



Regional-scale benthic monitoring for ecosystem-based fisheries management (EBFM) using an autonomous underwater vehicle (AUV)

Dan A. Smale^{1*}, Gary A. Kendrick¹, Euan S. Harvey¹, Timothy J. Langlois¹, Renae K. Hovey², Kimberly P. Van Niel², Kris I. Waddington³, Lynda M. Bellchambers⁴, Matthew B. Pember⁴, Russ C. Babcock⁵, Mathew A. Vanderklift⁵, Damian P. Thomson⁵, Michael V. Jakuba⁶, Oscar Pizarro⁶, and Stefan B. Williams⁶

¹School of Plant Biology and Oceans Institute, University of Western Australia, Crawley, WA 6009, Australia

²School of Earth and Environment and Oceans Institute, University of Western Australia, Crawley, WA 6009, Australia

³Centre for Marine Futures and Oceans Institute, University of Western Australia, Crawley, WA 6009, Australia

⁴Department of Fisheries, Western Australian Fisheries and Marine Research Laboratories, PO Box 20, North Beach, WA 6920, Australia

⁵CSIRO Marine and Atmospheric Research, Underwood Avenue, Floreat, WA 6014, Australia

⁶Australian Centre for Field Robotics, University of Sydney, Sydney, NSW 2006, Australia

*Corresponding author: tel: +61 8 6488 2219; fax: +61 8 6488 1126; e-mail: daniel.smale@uwa.edu.au

Smale, D. A., Kendrick, G. A., Harvey, E. S., Langlois, T. J., Hovey, R. K., Van Niel, K. P., Waddington, K. I., Bellchambers, L. M., Pember, M. B., Babcock, R. C., Vanderklift, M. A., Thomson, D. P., Jakuba, M. V., Pizarro, O., and Williams, S. B. 2012. Regional-scale benthic monitoring for ecosystem-based fisheries management (EBFM) using an autonomous underwater vehicle (AUV). – ICES Journal of Marine Science, 69: 1108–1118.

Received 13 January 2012; accepted 7 April 2012; advance access publication 18 May 2012.

Monitoring marine habitats and biodiversity is critical for understanding ecological processes, conserving natural resources, and achieving ecosystem-based fisheries management (EBFM). Here, we describe the application of autonomous underwater vehicle (AUV) technology to conduct ongoing monitoring of benthic habitats at two key locations in Western Australia. Benthic assemblages on rocky reefs were sampled with an AUV, which captured >200 000 geo-referenced images. Surveys were designed to obtain 100% coverage of 25 × 25 m patches of benthic habitat. In 2010, multiple patches were surveyed at 15–40-m depths at three reference sites at the Houtman Abrolhos Islands and at six reference sites at Rottneest Island. The following year, repeat surveys of the same geo-referenced patches were conducted. Benthic assemblages at the Houtman Abrolhos Islands were varied in that one reference site was dominated by hard corals, whereas the other two were macroalgae dominated. Conversely, assemblages at Rottneest Island were dominated by the kelp *Ecklonia radiata*. The AUV resurveyed each patch with high precision and demonstrated adequate power to detect change. Repeated observations at the reference sites will track natural variability in benthic habitat structure, which in turn will facilitate the detection of ecological change and ultimately feed back into EBFM processes.

Keywords: biodiversity monitoring, coastal management, community structure, marine habitats, remote technology, underwater surveys.

Introduction

Over the past decade, the management of fishing activities in marine ecosystems has shifted from a traditional single-species approach towards ecosystem-based fisheries management (EBFM), which considers multiple biological, physical, and socio-economic components of complex systems. In essence, the transition to EBFM represents an increase in the complexity of ecosystem processes and pressures that must be considered by managers. There is now

widespread agreement that the management of fisheries and marine resources must be ecosystem-based. This is because although the maintenance of target stocks has long been the primary management goal for fisheries agencies, non-target stocks and the broader ecosystem must also be maintained at acceptable levels to achieve stock sustainability (Pikitch *et al.*, 2004). However, the “nuts and bolts” of implementing EBFM at large spatial scales and across contrasting regions has generated much

debate (Smith *et al.*, 2007; Ray, 2011), and the rate of transition to EBFM has generally been inadequate (Pitcher *et al.*, 2009).

Documenting the status of marine habitats and biodiversity and tracking changes in the “health” of such biological assets over time are key activities in the EBFM process (Pikitch *et al.*, 2004). Marine ecosystems are both complex and dynamic, so that measuring absolute changes in biodiversity, structure, or function is often unrealistic and impractical. Instead, it is possible to employ a suite of ecological indicators which can be used to track the health of the overall system or specific resources. In this way, indicators act as early warning systems of environmental change (Niemi and McDonald, 2004). In this case, indicators are often “surrogates” for another measure (most commonly biodiversity, but also measures of ecosystem function, community structure, or productivity) that is either impractical or too costly to quantify in absolute form (e.g. Oliver and Beattie, 1996; Olsgard *et al.*, 2003).

The choice of methodology and sampling design employed to measure ecological indicators has implications for the effectiveness of all monitoring efforts (Hewitt *et al.*, 2001; Langlois *et al.*, 2010). The range of sampling tools and techniques available for marine research has rapidly increased in recent years, following widespread advances in underwater technologies. Traditionally, sampling for long-term subtidal monitoring has been conducted either by scuba divers (e.g. Dayton *et al.*, 1984) or by deploying destructive sampling equipment, such as trawls and grabs (e.g. Blomqvist, 1991). More recently, remotely deployed photographic sampling (e.g. with towed video or drop-cameras) has been conducted (e.g. Smale *et al.*, 2010a).

The deployment of autonomous underwater vehicles (AUVs) for mapping and sampling the deep-sea environment is a further technological development. In recent years, AUVs have been used to survey invasive pest species (Barrett *et al.*, 2010), describe benthic community structure at depths of >1000s m (Sherman and Smith, 2009), examine coral mound morphology (Grasmueck *et al.*, 2006), and assess environmental impacts of the Deepwater Horizon oil spill (Camilli *et al.*, 2010). Within the context of resource management, AUVs have facilitated surveys of groundfish and their associated habitat (Clarke *et al.*, 2009), as well as sensitive coral-dominated habitats (Singh *et al.*, 2004; Armstrong *et al.*, 2006). AUVs are emerging as highly effective tools for sampling marine environments, because they (i) are untethered and can conduct non-destructive sampling in remote marine habitats (i.e. under ice shelves, over untrawable terrain, and in depths >1000 m); (ii) can repeat spatial surveys to detect change over time; and (iii) are fitted with a range of instrumentation to facilitate the simultaneous acquisition of a variety of physical and biological data. Here, we describe the use of an AUV for conducting long-term monitoring of benthic habitats in temperate Western Australia (WA) to support EBFM activities within the “West Coast Bioregion” (as defined by the Department of Fisheries, WA).

In temperate WA, coastal marine ecosystems are impacted by fishing for targeted species such as the Western Rock Lobster, *Panulirus cygnus* (Australia’s most valuable single-species fishery) and Pink Snapper, *Pagrus auratus*. Widespread commercial and recreational fishing activities have influenced trophic processes (e.g. through the addition of bait, see Waddington and Meeuwig, 2009), population structure of targeted species (Langlois *et al.*, in press-a), and fish community structure (Watson *et al.*, 2007). Moreover, fragile habitats, such as some coral reefs around the

Houtman Abrolhos Islands, have been physical perturbed by the deployment of lobster pots (Webster *et al.*, 2002). Concurrently, the most pressing environmental change factors influencing marine ecosystems in WA are thought to include variability in ocean currents (Gaughan, 2007), changes in land run-off (Lough and Hobday, 2011), introduced marine pests (Sliwa *et al.*, 2009; Smale and Childs, 2012), and seawater acidification and warming (Orr *et al.*, 2005; Pearce and Feng, 2007; Wernberg *et al.*, 2011a). However, specific ecological responses to fishing and environmental change are not well known and the ability to predict (and mitigate for) changes in ecosystem structure is poor. A major impediment to marine research is the lack of historical baseline information against which to detect change, and many regions along the vast, remote coastline of WA are “data-poor” (Wernberg *et al.*, 2011a). However, the old, climatically buffered, infertile seascape of southwestern Australia with its limited environmental heterogeneity (Langlois *et al.*, 2012) has been shown to provide a simple model system in which to investigate the impacts of pressures.

Management of marine ecosystems in temperate WA is currently shifting from a single species to an ecosystem-based approach. To provide a framework for this transition, a decision-making/assessment process has been developed by the Department of Fisheries WA (Fletcher *et al.*, 2010) which defines assets that must be considered for ecosystem-based management. Environmental, ecological, social, and economic assets associated with the marine ecosystems of WA are included in component trees, and each asset has been assigned a risk rating to prioritize management efforts and to identify knowledge gaps (Fletcher *et al.*, 2010). However, to validate and (if necessary) alter individual risk ratings, information on the status of assets over time must be collected and fed back into the component trees. Submerged rocky reefs are key assets within the EBFM approach, as they support rich and abundant assemblages of macroalgae, invertebrates, and reef fish (see Smale *et al.*, 2011, for recent review). Reef assemblages are highly productive and generate food and habitat for economically important species such as the Western Rock Lobster and Pink Snapper. To achieve a cost-effective approach to monitoring these assets, several key indicators that capture the quality, extent, and composition of benthic habitats have been proposed. These include the relative abundance of warm and cool water habitat-forming species (e.g. reef-building corals vs. canopy forming macroalgae), the spatial coverage of species that may be particularly sensitive to ocean warming (e.g. the brown macroalgae *Scytothalia dorycarpa*), and multivariate assemblage structure at coarse taxonomic levels (Smale *et al.*, 2010a, 2011; Wernberg *et al.*, 2011b)

Here, we report on the commencement of a monitoring programme to quantify the structure of benthic habitats in temperate WA. The goals of the monitoring efforts are to (i) improve the understanding of natural variability in benthic assemblages, (ii) facilitate the detection of significant ecological change driven by, for example, ocean warming, and (iii) generate time-series data that can be used to validate and/or amend the risk ratings associated with ecological assets identified through the EBFM process. To achieve these goals, an AUV was deployed to conduct ongoing, repeat sampling at two key locations. To our knowledge, this represents the first application of the AUV technology for conducting routine monitoring of benthic habitat indicators to support EBFM. The work reported here also forms part of a nationwide programme of AUV surveys of benthic ecosystems

implemented as one of the biological components of Australia's Integrated Marine Observing System (IMOS).

Methods

AUV "Sirius"

Surveys were conducted with an AUV "Sirius", which is operated by the Australian Centre for Field Robotics (ACFR, University of Sydney) and supported by the IMOS. The vehicle is a modified Seabed class AUV, similar to that first described by Singh *et al.* (2004) and recently detailed in Williams *et al.* (2012). The AUV is equipped with a full suite of oceanographic instruments, including a high-resolution stereo camera pair and strobes, multibeam sonar, depth, conductivity, and temperature sensors, Doppler Velocity Log including a compass with integrated roll and pitch sensors, Ultra Short Baseline Acoustic Positioning System (USBL), and a forward-looking obstacle avoidance sonar (Supplementary ESM 1). Images were collected with a synchronized pair of high sensitivity 12 bit, 1.4 megapixel cameras (AVT Prosilica GC1380 and GC1380C; one monochrome and one colour). The "flight path" for each AUV dive can be precisely pre-programmed with a range of sampling designs, depending on the purpose of the survey. For the current surveys, the AUV tracked the seabed at an altitude of 2 m at a cruising speed of 0.5 m s^{-1} . The AUV was capable of conducting dives of up to 3 h (limited by battery-life) and three or four dives per day. The vehicle was operated and deployed by a team of four scientists/technicians from ACFR. All surveys were conducted during daytime, in April 2010 and April 2011.

Survey locations and design

Long-term monitoring was established at two key locations along the Western Australian coastline; the Houtman Abrolhos Islands ("Abrolhos") and Rottnest Island ("Rottnest"; Figure 1). These locations have been identified as key indicator regions because of their ecological sensitivity and socio-economic importance. The Abrolhos Islands, 80 km off mainland Australia, form a remote archipelago with only seasonal human habitation and are therefore "relatively" not impacted by anthropogenic pressures (but see Watson *et al.*, 2007). The islands play a key role in Australia's biggest single-species fishery (Western Rock Lobster). At the Abrolhos, patches of hard corals and *Ecklonia radiata* beds coexist to form mixed habitats (Hatcher, 1985). This unusual juxtaposition is primarily a consequence of the long environmental stability and the unique oceanography of the region. The islands are situated within the main flow of the Leeuwin Current, a geologically old, poleward flowing boundary current which ameliorates ocean conditions and transports tropical organisms and warm-water southwards (Smith *et al.*, 1991; Pearce, 1997). As such, many temperate species, including *E. radiata*, are found towards the northern "warm" edge of their ranges, whereas many tropical species are found towards the southern "cool" extreme of their distributions. The juxtaposition of corals and kelps is thought to represent a useful early warning signal for detecting ecological effects of increased seawater temperature in the region (Smale *et al.*, 2010a). Rottnest Island is a major asset for tourism and recreational fishing in the region and supports a highly diverse and abundant marine life, which includes warm-water fish and coral species at the southern limit of their distributions as well as extensive stands of the kelp, *E. radiata*. Kelp bed communities may be sensitive to both local anthropogenic

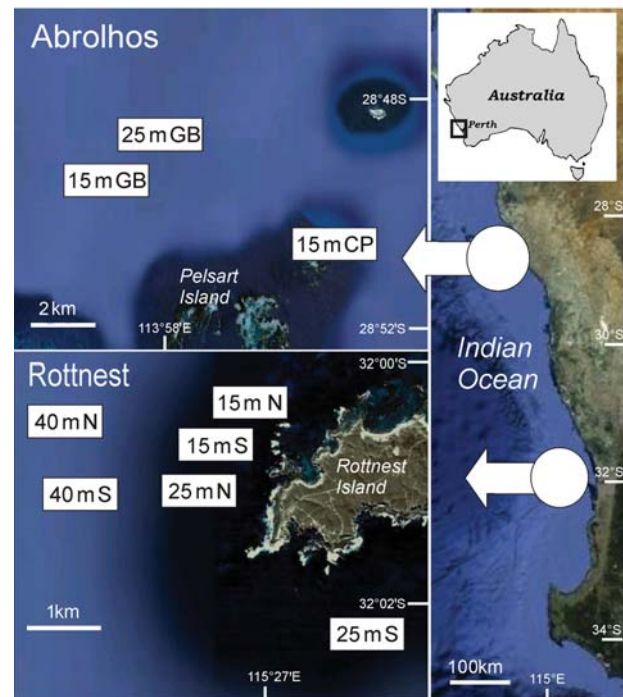


Figure 1. Locations and sites sampled by AUV in 2010 and 2011. The satellite map on the right indicates the locations of Rottnest and Abrolhos along the Western Australian coastline, with an inset map of the Australian continent. Expanded maps on the left indicate approximate position of each reference site at each location.

impacts and longer-term physical forcing from seawater warming and changes in storm patterns (Wernberg *et al.*, 2011a). Moreover, sensitive hard corals at both locations may be particularly prone to physical damage from fishing practices, such as the deployment of lobster pots.

The key objective of the sampling was to survey predominantly rocky reef/coral habitat that could be established as "reference sites" and revisited as a part of monitoring efforts. As such, sites were selected based on bathymetry maps and existing knowledge, to target moderate to high relief reef (i.e. with vertical extent of topographic features $>1 \text{ m}$) rather than low-lying, low-complexity reef platforms. The Abrolhos support a unique and diverse array of marine organisms and assemblages and represent the highest latitude extensive coral reef system in the southern hemisphere (Hatcher, 1985). The islands also support extensive stands of the kelp *E. radiata*, which represent the "warm" extreme of its distribution. At the Abrolhos, three reference sites were chosen to encompass both kelp-dominated (15 m Coral Patches and 25 m Geebank) and coral-dominated habitat types (15 m North). At Rottnest Island, replicate sites were positioned along a depth gradient, with two sites at 15, 25, and 40 m depth (Figure 1). At Rottnest, the sites "15 m South" and "25 m North" were positioned within a protected area (the "West End Demersal Sanctuary Zone", where all fishing activities on the seabed are prohibited). This design, with fished and unfished areas also allows for more accurate attribution of observed variation between fishing and other environmental sources of variation. Further site details are provided in Supplementary ESM 2.

Within each site, three replicate "grids" were initially surveyed by the AUV in 2010. The submersible conducted a series of

parallel, overlapping 25 m long transects, which covered a combined area of 625 m² of seabed (i.e. 25 × 25 m). Replicate grids within a site were situated 50–200 m apart to encapsulate spatial heterogeneity within each site and achieve spatial independence (*sensu* Hurlbert, 1984). During the dive, the AUV captured overlapping geo-referenced stereo images of the benthos, as well as multibeam data at two resolutions and physiochemical data (temperature, salinity, light, chlorophyll). The only exception was the “25 m Geebank” site at the Abrolhos, where only one grid on suitable hard substrata was surveyed. In 2011, the AUV resurveyed (usually) three or (in a few cases) two grids at every reference site (Supplementary ESM 2). The only exception was again the “25 m Geebank” site at the Abrolhos, which was relocated to more suitable substrata for long-term monitoring and, as such, the previously surveyed grid was not revisited. Finally, *in situ* loggers were deployed at the monitoring sites by scuba divers to obtain continuous temperature readings (every 30 min).

Post-processing and analysis

Over 1000 geo-referenced stereo image pairs were collected at each grid within the reference sites. Stereo image pairs were stitched together to generate composite “meshes” of the entire 25 × 25 m patch of seabed (see Johnson-Roberson *et al.*, 2010, for procedural details). These meshes provided a complete overview of each reference site and were used to assess broad-scale ecological structure,

such as kelp canopy cover and coral bleaching (Figure 2). For this study, individual high-resolution images (each capturing ~1.5 × 1.0 m of seabed) were subsampled from each grid to quantify all taxa >50 mm in size using the standard image analysis techniques. Subsamples were selected at 20 s intervals from each AUV dive, to generate a sample set of 101–129 non-overlapping images that maximized spatial coverage of each grid. In this way, benthic assemblage structure was characterized at 28 grids spread across nine reference sites by analysing ~5200 high-resolution images. Representative images of the common assemblage types are shown in Figure 2.

For image analysis, 50 random points were digitally overlaid onto each sample, and the number of points covering each benthic category was counted (using CPCe software, see Kohler and Gill, 2006). This value was then doubled to give a proxy of per cent cover. Forty benthic categories (Supplementary ESM 3) representing flora, fauna and substratum characteristics were determined *a priori* based on previous research (Smale *et al.*, 2010a) and used to classify each image. Care was taken to include conspicuous species of considerable ecological importance, such as the canopy-forming brown algae *E. radiata* and *S. dorycarpa*, while also using functional/morphological groups to achieve a broad, holistic approach to describing the benthos. Bleached coral was considered “alive”, but additional information on the spatial extent of any bleaching was recorded. It was evident

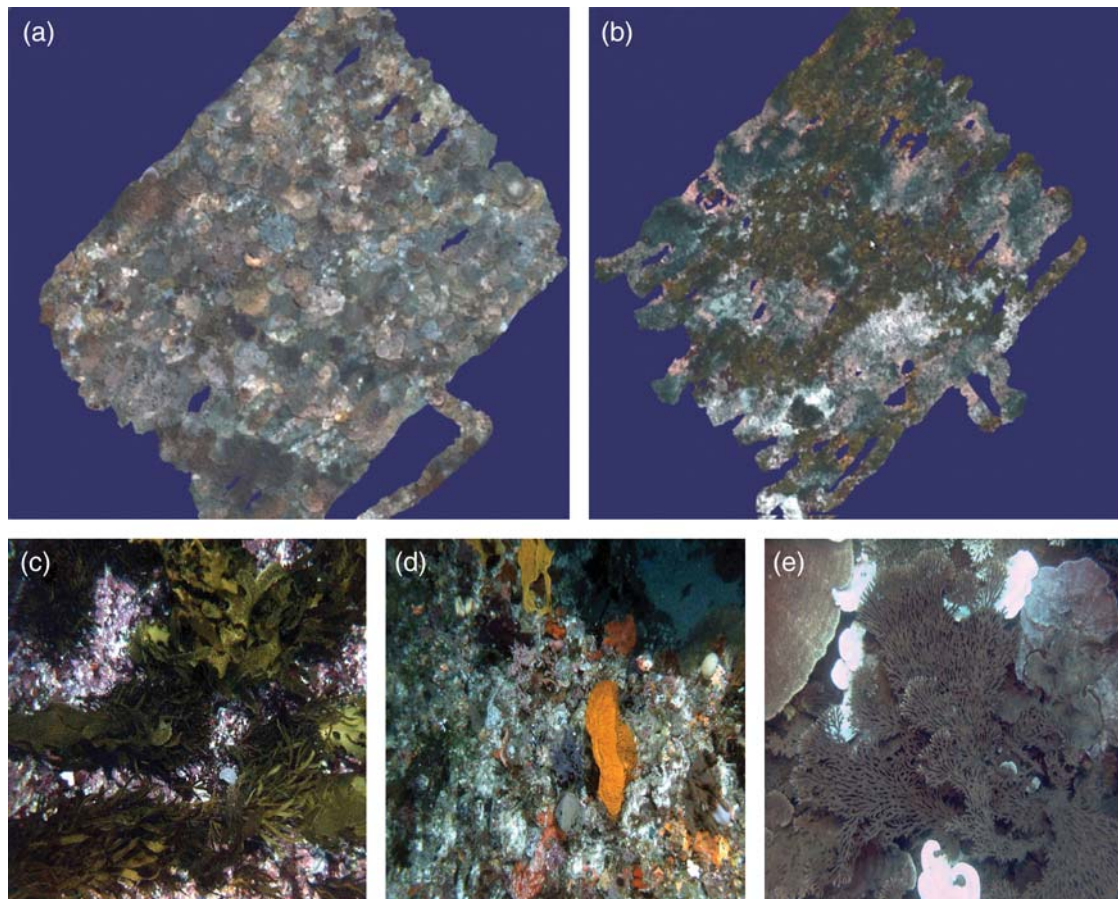


Figure 2. Representative meshes of 25 × 25 m grids sampled at (a) Abrolhos and (b) Rottneest, showing both coral- and kelp-dominated benthic assemblages. Individual images are representative of assemblages dominated by (c) kelp, (d) sessile invertebrates, and (e) hard coral (including bleached corals). Individual images captured ~1.5 × 1.0 m of seabed.

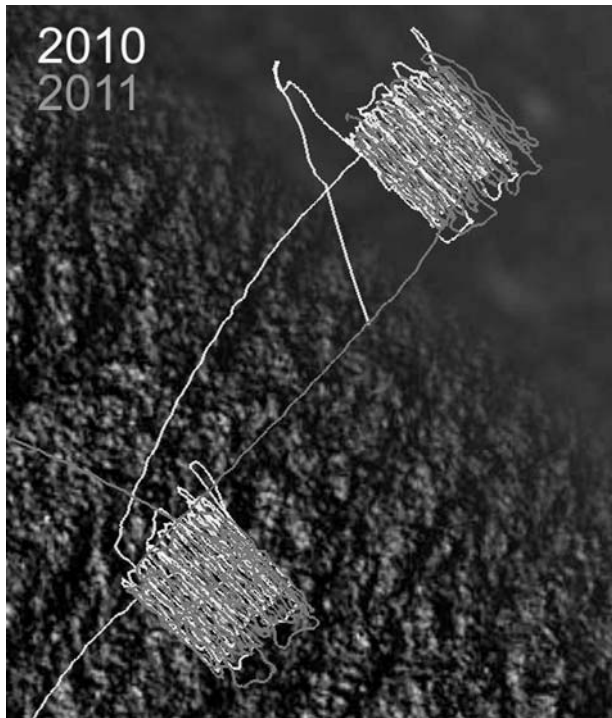


Figure 3. Actual AUV flight paths at the “15 m Coral Patches” reference site at Abrolhos during the 2010 (white) and 2011 (grey) surveys. Each survey grid covered an area of 0.0625 m².

from the images that the methods were inappropriate for sampling mobile fauna (i.e. echinoids, gastropod molluscs), which generally have low abundances and highly patchy distributions, and utilize cryptic habitats (i.e. reef ledges and crevices) in WA (Vanderklift and Kendrick, 2004). Therefore, mobile invertebrates—which were markedly low in abundance—were excluded from the analysis, and assemblage composition was derived from macroalgae and sessile invertebrates. However, in other ecosystems that are characterized by less complex physical habitat structure and higher densities of mobile invertebrates, AUVs have been shown to sample this component of the fauna effectively (e.g. Barrett et al., 2010)

Subsamples were pooled for each grid (means of 101–129 individual images), and analysis of benthic assemblages was conducted using the grids as spatially independent replicated units. Assemblage structure at each reference site was quantified by calculating the per cent cover of dominant benthic groupings. Plots of mean per cent covers of dominant benthic categories at each site were constructed to examine the variability between grids, sites, and years, whereas the significance of differences between years was tested with one-way ANOVA at each location. Data from all sites were pooled and transformed as necessary to reduce homogeneity of variance and improve normality. The power to detect change in the per cent cover of dominant habitat formers at the Abrolhos and Rottneest was also examined. Here, means and standard deviations for dominant benthic groups were used to calculate the power of a hypothetical monitoring programme, where reference sites were surveyed biennially for 8 years (i.e. four subsequent sampling events, with current data representing $t = 0$). The likely power ($\alpha = 0.05$) from a simple linear regression to detect changes in the cover (5–50%) of

dominant space occupiers at each depth stratum at Rottneest and at each site at the Abrolhos was calculated using the method of Dupont and Plummer (1998). The power calculations were performed using the PS power and sample size programme (Dupont and Plummer, 1998).

Finally, at each location, multivariate assemblage structure was visualized with MDS ordinations, constructed from Bray–Curtis similarity matrices based on square-root transformed per cent cover data. Centroids in MDS plots represent averages for each site, derived from two or three grids (except for “25 m Geebank” in 2010, see above). Statistical differences in multivariate assemblage structure between years were tested with ANOSIM using Primer 6.0 software (Clarke and Warwick, 2001).

Results

In 11 days at sea, the AUV captured over 200 000 benthic images, plus data on bathymetry, and a wide range of physiochemical variables which will form the basis of an ongoing time-series. Crucially, using differential GPS, USBL, and image registration technology, the AUV was able to relocate and survey the exact same patch of seabed in both years. For example, comparison of actual AUV flight paths for both years at an Abrolhos site indicated very high spatial overlap between years (Figure 3). Similar spatial overlap between years was observed at all reference sites (except “25 m Geebank”, see Methods).

The Abrolhos were characterized by rich, mixed assemblages dominated by both hard corals and large macroalgae. The “15 m Geebank” site was dominated by hard corals (primarily with branched, tabulate, and plate morphologies) which, on average, covered ~80% of the seabed (Figure 4). In contrast, the other sites were dominated by macroalgae, particularly the kelp *E. radiata* and red foliose algae (Figure 4). For all the dominant space occupiers at the Abrolhos, variability in per cent cover between years was minimal and no significant differences between years were detected (Table 1). At the coral-dominated “15 m Geebank” site, high levels of coral bleaching were recorded in 2011. The per cent cover of bleached corals, derived from both subsampled individual images and composite meshes, ranged from 4 to 19% across the three grids surveyed at “15 m Geebank”. In stark contrast, no coral bleaching was observed at the Abrolhos in 2010 (Figure 4).

At Rottneest, *E. radiata* was the dominant space occupier at all sites and depths (Figure 5) and covered over 50% of the entire seabed surveyed at three sites. Although its relative dominance decreased with depth, the areal coverage of *E. radiata* still exceeded 30% at 40-m depth. Other prevalent components of the benthic assemblage included the large brown macroalgae *S. dorycarpa*, red foliose algae, turfing algae, and encrusting coralline algae (Figure 5). Within-site variability in the per cent cover of these groups was generally low, although variability was markedly higher at the deeper sites. For the dominant space occupiers *E. radiata*, *S. dorycarpa*, red foliose algae, and sponges, variability between years was very low and no significant differences between years were detected (Table 1). For example, on average the per cent cover of *E. radiata* varied by less than 7% between years.

Likely power to detect change in benthic assemblages varied among taxa, and in relation to variables such as depth. At the Abrolhos, the monitoring programme would have adequate power from simple linear regression to detect a 20% change in the cover of *E. radiata*, a 35% change in branched corals, and a

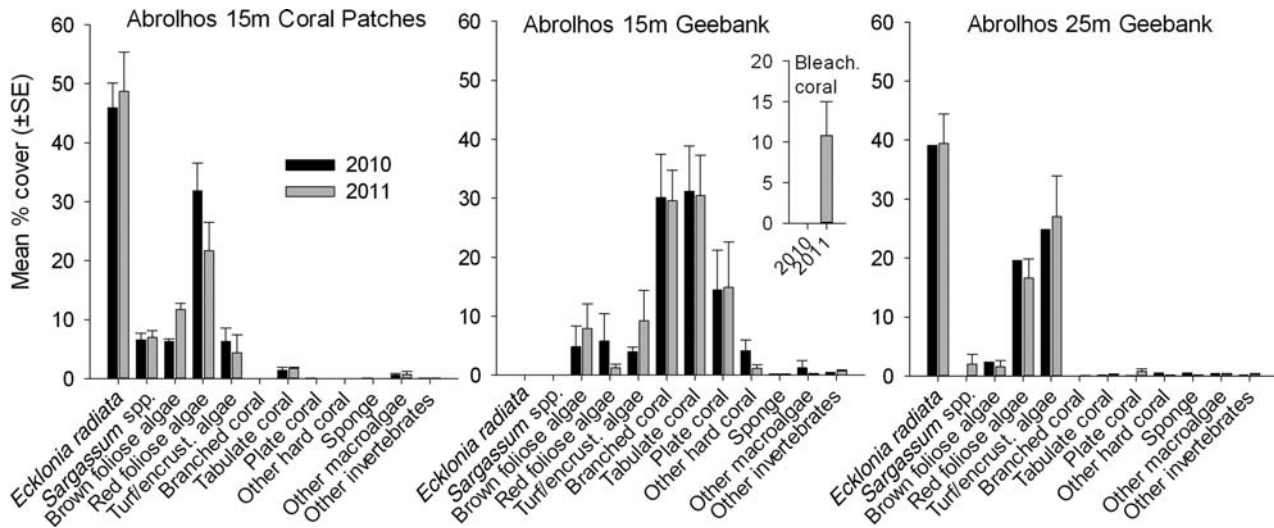


Figure 4. Mean per cent cover (\pm s.e.) of benthic groups at each of reference site at the Houtman Abrolhos Islands, WA. Cover estimates are means of two or three replicate 25×25 m grids within each site, for each survey year. For each grid, >100 individual images were subsampled from complete meshes for detailed analysis. Note that at the “25 m Geebank” site, only one grid was surveyed in 2010 and the three grids surveyed in 2011 did not overlap with the 2010 grid (see Methods).

50% change in tabulate corals after 8 years (Figure 6). At Rottneest, adequate power to detect a 20–25% change in the cover of *E. radiata* was estimated for the 15 and 25 m deep reference sites, whereas at 45 m depth, the monitoring would detect a 45% change in cover of *E. radiata*. Intuitively, greatest power was estimated for indicators with the highest mean values and lowest variability between replicate grids and years (i.e. *E. radiata*).

Multivariate community structure varied considerably between sites at the Abrolhos, with each site supporting very distinct benthic communities (Figure 7). However, variability between years was minimal, and a one-way ANOSIM detected no significant difference in multivariate assemblage between 2010 and 2011 (ANOSIM, $R = 0.00$, $p = 0.95$). At Rottneest, MDS ordination indicated a general shift in assemblage structure along the depth gradient, with moderate variability between sites and years (Figure 7). A one-way ANOSIM between year groups showed that variability in multivariate assemblage structure between years was not significant (ANOSIM, $R = 0.01$, $p = 0.54$), principally because variability within years (and within sites) was relatively pronounced.

In situ temperature loggers showed that daily mean temperature ranged from ~ 20 to 27°C at the Abrolhos and 18 – 26°C at Rottneest during the 13 months of continuous observation (Figure 8). Peak temperatures of 27.5°C at the Abrolhos and 26.1°C at Rottneest were recorded on 1 March 2011. Finally, temperatures in May 2011 were $\sim 2^\circ\text{C}$ higher compared with those recorded in May 2010 at both locations (Figure 8).

Discussion

The surveys described here are the first to be conducted by an AUV system in temperate WA and represent the commencement of a long-term monitoring programme that will inform EBFM in the region. There were four key benefits of the application of the AUV technology for sustained ecological observation. First, each reference site is precisely geo-referenced and can be revisited by pre-programming the flight path of the AUV. Second, each long-term monitoring grid is surveyed at $\sim 100\%$ areal coverage,

Table 1. One-way ANOVAs on the per cent cover of major benthic groupings at both locations, to test for differences between years.

Abrolhos	Trans.	MS	$F_{1,13}$	p-value
<i>Ecklonia radiata</i>	$\ln(x + 1)$	0.14	0.03	0.852
Brown foliose algae	$\ln(x + 1)$	0.01	0.01	0.958
Red foliose algae	$\ln(x + 1)$	0.68	0.45	0.511
Branched coral ^a	$\ln(x + 1)$	0.08	0.02	0.874
Tabulate coral	$\ln(x + 1)$	0.19	0.08	0.779
Plate coral ^a	$\ln(x + 1)$	0.01	0.01	0.968
Rottneest	Trans.	MS	$F_{1,30}$	p-value
<i>Ecklonia radiata</i>	None	50.15	0.35	0.555
<i>Scytothalia dorycarpa</i>	$\ln(x + 1)$	0.93	1.53	0.226
Red foliose algae	None	28.10	0.62	0.434
Turf/encrust. algae	$\ln(x + 1)$	0.02	0.07	0.794
Sponges	$\ln(x + 1)$	0.03	0.06	0.805

^aTaxa that did not pass normality tests after transformation.

generating precise ecological information for each 625 m^2 patch of seabed. The amount of data collected allows small-scale patchiness, which is pronounced in these ecosystems (Smale *et al.*, 2010b), to be accounted for, thereby facilitating monitoring of community structure at relevant spatial scales (i.e. scale of “patch”). Third, the AUV can resample the same specific organisms over time, with minimal risk of causing physical disturbance during the sampling procedure. This is particularly relevant to coral reef monitoring, where tagged colonies are revisited by scuba divers to monitor bleaching and mortality rates (Hill and Wilkinson, 2004). This procedure can be time consuming, while scuba divers may damage tagged corals and are severely restricted by depth and time. Using an AUV, specific organisms can be sampled over time at a range of depths, without the need to physically tag colonies beforehand and without the risk of causing physical damage. Finally, the AUV collected a range of physiochemical data that can, over time, be related to biological structure at fine spatial scales. For example, high-resolution multibeam data could be used to derive secondary physical variables (e.g. rugosity,

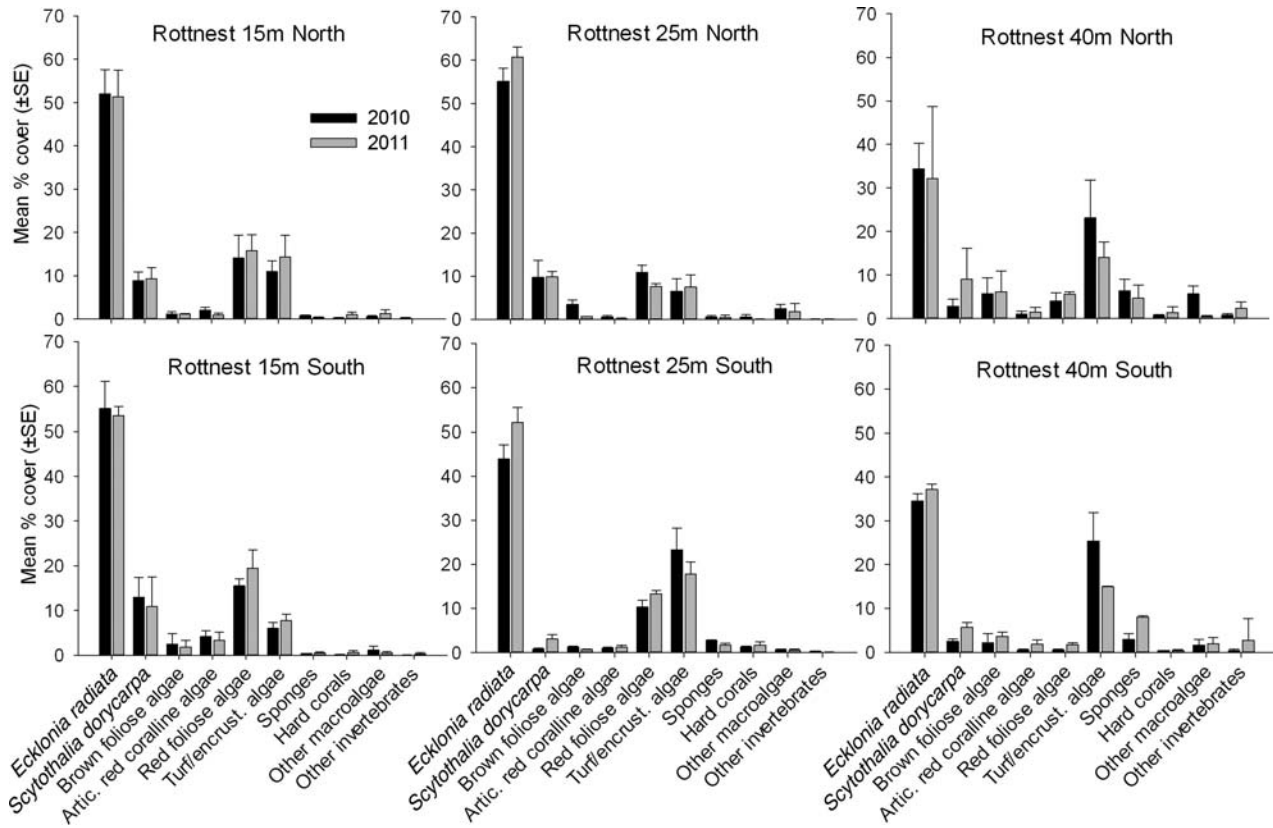


Figure 5. Mean per cent cover (\pm s.e.) of benthic groups at each of reference site at the Rottnest Island, WA. Cover estimates are means of two or three replicate 25×25 m grids within each site, for each survey year. For each grid, >100 individual images were subsampled from complete meshes for detailed analysis.

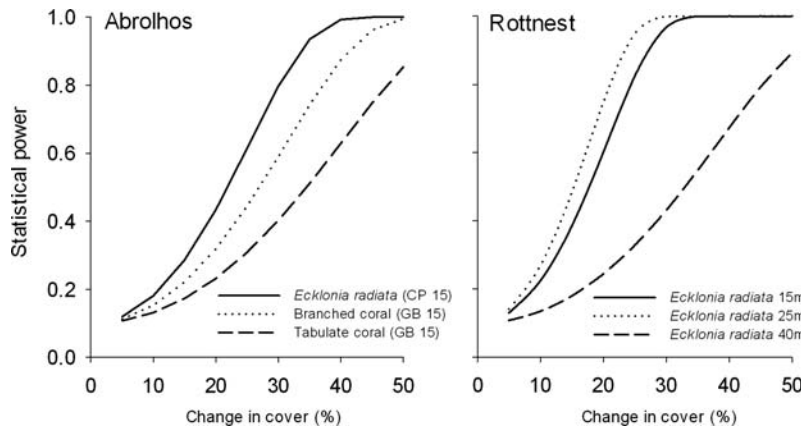


Figure 6. Estimates of power to detect change in per cent cover of dominant habitat formers at each site at Abrolhos and each depth increment at Rottnest. Power calculations are based on a simple linear regression of five biennial sampling events, with means and standard deviations representing $t = 0$ generated from the current survey data.

aspect, depth) that could explain small-scale variability in coral bleaching. The data presented here were collected from the shallow-water photic zone, to a maximum depth of ~ 40 m and within depths attainable by scuba divers and drop cameras. Another advantage of the AUV technology, however, is the ability to sample at greater depths with minimal additional cost or effort. For example, similar AUV systems have been used to survey benthic communities on coral mounds at 800-m depth

(Grasmueck *et al.*, 2006) and groundfish to depths of 500 m (Clarke *et al.*, 2009). Although these benefits are not necessarily exclusive to the AUV technology (other tools have precise geo-referencing capabilities, for example), it is the combination of these benefits and the efficiency and flexibility of sampling by AUV that will facilitate sustained ecological observations.

The AUV is a powerful tool for assessing the quantity and quality of benthic habitat for fisheries management, and far

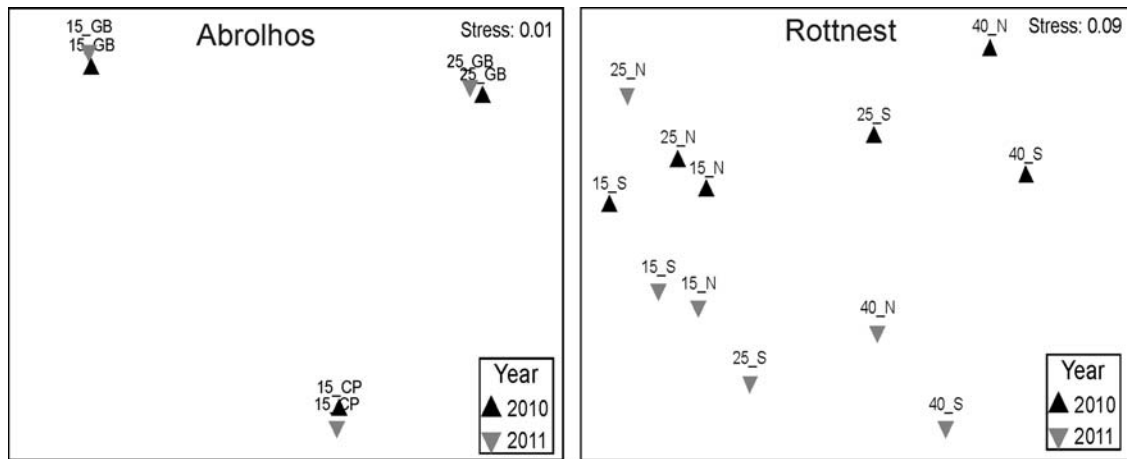


Figure 7. MDS ordinations, based on Bray–Curtis similarity matrices generated from square-root transformed per cent cover data, to indicate assemblage structure at each reference site and each sampling year at Abrolhos (left) and Rottnest (right), WA. Each centroid represents the average of two or three grids (25 × 25 m) sampled at each site (except “25 m Geebank” in 2010, see Methods). Low stress values (<0.1) indicate adequate representation of multidimensional distance in the two-dimensional MDS plots.

exceeds the ability of other commonly used methods to obtain large amounts of quantitative photographic data of the benthos. For example, photographic samples have been collected previously from submerged reefs in the region, by scuba divers (Bellchambers *et al.*, 2008), “drop” cameras (Smale *et al.*, 2010a), and towed video systems. In a single day in the field, a team of divers can typically collect ~400 individual (non-overlapping) images and a remotely deployed “drop” camera can collect ~700 images, depending on weather conditions and sampling protocol. During the surveys described here, the AUV collected >15 000 stereo image pairs each day, a considerable increase in efficiency. Therefore, while initial expenditure on an AUV is greater (an AUV suitable to conduct the monitoring programme would cost a total of ~\$300 000), there are considerable reductions in cost relating to time at sea and personnel training (i.e. for scientific divers). Conversely, additional costs relating to expertise needed to operate the vehicle and maintenance may be higher than those associated with the other methods and would need to be considered in any formal analysis of cost-effectiveness.

Benthic assemblage structure at the Abrolhos and Rottnest was quantified to provide a benchmark against which to detect future ecological change. The coexistence of coral- and kelp-dominated assemblages at the Abrolhos and, to a lesser extent, Rottnest provides a rare opportunity to test ecological hypotheses related to species interactions and oceanic warming, whereas the relative dominance of these habitat formers may represent a useful early warning signal for detecting ecological effects of seawater warming (Smale *et al.*, 2010a). Here, continued sampling at the Abrolhos will have adequate power (>0.80) to detect <35% change in cover of kelps and branching corals (i.e. principally Acroporids) over time. As such, the power of AUV sampling to detect change in the cover of these key indicators bodes well for effective monitoring of these sensitive and important habitats.

The AUV surveys detected high rates of coral bleaching (>10% of the entire seabed area) at one of the Abrolhos sites 2011, whereas no bleached corals were recorded in 2010. In early 2011, most of the Western Australian coastline experienced a “marine heat wave”, in that seawater temperatures were 2–4°C higher than normal for up to 8 weeks (Pearce *et al.*, 2011; Smale and Wernberg, 2012). Satellite-derived SSTs were the highest on

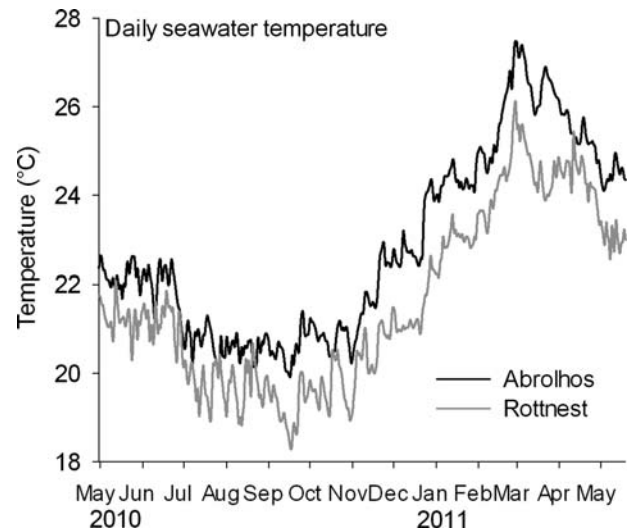


Figure 8. Daily mean temperature at monitoring sites at Abrolhos and Rottnest during 2010 and 2011. Temperature data were collected (every 30 min) by *in situ* loggers deployed onto rocky reefs at ~25 m depth.

record (i.e. for 30 years), while *in situ* loggers recorded warming anomalies of up to 5°C in inshore waters (Pearce *et al.*, 2011). Over 2000 km of coastline was subjected to the warming event, which was principally driven by strong *La Niña* conditions superimposed onto a decadal-scale seawater warming trend (Pearce *et al.*, 2011). At the Abrolhos, surface seawater temperature peaked at 28.7°C in March, which is ~5°C higher than the long-term monthly mean for the region. Even at 25-m depth at the Geebank monitoring site, seawater temperature exceeded 27°C. The high incidence of bleached corals was a direct consequence of persistent seawater warming, and ongoing regular AUV surveys will facilitate assessments of the longer-term ecological consequences of the bleaching event. Understanding the levels of resilience and rates of recovery of these coral assemblages will become critically important, if, as predicted, the frequency and intensity of short-term extreme warming events increases (Solomon *et al.*, 2007).

At Rottneest Island, rocky reefs are dominated by *E. radiata*, which decreases in cover with depth and consequently drives depth-related changes in overall assemblage structure. Even at 40-m depth, however, the kelp was the dominant space occupier, which highlights the importance of this primary producer in near-shore benthic habitats. Crucially, the analysis of a large number of subsamples (i.e. >100) at each grid generated realistic and precise estimates of the per cent cover for dominant, conspicuous space occupiers, which can be tracked over time to provide powerful tests of ecological change. By monitoring at the scale of grid or “patch” along a depth gradient, changes in kelp canopy structure can be assessed. Given the possible interactive effects of seawater warming and increased physical disturbance (through enhanced storm frequency and/or intensity) on kelp beds and local biodiversity (Wernberg *et al.*, 2010, 2011a), observations at the spatial scale of patch will be most appropriate in detecting the effects of climate change in the region.

Rottneest Island was also affected by the 2011 marine heat wave, with *in situ* measurements confirming temperatures were up to 4°C higher (maximum = 26.4°C) than the long-term monthly average (21.7°C) for 20 weeks before surveys (Thomson *et al.*, 2011). Ecological observations attributed to the temperature anomaly included restricted coral bleaching (Thomson *et al.*, 2011) and the possible enhanced recruitment of tropical fish species (Pearce *et al.*, 2011). No significant changes in the cover of dominant space occupiers were recorded in the AUV surveys, although, based on their distribution patterns, some cool-water macroalgae species (e.g. *S. dorycarpa*) are thought to be useful ecological indicators of seawater warming in the region (Smale *et al.*, 2010a; Wernberg *et al.*, 2011b). However, as these species are not at the “warm” extreme of their distributions at Rottneest Island, it seems likely that they were able to tolerate the warming event, and there is some evidence that marginal populations of cool-affinity macroalgae further north did exhibit a response to warming (D.A. Smale, unpublished data).

Indicators of ecosystem change derived from multivariate assemblage composition are often more sensitive to ecologically relevant change than univariate indicators (Anderson and Thompson, 2004) and provide a more holistic view of ecological structure than would be gained by targeting individual indicators species or taxa (Clarke and Warwick, 2001). At the Abrolhos, differences in multivariate structure between sites were pronounced, with each reference site supporting very distinct assemblages. Interannual variability was negligible, indicating that short-term variability in benthic assemblage structure was minimal and providing a robust “baseline” against which to monitor change. At Rottneest, MDS ordination suggested a general shift in assemblage structure along the depth gradient, which was driven by decreasing dominance of *E. radiata* and greater occurrence of invertebrate taxa, such as sponges, with increased depth. Over time, multivariate data can be used to develop powerful statistical tools such as “control charts” (Anderson and Thompson, 2004), which can be used to inform resource managers of acceptable levels of natural variability and to trigger an “alarm bell” on the detection of unacceptable levels of ecological change. The detection of change could subsequently invoke various adaptation and mitigation strategies (see Koehn *et al.*, 2011, for further discussion).

Without long-term data, short-term natural variability can mask chronic or cumulative impacts, often until critical levels are reached (Hewitt *et al.*, 2001). The necessity of long-term monitoring for understanding ecological variation and processes has

been acknowledged for decades (Bernstein and Zalinski, 1983). Monitoring for EBFM is essential because of the uncertainty involved in predicting the response of ecosystems to particular impacts or environmental change. Sustained AUV surveys at these locations will facilitate greater understanding of temporal variability in benthic community structure and population dynamics of key habitat-forming species, such as kelps and corals. In the immediate term, sampling each year is desirable to determine the magnitude of interannual variability, but eventually sampling frequency could be reduced or nested (e.g. Hewitt and Thrush, 2007) to achieve cost-effectiveness while retaining sufficient power to detect change. Ultimately, the monitoring programme described here for WA will form part of a national incentive to establish reference sites around the entire >25 000 km of coastline (Williams *et al.*, 2012). This will facilitate greater understanding of the ecological pattern and process in subtidal habitats and provide better information for resource managers on ecosystem responses to key drivers such as fishing and climate change.

Supplementary data

Supplementary data are available at ICESJMS online.

Acknowledgements

This work forms part of a large, collaborative effort involving the University of Western Australia (UWA), CSIRO, the Department of Fisheries (Western Australia), and the University of Sydney’s Australian Centre for Field Robotics (ACFR). The Western Australian Marine Science Institute (WAMSI) and the Integrated Marine Observing System (IMOS) marine facility provided funding. The field surveys would not have been possible without the expertise and commitment of a number of scientists and technicians; we are especially grateful to the skippers and crews of the “RV Linnaeus” and the “RV Naturaliste”.

References

- Anderson, M. J., and Thompson, A. A. 2004. Multivariate control charts for ecological and environmental monitoring. *Ecological Applications*, 14: 1921–1935.
- Armstrong, R. A., Singh, H., Torres, J., Nemeth, R. S., Can, A., Roman, C., Eustice, R., *et al.* 2006. Characterizing the deep insular shelf coral reef habitat of the Hind Bank marine conservation district (US Virgin Islands) using the Seabed autonomous underwater vehicle. *Continental Shelf Research*, 26: 194–205.
- Barrett, N., Seiler, J., Anderson, T., Williams, S., Nichol, S., and Hill, S. N. 2010. Autonomous Underwater Vehicle (AUV) for mapping marine biodiversity in coastal and shelf waters: implications for marine management. OCEANS 2010 IEEE Conference, Sydney, pp. 1–6.
- Bellchambers, L., Bridgwood, S., How, J., Lewis, P., de Lestang, S., Mackie, M., and Coutts, T. 2008. Development of a long-term program to monitor coastal communities within the Swan region. Fisheries Research Report, 183. Department of Fisheries, Western Australia.
- Bernstein, B. B., and Zalinski, J. 1983. Optimum sampling design and power tests for environmental biologists. *Journal of Environmental Management*, 16: 35–43.
- Blomqvist, S. 1991. Quantitative sampling of soft-bottom sediments: problems and solutions. *Marine Ecology Progress Series*, 72: 295–304.
- Camilli, R., Reddy, C. M., Yoerger, D. R., Van Mooy, B. A. S., Jakuba, M. V., Kinsey, J. C., McIntyre, C. P., *et al.* 2010. Tracking

- hydrocarbon plume transport and biodegradation at Deepwater Horizon. *Science*, 330: 201–204.
- Clarke, K. R., and Warwick, R. M. 2001. *Change in Marine Communities: an Approach to Statistical Analysis and Interpretation*. PRIMER-E Ltd, Plymouth, UK.
- Clarke, M. E., Tolimieri, N., Singh, H., Beamish, R. J., and Rothschild, B. J. 2009. Using the seabed AUV to assess populations of groundfish in untrawlable areas. *In* *The Future of Fisheries Science in North America*, pp. 357–372. Springer, Netherlands.
- Dayton, P. K., Currie, V., Gerrodette, T., Keller, B. D., Rosenthal, R., and Ven Tresca, D. 1984. Patch dynamics and stability of some California kelp communities. *Ecological Monographs*, 54: 253–289.
- Dupont, W. D., and Plummer, W. D. 1998. Power and sampling size calculations for studies involving linear regression. *Controlled Clinical Trials*, 19: 589–601.
- Fletcher, W. J., Shaw, J., Metcalf, S. J., and Gaughan, D. J. 2010. An ecosystem based fisheries management framework: the efficient, regional-level planning tool for management agencies. *Marine Policy*, 34: 1226–1238.
- Gaughan, D. J. 2007. Potential mechanisms of influence of the Leeuwin Current eddy system on teleost recruitment to the West Australian continental shelf. *Deep Sea Research Part II: Topical Studies in Oceanography*, 54: 1129–1140.
- Grasmueck, M., Eberli, G. P., Viggiano, D. A., Correa, T., Rathwell, G., and Luo, J. 2006. Autonomous underwater vehicle (AUV) mapping reveals coral mound distribution, morphology, and oceanography in deep water of the Straits of Florida. *Geophysical Research Letters*, 33: L23616.
- Hatcher, B. G. 1985. Ecological research at the Houtman Abrolhos Islands: High latitude reefs of Western Australia. *In* *Proceedings of the 5th International Coral Reef Congress, Tahiti*, 6, pp. 291–297. Ed. by C. Gabrie, and M. Harmelin Vivien. Tahiti, French Polynesia: Antenne Museum-EPHE.
- Hewitt, J., and Thrush, S. 2007. Effective long-term ecological monitoring using spatially and temporally nested sampling. *Environmental Monitoring and Assessment*, 133: 295–307.
- Hewitt, J. E., Thrush, S. E., and Cummings, V. J. 2001. Assessing environmental impacts: effects of spatial and temporal variability at likely impact scales. *Ecological Applications*, 11: 1502–1516.
- Hill, J., and Wilkinson, C. 2004. *Methods for Ecological Monitoring of Coral Reefs*. Australian Institute of Marine Science, Townsville, 117pp.
- Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*, 54: 187–211.
- Johnson-Roberson, M., Pizarro, O., Williams, S. B., and Mahon, I. 2010. Generation and visualization of large-scale three-dimensional reconstructions from underwater robotic surveys. *Journal of Field Robotics*, 27: 21–51.
- Koehn, J. D., Hobday, A. J., Pratchett, M. S., and Gillanders, B. M. 2011. Climate change and Australian marine and freshwater environments, fishes and fisheries: synthesis and options for adaptation. *Marine and Freshwater Research*, 62: 1148–1164.
- Kohler, K. E., and Gill, S. M. 2006. Coral Point Count with Excel extensions (CPCe): a Visual Basic program for the determination of coral and substrate coverage using random point count methodology. *Computers and Geosciences*, 32: 1259–1269.
- Langlois, T. J., Harvey, E. S., Fitzpatrick, B., Shedrawi, J. J. M. G., and Watson, D. L. 2010. Cost-efficient sampling of fish assemblages: comparison of baited video stations and diver video transects. *Aquatic Biology*, 9: 155–168.
- Langlois, T. J., Harvey, E. S., and Meeuwig, J. J. Strong direct and inconsistent indirect effects of fishing found using stereo-video: testing indicators from fisheries closures. *Ecological Indicators* (In press.)
- Langlois, T. J., Radford, B., Van Niel, K. P., Meeuwig, J. J., Pearce, A., Rousseaux, C. S. G., Kendrick, G. A., *et al.* 2012. Consistent abundance distributions of marine fishes in an old, climatically buffered, infertile seascape. *Global Ecology and Biogeography*. doi: 10.1111/j.1466-8238.2011.00734.x.
- Lough, J. M., and Hobday, A. J. 2011. Observed climate change in Australian marine and freshwater environments. *Marine and Freshwater Research*, 62: 984–999.
- Niemi, G. J., and McDonald, M. E. 2004. Application of ecological indicators. *Annual Review of Ecology, Evolution, and Systematics*, 35: 89–111.
- Oliver, I., and Beattie, A. J. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. *Ecological Applications*, 6: 594–607.
- Olsgard, F., Brattegard, T., and Holthe, T. 2003. Polychaetes as surrogates for marine biodiversity: lower taxonomic resolution and indicator groups. *Biodiversity and Conservation*, 12: 1033–1049.
- Orr, J. C., Fabry, V. J., Aumont, O., Bopp, L., Doney, S. C., Feely, R. A., Gnanadesikan, A., *et al.* 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature*, 437: 681–686.
- Pearce, A. 1997. The Leeuwin Current and the Houtman Abrolhos Islands. *In* *The Marine Flora and Fauna of the Houtman Abrolhos Islands, Western Australia*. Ed. by F. E. Wells. Western Australian Museum, Perth.
- Pearce, A., and Feng, M. 2007. Observations of warming on the Western Australian continental shelf. *Marine and Freshwater Research*, 58: 914–920.
- Pearce, A., Lenanton, R., Jackson, G., Moore, J., Feng, M., and Gaughan, D. 2011. The “marine heat wave” off Western Australia during the summer of 2010/11. *Fisheries Research Report*, 222. Department of Fisheries, Perth, Australia.
- Pikitch, E. K., Santora, C., Babcock, E. A., Bakun, A., Bonfil, R., Conover, D. O., Dayton, P., *et al.* 2004. Ecosystem-based fishery management. *Science*, 305: 346–347.
- Pitcher, T. J., Kalikoski, D., Short, K., Varkey, D., and Pramod, G. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. *Marine Policy*, 33: 223–232.
- Ray, H. 2011. Future directions in ecosystem based fisheries management: a personal perspective. *Fisheries Research*, 108: 235–239.
- Sherman, A. D., and Smith, K. L., Jr. 2009. Deep-sea benthic boundary layer communities and food supply: a long-term monitoring strategy. *Deep Sea Research Part II: Topical Studies in Oceanography*, 56: 1754–1762.
- Singh, H., Armstrong, R., Gilbes, F., Eustice, R., Roman, C., Pizarro, O., and Torres, J. 2004. Imaging coral I: imaging coral habitats with the SeABED AUV. *Subsurface Sensing Technologies and Applications*, 5: 25–42.
- Sliwa, C., Migus, S., McEnnulty, F., Hayes, K. R., Rilov, G., and Crooks, J. A. 2009. Marine bioinvasions in Australia. *In* *Biological Invasions in Marine Ecosystems*, pp. 425–437. Ed. by M. M. Caldwell, G. Heldmaier, R. B. Jackson, O. L. Lange, H. A. Mooney, E. D. Schulze, and U. Sommer. Springer, Berlin.
- Smale, D. A., and Childs, S. 2012. The occurrence of a widespread marine invader, *Didemnum perlucidum* (Tunicata, Ascidiacea) in Western Australia. *Biological Invasions*, doi: 10.1007/s10530-011-0167-8.
- Smale, D. A., Kendrick, G. A., Waddington, K. I., Van Niel, K. P., Meeuwig, J. J., and Harvey, E. S. 2010a. Benthic assemblage composition on subtidal reefs along a latitudinal gradient in Western Australia. *Estuarine Coastal and Shelf Science*, 86: 83–92.
- Smale, D. A., Kendrick, G. A., and Wernberg, T. 2010b. Assemblage turnover and taxonomic sufficiency of subtidal macroalgae at multiple spatial scales. *Journal of Experimental Marine Biology and Ecology*, 384: 76–86.
- Smale, D. A., Langlois, T. J., Kendrick, G. A., Meeuwig, J. J., and Harvey, E. S. 2011. From fronds to fish: the use of indicators for ecological monitoring in marine benthic ecosystems, with

- particular reference to the temperate nearshore system of Western Australia. *Reviews in Fish Biology and Fisheries*, 21: 311–337.
- Smale, D. A., and Wernberg, T. 2012. Ecological observations associated with an anomalous warming event at the Houtman Abrolhos Islands, Western Australia. *Coral Reefs*, 31: 441.
- Smith, A. D. M., Fulton, E. J., Hobday, A. J., Smith, D. C., and Shoulder, P. 2007. Scientific tools to support the practical implementation of ecosystem-based fisheries management. *ICES Journal of Marine Science*, 64: 633–639.
- Smith, R. L., Huyer, A., Godfrey, J. S., and Church, J. A. 1991. The Leeuwin Current off Western Australia. *Journal of Physical Oceanography*, 21: 323–345.
- Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., *et al.* 2007. Contribution of Working Group 1 to the Fourth Assessment Report of the IPCC. Cambridge University Press, Cambridge, UK and New York, USA.
- Thomson, D. P., Bearham, D., Graham, F., and Eagle, J. V. 2011. High latitude, deeper water coral bleaching at Rottneest Island, Western Australia. *Coral Reefs*, 30: 1107.
- Vanderklift, M. A., and Kendrick, G. A. 2004. Variations in abundances of herbivorous invertebrates in temperate subtidal rocky reef habitats. *Marine and Freshwater Research*, 55: 93–103.
- Waddington, K. I., and Meeuwig, J. J. 2009. Contribution of bait to lobster production in an oligotrophic marine ecosystem as determined using a mass balance model. *Fisheries Research*, 99: 1–6.
- Watson, D. L., Harvey, E. S., Kendrick, G. A., Nardi, K., and Anderson, M. J. 2007. Protection from fishing alters the species composition of fish assemblages in a temperate-tropical transition zone. *Marine Biology*, 152: 1197–1206.
- Webster, F. J., Dibden, C. J., Weir, K. E., and Chubb, C. F. 2002. Towards an assessment of the natural and human use impacts on the marine environment of the Abrolhos Islands, vol. 1. Fisheries Research Report, 134. Department of Fisheries, Western Australia. 120 pp.
- Wernberg, T., Russell, B. D., Moore, P. J., Ling, S. D., Smale, D. A., Coleman, M., Steinberg, P. D., *et al.* 2011a. Impacts of climate change in a global hotspot for temperate marine biodiversity and ocean warming. *Journal of Experimental Marine Biology and Ecology*, 400: 7–16.
- Wernberg, T., Thomsen, M. S., Tuya, F., and Kendrick, G. A. 2011b. Biogenic habitat structure of seaweeds change along a latitudinal gradient in ocean temperature. *Journal of Experimental Marine Biology and Ecology*, 400: 264–271.
- Wernberg, T., Thomsen, M. S., Tuya, F., Kendrick, G. A., Staehr, P. A., and Toohey, B. D. 2010. Decreasing resilience of kelp beds along a latitudinal temperature gradient: potential implications for a warmer future. *Ecology Letters*, 13: 685–694.
- Williams, S. B., Pizarro, O., Jakuba, M. V., Johnson, C. R., Barrett, N., Babcock, R. C., Kendrick, G. A., *et al.* 2012. Monitoring of benthic reference sites. *IEEE Robotics and Automation Magazine*, 19: 73–84.

Handling editor: David Demer