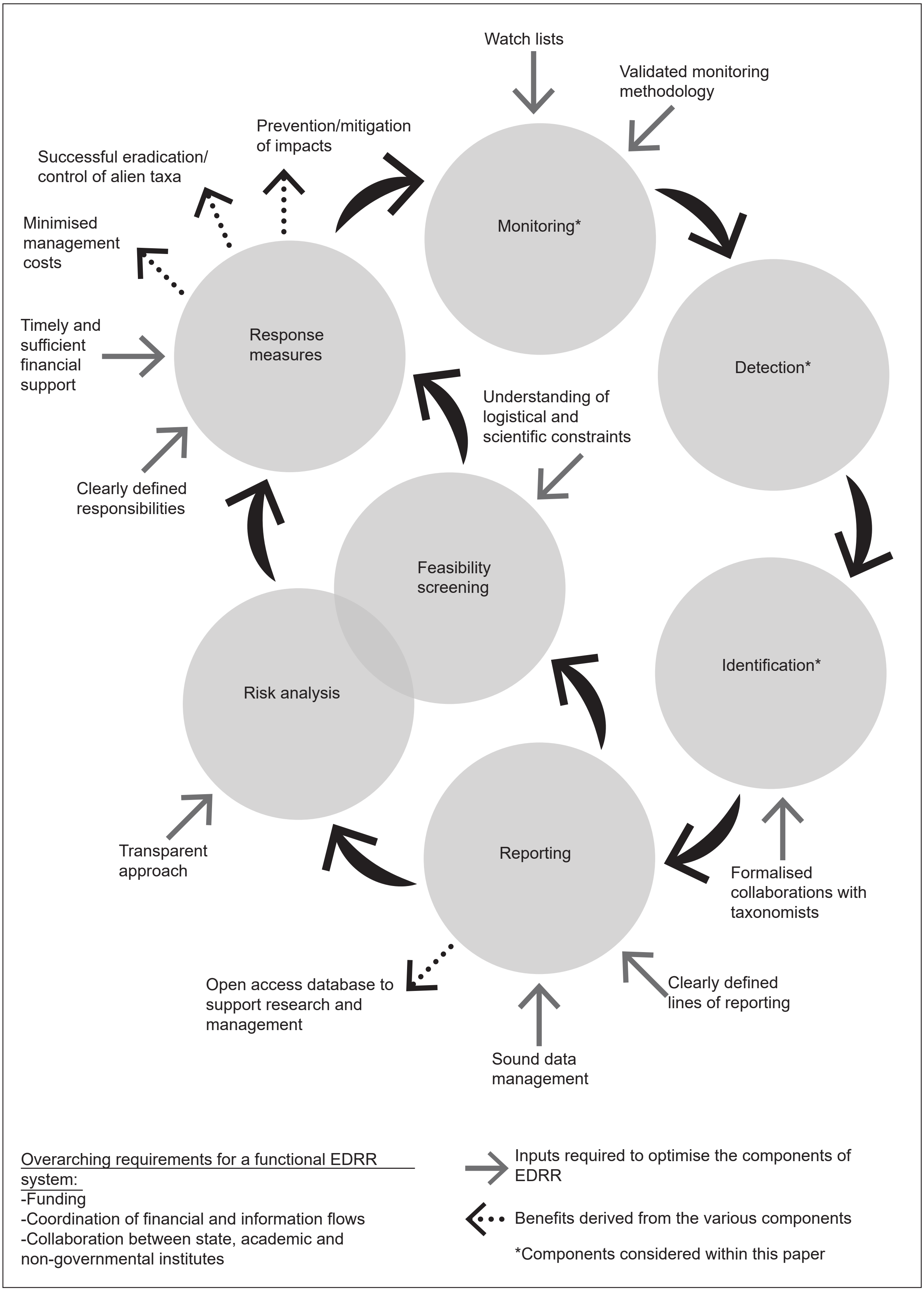

Figure 6: A schematic demonstrating how a monitoring programme forms part of a comprehensive EDRR system (adapted from Reaser et al. 2020). Inputs required to optimise each component are highlighted, as are the benefits derived from each. EDRR = early detection of and rapid response to bioinvasions

Species (WRiMS). Available at http://www.marinespecies.org/introduced [accessed 14 January 2021].
Anderson LW. 2005. California's reaction to *Caulerpa taxifolia*: a model for invasive species rapid response. *Biological Invasions* 7: 1003–1016.
Bachelet G, Blanchet H, Cottet M, Dang C, de Montaudouin X, de Moura Queiros A et al. 2009. A round-the-world tour almost completed: first records of the invasive mussel *Musculista senhousia* in the north-east Atlantic (southern Bay of Biscay). *Marine Biodiversity Records* 2: e199.
Bailey SA, Brown L, Campbell ML, Canning-Clode J, Carlton JT, Castro N et al. 2020. Trends in the detection of aquatic non-indigenous species across global marine, estuarine and freshwater ecosystems: a 50-year perspective. *Diversity and Distributions* 26: 1780–1797.
Bannister J, Sievers M, Bush F, Bloecher N. 2019. Biofouling in marine aquaculture: a review of recent research and developments. *Biofouling* 35: 631–648.
Bellard C, Cassey P, Blackburn TM. 2016. Alien species as a driver of recent extinctions. *Biology Letters* 12: article 20150623.
Blum JC, Chang AL, Liljesthröm M, Schenk ME, Steinberg MK, Ruiz GM. 2007. The non-native solitary ascidian *Ciona intestinalis* (L.) depresses species richness. *Journal of Experimental Marine Biology and Ecology* 342: 5–14.
Burgiel SW. 2020. The incident command system: a framework for rapid response to biological invasion. *Biological Invasions* 22: 155–165.
Capinha C, Essl F, Seebens H, Moser D, Pereira HM. 2015. The dispersal of alien species redefines biogeography in the Anthropocene. *Science* 348: 1248–1251.
CBD (Convention on Biological Diversity). 2018. Analysis of the contribution of targets established by parties and progress towards the Aichi biodiversity targets. CBD/SBI/2/2/Add.2. Montreal, Canada: CBD. Available at https://www.cbd.int/doc/c/e24a/347c/a8b84521f326b90a198b1601/sbi-02-02-add2-en.pdf [accessed 15 March 2020].
Clarke K, Warwick R. 2001. *Change in marine communities: an approach to statistical analysis and interpretation.* Plymouth, UK: Primer-E.
Darwin C. 1854. *A monograph on the sub-class Cirripedia with figures of all the species*. London: Ray Society Publication.
Early R, Bradley BA, Dukes JS, Lawler JJ, Olden JD, Blumenthal DM et al. 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications* 7: article 12485.
Egoh BN, Ntshotsho P, Maoela MA, Blanchard R, Ayompe LM, Rahlao S. 2020. Setting the scene for achievable post-2020 convention on biological diversity targets: a review of the impacts of invasive alien species on ecosystem services in Africa. *Journal of Environmental Management* 261: e110171.
Essl F, Latombe G, Lenzner B, Pagad S, Seebens H, Smith K et al. 2020. The Convention on Biological Diversity (CBD)'s post-2020 target on invasive alien species – what should it include and how should it be monitored? *NeoBiota* 62: 99–121.
Evans T, Blackburn TM. 2020. Global variation in the availability of data on the environmental impacts of alien birds. *Biological Invasions* 22: 1027–1036.
Faulkner KT, Robertson MP, Rouget M, Wilson JR. 2014. A simple, rapid methodology for developing invasive species watch lists. *Biological Conservation* 179: 25–32.
Faulkner KT, Robertson MP, Rouget M, Wilson JRU. 2017. Prioritising surveillance for alien organisms transported as stowaways on ships travelling to South Africa. *PLoS ONE* 12: e0173340.
Faulkner KT, Robertson MP, Wilson JR. 2020 Stronger regional biosecurity is essential to prevent hundreds of harmful biological invasions. *Global Change Biology* 26: 2449–2462.
Fitridge I, Dempster T, Guenther J, De Nys R. 2012. The impact and control of biofouling in marine aquaculture: a review. *Biofouling* 28: 649–669.
Griffiths CL, Robinson TB, Lange L, Mead A. 2010. Marine biodiversity in South Africa: an evaluation of current states of knowledge. *PLoS ONE* 5: e12008.
Holmes S, Callaway R. 2020. Design and implementation of two surveys targeted at describing fouling communities and identifying non-native species within active ports. *Journal of the Marine Biological Association of the United Kingdom* 100: 1191–1204.
Kumschick S, Wilson JRU, Foxcroft LC. 2020. A framework to support alien species regulation: the Risk Analysis for Alien Taxa (RAAT). *NeoBiota* 62: 213–239.
Lambert L. 2009. Adventures of a sea squirt sleuth: unraveling the identity of *Didemnum vexillum*, a global ascidian invader. *Aquatic Invasions* 4: 5–28.
Latombe G, Pyšek P, Jeschke JM, Blackburn TM, Bacher S, Capinha C et al. 2017. A vision for global monitoring of biological invasions. *Biological Conservation* 213: 295–308.
Lehtiniemi M, Outinen O, Puntila-Dodd R. 2020. Citizen science provides added value in the monitoring for coastal non-indigenous species. *Journal of Environmental Management* 267: e110608.
Lins DM, De Marco P, Andrade AFA, Rocha RM. 2018. Predicting global ascidian invasions. *Diversity and Distributions* 24: 692–704.
Lins DM, Rocha RM. 2020. Cultivated brown mussel (*Perna perna*) size is reduced through the impact of three invasive fouling species in southern Brazil. *Aquatic Invasions* 15: 114–126.
Lodge DM, Simonin PW, Burgiel SW, Keller RP, Bossenbroek JM et al. 2016. Risk analysis and bioeconomics of invasive species to inform policy and management. *Annual Review of Environment and Resources* 41: 453–488.
Loureiro TG, Peters K, Robinson T. 2021. Light, shade and predation: who wins and who loses in benthic sessile communities? *Marine Biodiversity* 51: article 94.
Lourenço PM, Henriques M, Catry I, Granadeiro JP, Catry T. 2018. First record of the invasive Asian date mussel *Arcuatula senhousia* (Benson, 1842) (Mollusca: Bivalvia: Mytilidae) in West Africa. *Journal of Natural History* 52: 2567–2571.
Lusby CM, Anderson S. 2008. Community and quality of life – the case of ocean cruising. *World Leisure Journal* 50: 232–242.
Mabin CA, Wilson JRU, le Roux JJ, Majiedt P, Robinson TB. 2020. The first management of a marine invader in Africa: the importance of trials prior to setting long-term management goals. *Journal of Environmental Management* 261: e110213.
Mack RN, Simberloff D, Lonsdale WM, Evans H, Clout M, Bazzaz FA. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications* 10: 689–710.
McGeoch MA, Genovesi P, Bellingham PJ, Costello MJ, McGrannachan C, Sheppard A. 2016. Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. *Biological Invasions* 18: 299–314.
Measey J, Wagener C, Mohanty NP, Baxter-Gilbert J, Pienaar EF. 2020. The cost and complexity of assessing impact. *NeoBiota* 62: 279–299.
Minchin D, Olenin S, Liu TK, Cheng M, Huang SC. 2016. Rapid assessment of target species: byssate bivalves in a large tropical port. *Marine Pollution Bulletin* 112: 177–182.
Mineur F, Le Roux A, Maggs CA, Verlaque M. 2014. Positive feedback loop between introductions of non-native marine species and cultivation of oysters in Europe. *Conservation Biology* 28: 1667–1676.
Monniot C, Monniot F, Griffiths CL, Schleyer M. 2001. South African ascidians. *Annals of the South African Museum* 108: 1–141.
Montagu G. 1814. An essay on sponges, with descriptions of all the species that have been discovered on the coast of Great Britain. *Memoirs of the Wernerian Natural History Society* 2: 67–121.

Oka A. 1927. Zur kenntnis der japanischen Botryllidae. *Proceedings of the Imperial Academy* 3: 607–609.

Ordóñez V, Pascual M, Fernández-Tejedor M, Pineda MC, Tagliapietra D, Turon X. 2015. Ongoing expansion of the worldwide invader *Didemnum vexillum* (Ascidiacea) in the Mediterranean Sea: high plasticity of its biological cycle promotes establishment in warm waters. *Biological Invasions* 17: 2075–2085.

Peters K, Robinson TB. 2017. From Chile to the South African west coast: first reports of the marine alien amphipod *Caprella mutica* (Schurin, 1935) in South Africa. *BioInvasions Records* 6: 61–66.

Peters K, Robinson TB. 2018. From Chile to the South African west coast: first reports of the Chilean stone crab *Homalaspis plana* (H. Milne Edwards, 1843) and the South American sunstar *Heliaster helianthus* (Lamarck, 1816) outside their natural ranges. *BioInvasions Records* 7: 421–426.

Peters K, Sink KJ, Robinson TB. 2017. Raising the flag on marine alien fouling species. *Management of Biological Invasions: International Journal of Applied Research on Biological Invasions* 8: 1–11.

Peters K, Sink KJ, Robinson TB. 2019a. Sampling methods and approaches to inform standardized detection of marine alien fouling species on recreational vessels. *Journal of Environmental Management* 230: 159–167.

Peters K, Sink K, Robinson TB. 2019b. Aliens cruising in: explaining alien fouling macro-invertebrate species numbers on recreational yachts. *Ocean and Coastal Management* 182: article 104986.

Pyšek P, Hulme PE, Simberloff D, Bacher S, Blackburn TM, Carlton JT et al. 2020. Scientists' warning on invasive alien species. *Biological Reviews of the Cambridge Philosophical Society* 95: 1511–1534.

R Core Team. 2020. *R: a language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing.

Reaser JK, Burgiel SW, Kirkey J, Brantley KA, Veatch SD, Burgos-Rodriguez J. 2020. The early detection of and rapid response (EDRR) to invasive species: a conceptual framework and federal capacities assessment. *Biological Invasions* 22: 1–19.

Rius M, Heasman KG, McQuaid CD. 2011. Long-term coexistence of non-indigenous species in aquaculture facilities. *Marine Pollution Bulletin* 62: 2395–2403.

Rius M, Bourne S, Hornsby HG, Chapman MA. 2015. Applications of next-generation sequencing to the study of biological invasions. *Current Zoology* 61: 488–504.

Robinson TB, Alexander ME, Simon CA, Griffiths CL, Peters K, Sibanda S et al. 2016. Lost in translation? Standardising the terminology used in marine invasion biology and updating South African alien species lists. *African Journal of Marine Science* 38: 129–140.

Robinson TB, Havenga B, van der Merwe M, Jackson S. 2017. Mind the gap – context dependency in invasive species impacts: a case study of the ascidian *Ciona robusta*. *NeoBiota* 32: 127–141.

Robinson TB, Martin N, Loureiro TG, Matikinca P, Robertson MP. 2020a. Double trouble: the implications of climate change for biological invasions. *NeoBiota* 62: 463–487.

Robinson TB, Peters K, Brooker B. 2020b. Coastal invasions: the South African context. In: van Wilgen BW, Measey J, Richardson DM, Wilson JR, Zengeya TA (eds), *Biological invasions in South Africa*. Cham, Switzerland: Springer International Publishing. pp 229–247.

Rocha RM, Salonna M, Griggio F, Ekins M, Lambert G, Mastrototaro F et al. 2019. The power of combined molecular and morphological analyses for the genus *Botrylloides*: identification of a potentially global invasive ascidian and description of a new species. *Systematics and Biodiversity* 17: 509–526.

Rocha RM, Zanata TB, Moreno TR. 2012. Keys for the identification of families and genera of Atlantic shallow water ascidians. *Biota Neotropica* 12: 269–303.

Rogers P, Nunan F, Fentie AA. 2017. Reimagining invasions: the social and cultural impacts of *Prosopis* on pastoralists in southern Afar, Ethiopia. *Pastoralism* 7: article 22.

RSA (Republic of South Africa). 2004. National Environmental Management: Biodiversity Act (NEMBA) (Act No. 10 of 2004). *Government Gazette, South Africa* 467 (26436).

Rueden CT, Schindelin J, Hiner MC, DeZonia BE, Walter AE, Arena ET, Eliceiri W 2017. ImageJ2: ImageJ for the next generation of scientific image data. *BMC Bioinformatics* 18: article 529.

Ruiz GM, Fofonoff PW, Steves B, Foss SF, Shiba SN. 2011. Marine invasion history and vector analysis of California: a hotspot for western North America. *Diversity and Distributions* 17: 362–373.

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM et al. 2018. Global rise in emerging alien species results from increased accessibility of new source pools. *Proceedings of the National Academy of Sciences of the United States of America* 115: E2264–E2273.

Shackleton RT, Shackleton CM, Kull CA. 2019. The role of invasive alien species in shaping local livelihoods and human well-being: a review. *Journal of Environmental Management* 229: 145–157.

Simberloff D, Martin J-L, Genovesi P, Maris V, Wardle DA, Aronson J et al. 2013. Impacts of biological invasions: what's what and the way forward. *Trends in Ecology and Evolution* 28: 58–66.

Simon C, Niekerk HV, Burghardt I, Hove HT, Kupriyanova E. 2019. Not out of Africa: *Spirobranchus kraussii* (Baird, 1865) is not a global fouling and invasive serpulid of Indo-Pacific origin. *Aquatic Invasions* 14: 221–249.

Sink KJ, Holness S, Harris L, Majiedt PA, Atkinson L, Robinson T et al. 2012. National Biodiversity Assessment 2011: Technical report, vol. 4: marine and coastal component. Pretoria, South Africa: South African National Biodiversity Institute.

Smith AM, Gordon DP. 2011. Bryozoans of southern New Zealand: a field identification guide. *New Zealand Aquatic Environment and Biodiversity Report* No. 75. Wellington, New Zealand: Ministry of Fisheries.

Stefaniak LM. 2017. Mechanisms for invasion success by *Didemnum vexillum* (Chordata: Ascidiacea): observations versus manipulations. *Biological Invasions* 19: 1213–1225.

Swart C, Robinson TB. 2019. Horizon scanning for alien predatory crabs: insights from South Africa. *African Journal of Marine Science* 41: 125–135.

Turbelin AJ, Malamud BD, Francis RA. 2017. Mapping the global state of invasive alien species: patterns of invasion and policy responses. *Global Ecology and Biogeography* 26: 78–92.

Ugland KI, Gray JS, Ellingsen KE. 2003. The species-accumulation curve and estimation of species richness. *Journal of Animal Ecology* 72: 888–897.

Valentine PC, Carman MR, Blackwood DS, Heffron EJ. 2007. Ecological observations on the colonial ascidian *Didemnum* sp. in a New England tide pool habitat. *Journal of Experimental Marine Biology and Ecology* 342: 109–121.

van Wilgen BW, Wilson JR. 2018. The status of biological invasions and their management in South Africa in 2017. Cape Town, South Africa: South African National Biodiversity Institute, and Stellenbosch, South Africa: DST-NRF Centre of Excellence for Invasion Biology.

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S et al. 2010. How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Frontiers in Ecology and the Environment* 8: 135–144.

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