

## ORIGINAL RESEARCH

# Regional estimates of a range-extending ecosystem engineer using stereo-imagery from ROV transects collected with an efficient, spatially balanced design

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## Keywords

Balanced adaptive sampling, invasive species monitoring, remotely operated vehicles, robust monitoring, stereo video, surveying strategies

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## Abstract

The redistribution of marine ecosystem engineers in response to changing climate is restructuring endemic benthic communities globally. Therefore, developing and implementing efficient monitoring programs across the complete depth range of these marine ecosystem engineers is often an urgent management priority. Traditionally, many monitoring programs have been based on a systematically selected set of survey locations that, while able to track trends at those sites through time, lack inference for the overall region being monitored. This study trialled a probabilistic sampling design to address this need, taking advantage of an important prerequisite for such designs, extensive multibeam echosounder (MBES) mapping, to inform a spatially balanced sample selection. Here, we allocated 170 remotely operated vehicles (ROVs) transects based on a spatially balanced probabilistic sampling design across three locations with extensive mapping. Generalized additive models were used to estimate the density and associated barren cover of the range-expanding ecosystem engineer, the long spined urchin (*Centrostephanus rodgersii*). Estimates were generated at a reef-wide scale across three locations on the east coast of Tasmania, Australia, representing the leading edge of the species recent range extension. Model-based estimates of urchin density and barren cover incorporated seabed structure attributes, such as depth and ruggedness, with differences in these modelled relationships being identified between locations. Estimates ranged from 0.000065 individuals  $m^{-2}$  and 0.018% barren cover in the Tasman Peninsula to 0.167 individuals  $m^{-2}$  and 2.10% barren cover at Governor Island Marine Reserve, reflecting a north to south distributional gradient. This study highlights the value of combining probabilistic sampling designs, ROV transects, stereo video, and MBES mapping to generate reliable and robust estimates of important ecosystem species needed to protect reef-based fishery and conservation values via adaptive and informed management.

## Introduction

In the marine environment, south-eastern Australia is recognized as a climate change hotspot (Johnson et al., 2011; Last et al., 2011; Wernberg et al., 2011). Climate change-driven intensification and expansion of the East Australian Current has facilitated the transportation of warm, nutrient-poor water down the east coast of Tasmania (Frusher et al., 2013; Johnson et al., 2011; Last et al., 2011; Oliver et al., 2017; Oliver et al., 2018). This has resulted in a change in the distribution of species, particularly those

limited by a developmental temperature threshold, such as the long-spined urchin (*Centrostephanus rodgersii*) which have extended south beyond their historic range (Banks et al., 2010; Ling & Keane, 2018; Perkins et al., 2015; Verges et al., 2014). Intense and persistent overgrazing by urchins outcompetes native species for food (Andrew & Underwood, 1992) and modifies the biogenic habitat structure to form barrens absent of macroalgae (Perkins et al., 2015). These barren habitats have many negative impacts, including a change in the composition of demersal fish assemblages (e.g. decline in abundance in herbivorous

fishes; Jones & Andrew, 1990) and reduced abundances of commercially important macroinvertebrates (e.g. rock lobster and abalone; Ling et al., 2009; Ling & Keane, 2018; Strain & Johnson, 2009). Providing robust quantitative assessments of urchins and barren cover at a regional scale may improve the ability to plan for, detect and respond to the early signs of *C. rodgersii* establishment into Tasmanian waters before the onset of difficult-to-reverse ecosystem shifts of macroalgae into barren habitat (Filbee-Dexter & Scheibling, 2014; Ling et al., 2014).

Historical monitoring of *C. rodgersii* and their associated barren cover has largely been undertaken using SCUBA diver-based methods (Andrew & Underwood, 1989, 1993; Ling et al., 2008). However, urchins in Tasmania are present at depths greater than those safely accessible by divers (>35 m; Perkins et al., 2015) resulting in the need to explore alternative platforms for monitoring this species. Underwater towed video and autonomous underwater vehicles have since been used to monitor urchin density and barren cover at these greater depths (Johnson et al., 2005; Ling & Keane, 2018; Perkins et al., 2015). These monitoring platforms require little or no human interaction at the expense of increased entanglement risk in kelp-dominated environments due to their inability to navigate canopies. Towed video systems circumvent this risk with live monitoring of video footage, but these systems lack the dexterity of being able to survey complex seafloor at a consistent elevation. A remotely operated vehicle (ROV) potentially combines the benefits of being able to maintain a consistent elevation, as well as providing a live feed to decrease entanglements across the entire urchin depth and habitat range. ROVs have been trialled elsewhere for monitoring urchin density (Crook & Davoren, 2016), but have yet to be tested in Tasmania despite this potential benefit.

Another fundamental aspect of quantifying the density of urchins and barren cover is the sampling design. Ideally, sampling designs for accurately quantifying the extent of barrens at regional scales need adequate replication and should consider the spatial heterogeneity of reef systems within the region. Despite this, previous urchin studies using SCUBA divers and towed video have included haphazardly or strategically selected sample sites as part of their sampling design (e.g. Ling & Keane, 2018). While such approaches can be sufficient to track the general magnitude of trends through time, they may not be reflective of the broader region, and reflect site selection bias (Briscoe et al., 2019). Newly developed spatially balanced sampling approaches provide mechanisms to improve the generality of abundance and percentage cover estimates of benthic and mobile marine organisms over a range of spatial scales when sufficient sampling replication is possible and comprehensive multibeam

echosounder (MBES) mapping is available (Foster et al., 2017; Foster et al., 2018; Hill et al., 2018; Lawrence et al., 2015; Monk et al., 2018). One such approach is balanced adaptive sampling (BAS) with unequal inclusion probabilities (Foster, 2020), which reduces spatial correlation between the observations and approximately balances sampling on many environmental gradients (Thompson, 2012). Accordingly, this approach is thought to provide more information per unit of sampling effort than with simple random sampling (Foster et al., 2019; Robertson et al., 2013).

Here, we apply a BAS approach to quantify urchins and the extent of barren habitat along Tasmania's east to southeast coast where barrens are currently expanding (Johnson et al., 2005). Sampling was undertaken across a gradient of reported barren density from Governor Island Marine Reserve (GIMR) and Butler's Point, two regions on the north-east coast where barrens have reportedly become extensive, to the Tasman Peninsula, a broad-scale region on the south-east coast that is reported to have urchins but low barren cover. The BAS approach takes advantage of existing comprehensive MBES mapping of these locations to preferentially select study sites with urchin-associated reef habitat and to provide balanced sampling of all depth strata at each location. This study intends to (1) evaluate ROV-based stereo video sampling within a probabilistic sampling framework for generating reliable and generalisable quantitative estimates of urchin density and barren cover across the range of depths found within temperate reef systems in these regions (5–105 m), (2) further our ecosystem-based understanding of barren development, and (3) inform future monitoring efforts on the expansion of urchins along the east coast of Tasmania.

## Materials and Methods

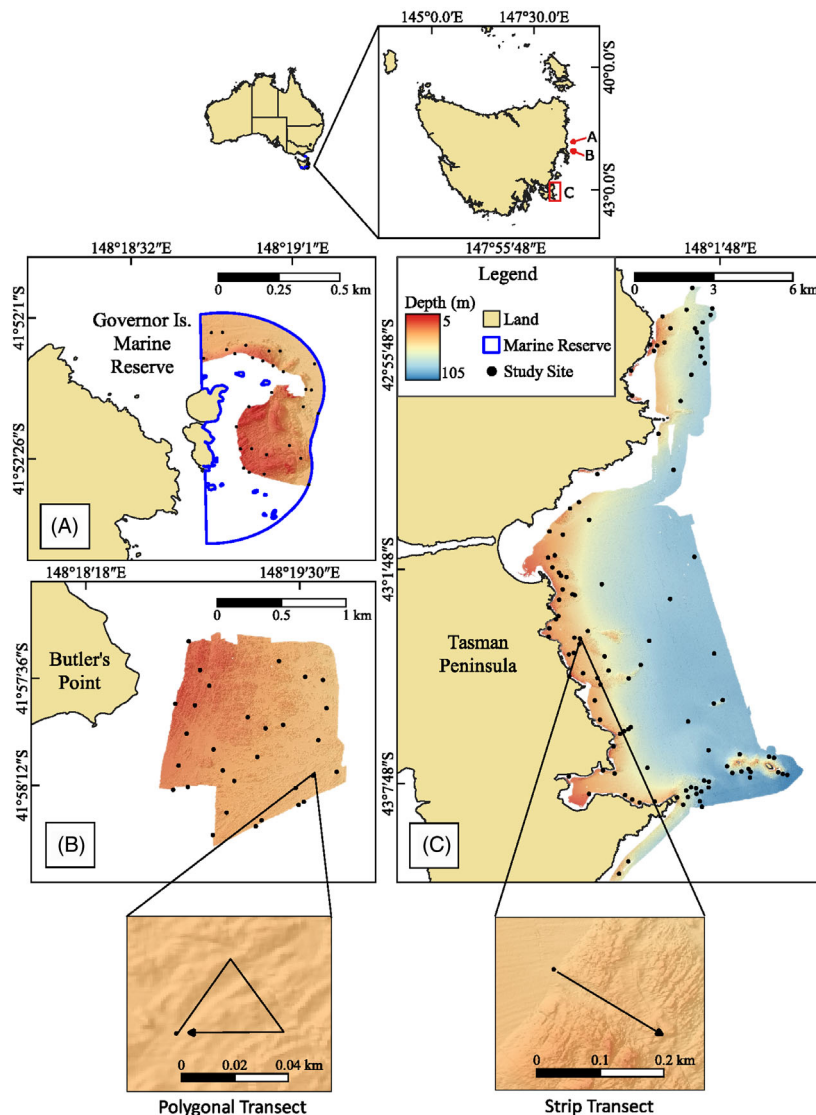
### Study locations

Fieldwork was undertaken at three locations on the east coast of Tasmania, Australia, chosen primarily due to their extensive coverage of the fine-scale (1 m) MBES mapping that provided the necessary information on habitat characteristics to allow use of spatially balanced survey designs. The GIMR (41.8669°S, 148.3124°E) is a 60-hectare no-take marine reserve (IUCN category II) established in 1991 (Tasmania Parks & Wildlife Service, 2013), of which 0.296 km<sup>2</sup> of reef habitat has been mapped using MBES. Butler's Point is a commercially and recreationally fished area located c. 8 km south of GIMR, encompassing 2.247 km<sup>2</sup> of MBES mapped reef habitat. Both GIMR and Butler's Point are characterized by steep algal-covered granite reefs that transition to diverse sessile invertebrate beds at depths below

40 m (Lucieer et al., 2007). These two surveys are characterised as 'local' scale studies. The southernmost site encompasses 118.182 km<sup>2</sup> of MBES-mapped area on the eastern coastline of the Tasman Peninsula in south-eastern Tasmania (43.0853°S, 147.7983°E). This site is subject to commercial and recreational fishing and is composed of a wide variety of geomorphology ranging from pavement to complex boulder fields with clearly defined patch reefs of irregular relief. This survey is characterised as a 'regional' scale study.

For BAS sampling we followed protocols set out in (Foster et al., 2020). The BAS approach was implemented using the *quasiSamp()* function in *MBHdesign* R-package (Foster, 2020) to distribute the start location of each

ROV across each study location. Inclusion probabilities were altered so that sampling was balanced across all depths at each location, but with greater sampling effort (80%) being allocated to areas of high seafloor ruggedness (representing rocky reefs) than to low ruggedness (representing sediment). The BAS approach enables a master sample to be generated to allow for the addition of contingency sites should some be missed (Foster et al., 2017). A master sample of 60 ROV transects was allocated to GIMR and Butler's Point, of which 26 were completed at GIMR and 24 were completed at Butler's Point (Fig. 1) between June and August 2017. At the Tasman Peninsula, all 110 ROV surveys (Fig. 1) were completed between March and April 2018.



**Figure 1.** Study locations of remotely operated vehicle surveys in (A) Governor Island Marine Reserve, (B) Butler's Point, and (C) the Tasman Peninsula off the east coast of Tasmania, Australia, selected based on a Balanced Adaptive Sampling approach.

## ROV setup

Fieldwork in GIMR and Butler's Point were undertaken using a Seabotix LBV 300-6 ROV and in the Tasman Peninsula using a Seaeye Falcon ROV. Both ROVs employed LED lights (700 Lumens for the Seabotix and 6400 Lumens for the Seaeye Falcon) adjusted to medium-high intensity for all sites to reduce backscattering in the footage (Lauermaun, 2014). The real-time position of the ROV was recorded using Garmin GPSMAP 78 Handheld GPS and Tritech Ultra Short Baseline Acoustic Positioning System (USBL) paired with SeaNet software (www.seanet.com). Two different calibrated stereo camera units (HERO5 Black GoPro and Sony Action Cameras both set to medium and frame rates of >30 pps) were used, both having a similar field of view of ~5 m total width at a 20° downward angle. Calibration of cameras was undertaken in Cal software (seagis.com.au) following protocols outlined in Monk et al. (2020).

## Data collection

ROV surveys in GIMR and Butler's Point followed a 150 m polygonal transect design composed of three 50 m-long strip transect segments at each study site, loosely forming the shape of a triangle (Fig. 1). The starting of each transect was begun as close to the target study site as practical in current and wave activity, with the heading of each transect segment chosen arbitrarily. This transect design was chosen for its suitability in sampling a relatively small area while maintaining a similar depth and habitat structure and minimizing the risk of transect overlap. At the Tasman Peninsula, ROV transects were

undertaken using 200 m-long strip transects following a straight-line, when possible, to provide efficient and effective cover across this large-scale cross-shelf location (Fig. 1). A consistent velocity of ~1 m s<sup>-1</sup> and altitude above the seafloor (~0.5–1 m) was used throughout all deployments, which gives a ~5 m-wide field of view. Sampling was conducted during daylight hours (c. 1 h after sunrise to 1 h before sunset) to prevent confounding effects from the nocturnal emergence of urchins from the refuge of crevices in rocky reefs (Jones & Andrew, 1990) that are known to influence detection rates (Ling et al., 2016).

## Data treatment

The density of urchins (number of *C. rodgersii* per m<sup>2</sup>) and presence of barren habitat within a 5 m-wide field of view in the video footage were recorded at 1 s intervals throughout each transect. Barren habitat was designated as rocky reef substrate absent of foliose algae in which urchins or a clear delineation between barren and healthy reef habitat indicative of urchin activity was present (Hill et al., 2003). A larger transect width was selected than in SCUBA diver-based Underwater Visual Census surveys (~2.5 m total width; Andaloro et al., 2013) because the video camera was able to record urchins reliably and accurately at this larger distance. Due to positional error with the USBL, estimates were aggregated using the sum of urchin density and barren habitat presence for each site. To account for this issue, the seabed terrain covariates (Table 1) were extracted in QGIS using circular buffers with a 50 m radius from the centre of each site in GIMR and Butler's Point and with a 100 m radius from

**Table 1.** Covariates used in GAM (generalized additive model) analyses for estimating urchin density and barren cover.

Covariate	Description	Source	Transformation
Urchin Density	Number of <i>Centrostephanus rodgersii</i> individuals observed m <sup>-2</sup>	ROV video data	Square root
Depth	Mean water depth (m) determined for the transect	MBES	Square root
Northness	The extent in radians to which habitat deviates from a north-facing position (calculated by $\sin[\text{aspect in radians}]$ ), with values closer to 1 indicating a more north-facing location and values closer to -1 indicating a more south-facing location (Wilson et al., 2007)	MBES	None
Eastness <sup>1</sup>	The extent in radians to which habitat deviates from an east-facing position (calculated by $\cos[\text{aspect in radians}]$ ), with values closer to 1 indicating a more east-facing location and values closer to -1 indicating a more west-facing location (Wilson et al., 2007)	MBES	None
Seafloor ruggedness (VRM)	The amount of deviation between adjacent 3 × 3 m MBES cells of a digital elevation model (Riley et al., 1999)	MBES	Square root
Slope <sup>1</sup>	Maximum change in elevation from an MBES cell to adjacent 3 × 3 m MBES cells, measured in degrees from the horizontal	MBES	Square root

<sup>1</sup>Non-significant covariates not included in final models.

the centre of each site in the Tasman Peninsula. A Spearman's correlation value of 0.8 was used to identify seabed terrain attributes that were correlated. Terrain attributes that have been previously found to influence urchin distribution (i.e. Perkins et al., 2020) were kept as explanatory variables in the analysis.

## Analytical approach

Generalized additive models (GAMs), a flexible, nonparametric generalization of generalized linear regression (Hastie & Tibshirani, 1986), were used to provide estimates of urchin density (individuals per m<sup>2</sup>) and barren cover (%) based on the scored ROV imagery and the seabed structure attributes. The GAMs are used as a model-based approach to calculate estimates that can accommodate various sampling/design regimes if sampling is reasonably representative of the covariate space, and there is sufficient data to estimate model parameters. The response variables slope, terrain ruggedness (VRM), depth (m), and urchin density were square root transformed to meet normality assumptions (Table 1). Data were partitioned into training (80%) and test (20%) data sets to evaluate model fit. Explained deviance ( $d^2$ ) with respect to degrees freedom (Engler et al. 2004), Akaike's information criterion (AIC) (Wood, 2017) and root mean square error (RMSE) were used as measures of model accuracy, with models displaying high explained deviance (>0.6; Castella et al., 2001) and low RMSE and AIC (relative to the models created at each region) were considered more accurate (Caruana & Niculescu-Mizil, 2004). The coefficient of variation (CV) was used as a measure of precision for these estimates (Monk et al., 2018; Peel et al., 2013) and represents the variance of the model predictions approximated using the delta method (Oehlert, 1992; Wood, 2006). A CV value <0.4 was considered to be adequately precise (Maravelias et al., 2012). An individual tweedie GAM using the restricted maximum likelihood estimator in the *mgcv* package in R software (Wood, 2019) was constructed for each location to model the patchy distribution of urchin density and urchin barren cover (Peel et al., 2013) and to take into account the relatively small sample sizes at GIMR and Butler's Point (McNeish, 2017). The argument `select = TRUE` was used to add an extra penalty to the GAM to allow coefficients to tend toward zero (Wood, 2017). The GAMs were fitted with a cubic spline smooth and three degrees of freedom after varying these parameters (Wood, 2006). The *gam.check()* function in the *mgcv* package was used to evaluate whether the basis dimension for the smooths were adequate (Wood, 2017), which found that a  $k = 4$  basis dimension was necessary to fit all covariates in the urchin density models except for depth at GIMR. Each model

was constructed manually in a backward stepwise manner. Terms were removed from the model such that each step resulted in the smallest significant reduction in residual deviance when compared to the previous model using an approximate chi-square test (Wood, 2006). Only significant terms were retained in each GAM for final model estimates. Total barren cover was calculated by taking the sum of GAM predictions across a fine-scale grid of equally spaced points in the region. Spatial autocorrelation between sites was assessed using Moran's I correlation coefficient in the *ape* package in R. Model residuals were also assessed using the *acf()* function in the *stats* package. No residual spatial autocorrelation was detected in any of the models, and thus no correction was applied. The *predict()* function in the *car* package was used to obtain model predictions and the standard error for each of the regions.

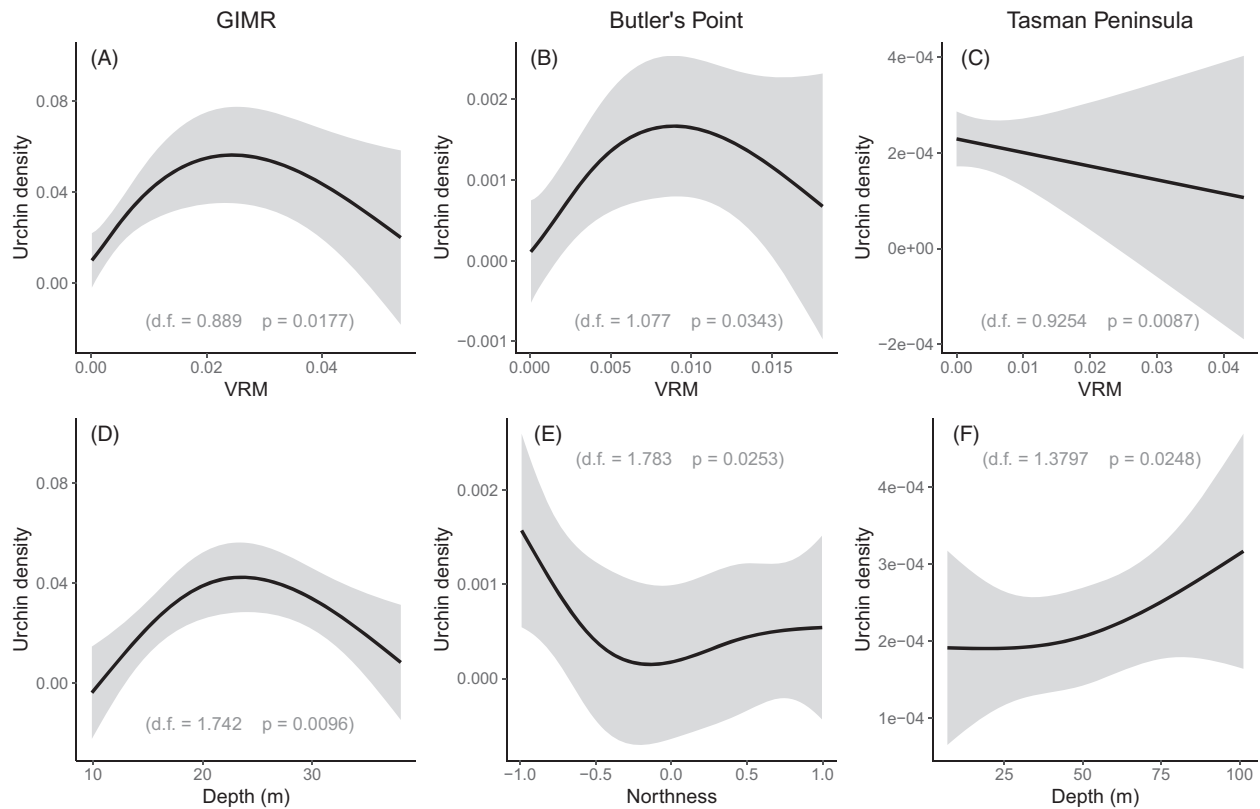
## Results

### General patterns in the prevalence of urchins and barren cover

The ROV footage revealed that the prevalence of urchins and barren habitat was highest at GIMR, wherein ~53% of sites had urchins and barrens present. The prevalence of urchins at Butler's Point and the Tasman Peninsula was substantially less, occurring at 25 and 8% of sites, respectively. The ROV footage at Butler's Point and the Tasman Peninsula revealed barren cover to be at 21 and 27% of sites, respectively.

### Seafloor structure relationships with urchin density and barren cover

The GAMs revealed some regional differences in the relationships between depth and seafloor structure attributes for predicting urchin density and barren cover (Figs. 2 and 3). Seabed ruggedness was significant for predicting urchin density and barren cover across all models except for barren cover at the Tasman Peninsula, wherein it showed a bell-shaped relationship at GIMR and Butler's Point (peaking at a ruggedness of ~0.025 and 0.008, respectively; Figs. 2A, B and 3A, B) and decreasing urchin density with ruggedness at the Tasman Peninsula (Fig. 2C). Depth also displayed a bell-shaped relationship with density at GIMR (peaking at ~22 m; Fig. 2D) but had a positive relationship on urchin density (Fig. 2F) and a slightly negative relationship with barren cover at the Tasman Peninsula (Fig. 3C). No significant relationships with seabed slope or eastness were revealed in any of the models. Northness exhibited a slightly negative bell-shape curve effect on urchin density



**Figure 2.** Variation in urchin density (individuals  $m^{-2}$ ) predicted from the GAMs for each location (left column: Governor Island Marine Reserve (A, D), middle column: Butler's Point (B, E), right column: Tasman Peninsula (C, F)). Solid lines are mean fitted values from cubic regression splines while holding other variables within the GAMs constant; and grey shading represents 95% CI. Vertical bars at the bottom depict the distribution of data points across the predictor space. Significance levels ( $P$ -value) and degrees of freedom (d.f.) are displayed for each of the covariates. Note the  $x$ - and  $y$ -axis values are different between covariates and regions. GAMs, generalized additive models.

at Butler's Point (Fig. 2E), suggesting density was more likely to be greater on more exposed south-facing reefs. The GAMs revealed that predicted urchin density had a significant, positive relationship with barren cover across all regions (Fig. 3D–F).

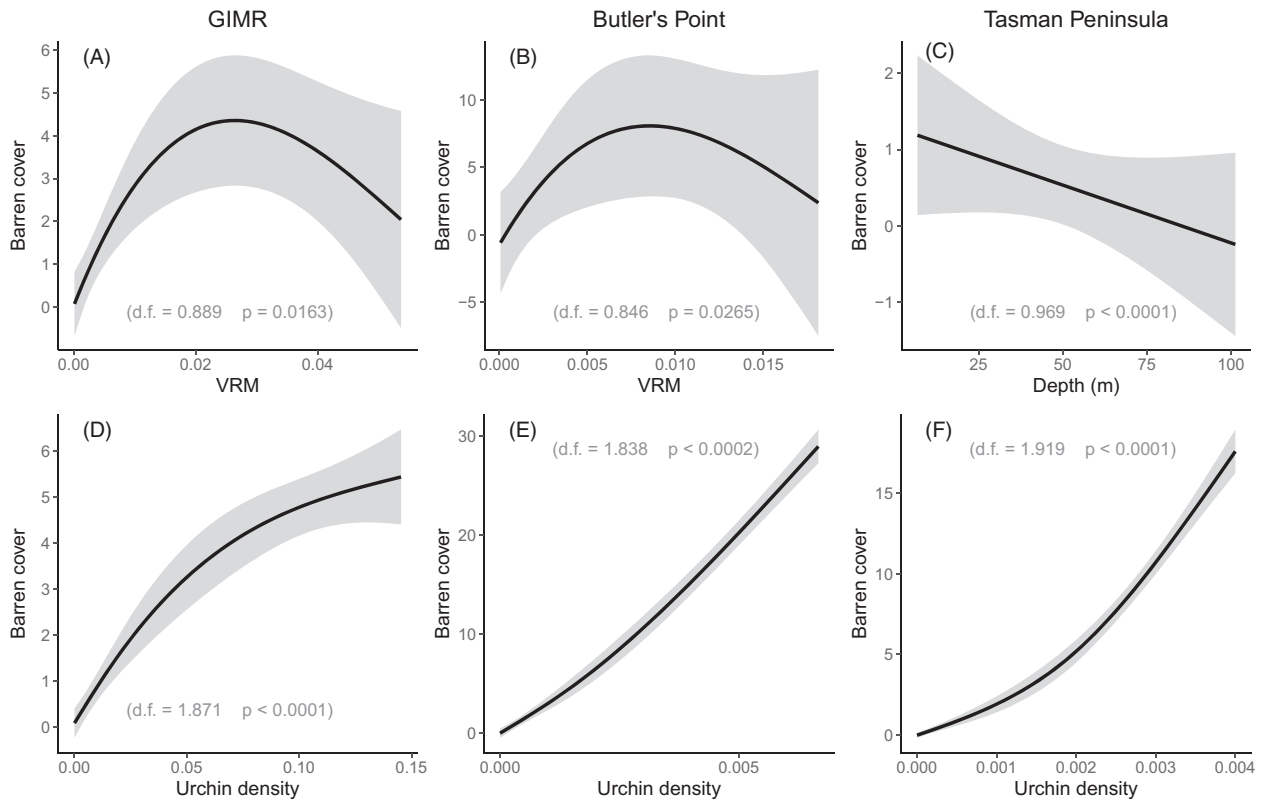
### Model estimates and spatial predictions

The accuracy of model estimates was moderate to high based on explained deviance (ranging from *c.* 0.49 to 0.996) and had reasonable values for AIC and RMSE (Table 2). There was a moderate level of precision for the estimates across all locations (Table 2). Model estimates across GIMR, Butler's Point, and the Tasman Peninsula revealed urchin densities of 0.1666, 0.00073, and 0.000065 individuals per  $m^2$  and  $\sim$ 2.10, 3.33, and 0.018% barren cover, respectively. This equates to total urchin abundances and barren habitat cover of 49 368 urchins and 0.006  $km^2$  of barren habitat at GIMR, 1640 urchins and 0.075  $km^2$  barren habitat at Butler's Point, and 7682

urchins and 0.022  $km^2$  of barren habitat at the Tasman Peninsula. Similar patterns were revealed in model predictions, reflective of the response curves from the GAMs (Figs. 2 and 3). Spatial predictions of urchin densities and barren cover between locations showed similar patterns, with higher numbers of individuals being predicted at higher ruggedness reef (see zoom examples in Fig. 4) at each location. Variation in the uncertainty in the spatial predictions was also observed, exhibiting the highest uncertainty on the shallow reefs features for both urchin density and barren cover (Fig. 4).

### Discussion

Preventing the phase-shift from healthy kelp-dominated rocky reef ecosystems to large areas of barren habitat because of the redistribution of marine ecosystem engineers (such as long-spined urchin) is a management priority. Underpinning this priority is the need for adequate monitoring with sufficient accuracy and precision to



**Figure 3.** Variation in urchin barrens cover predicted from the GAMs for each location (left column: Governor Island Marine Reserve (A, D), middle column: Butler’s Point (B, E), right column: Tasman Peninsula (C, F)). Solid lines are mean fitted values from cubic regression splines while holding other variables within the GAMs constant; and grey shading represents 95% CI. Vertical bars at the bottom depict the distribution of data points across the predictor space. Significance levels (*P*-value) and degrees of freedom (d.f.) are displayed for each of the covariates. Note the *x*- and *y*-axis values are different between covariates and regions. GAMs, generalized additive models.

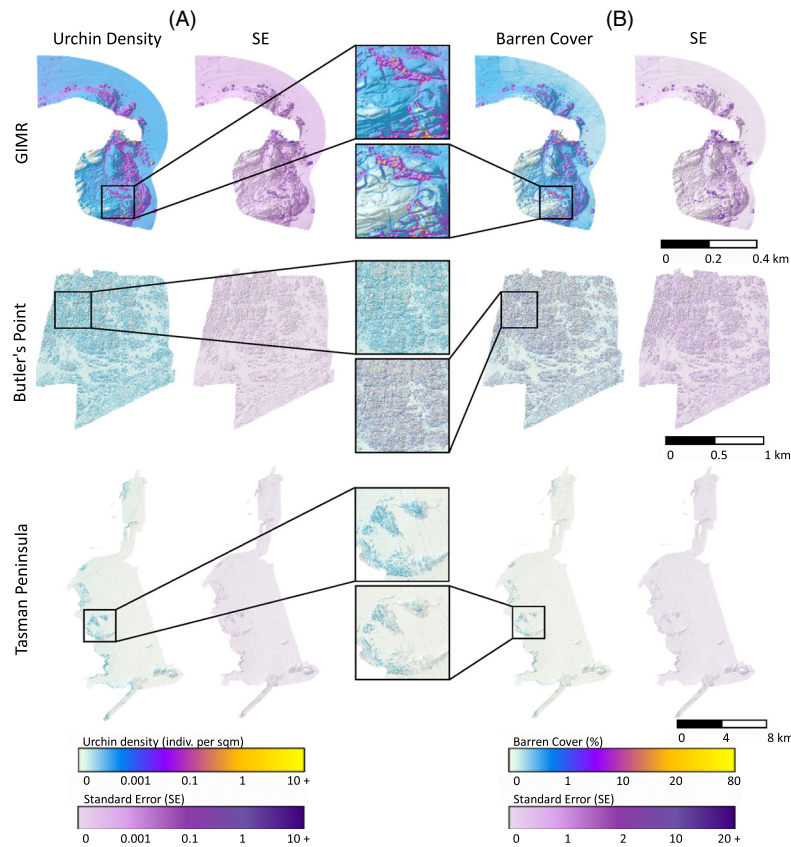
**Table 2.** Model estimates for urchin density (individuals m<sup>-2</sup>), barren cover (%), and the accuracy (explained deviance, AIC, and RMSE of training and testing datasets) and the precision (coefficient of variation) of the models.

		GIMR	Butler’s Point	Tasman Peninsula	
Urchin density	Estimate (individuals per m <sup>2</sup> )	0.16660	0.000731	0.000065	
	Accuracy	Explained deviance	0.493	0.694	0.396
		AIC	-13.884	-35.670	16.735
		RMSE of training dataset	4.793	8.428	9.983
		RMSE of test dataset	4.233	7.223	10.391
		Precision	Coefficient of variation	0.3973	0.4139
Urchin barren	Estimate (% cover)	2.0965	3.3322	0.0184	
	Accuracy	Explained deviance	0.866	0.993	0.996
		AIC	35.227	33.914	91.472
		RMSE of training dataset	2.414	0.268	5.159
		RMSE of test dataset	2.143	1.823	5.034
		Precision	Coefficient of variation	0.0888	0.1951

reliably estimate the regional distribution of the abundance of urchins and extent of associated barren habitat formation. Here, we undertook ROV surveys based on a probabilistic sampling design, taking advantage of existing comprehensive MBES mapping, an essential prerequisite

to underpin such sophisticated designs and model-based analyses. We found that precise estimates of urchin density and barren cover were possible when ROV observations are combined with MBES-derived seafloor structure attributes, although accuracies could be further improved.





**Figure 4.** Predicted distribution and standard error of *Centrostephanus rodgersii* density (A) and barren cover (B) at Governor Island Marine Reserve (top), Butler's Point (middle) and the Tasman Peninsula (bottom) estimated using ROV video data. Zoom boxes show examples of spatial areas of ~200, 500 and 3000 m, respectively.

Using this representative survey design, we generated local (GIMP and Butler Point) and regional (Tasman Peninsula) urchin density and barren cover estimates as a baseline for future monitoring and as an indication of current status to inform potential adaptive management. Overall, our urchin densities and barren cover were markedly less than encountered in the historical native range of *C. rodgersii* within New South Wales, where mean densities of between ~2.5 and 8 urchins  $m^{-2}$  have been reported (Andrew & O'Neill, 2000). This is not surprising given that the expansion of *C. rodgersii* into Tasmania is relatively recent, and initial monitoring efforts have reported urchin densities substantially lower than found in their native range of New South Wales (e.g. Perkins et al., 2020) as this species gradually increases its range and abundance within Tasmanian waters. Likewise, our results demonstrated a significant decline in abundance and barren cover from the northern locations (GIMR and Butlers Point) to the most southern region of the Tasman Peninsula, following the spatial pattern observed in previous studies (e.g. Johnson et al., 2005). Interestingly, our

results were generally in the same range as predicted by Perkins et al. (2020) for GIMR and Butler's Point in 2017 sampled using random AUV-derived transects, with an estimated 2.1% cover (this study) versus 1.32% for GIMP, and 3.3% versus 3.96% for Butler Point. However, they differ markedly from towed video results from a survey in 2016/17 (Ling & Keane, 2018; their figure 6), who estimated barren cover in the Bicheno region (covering the GIMP to Butlers Point coast) at *c.* 13.5% between 4 and 40 m depth, and the mean cover in two locations on the Tasman Peninsula to be around 3.6% at similar depths. As our estimates are considerably less than these, we suggest these discrepancies could be due, in part, to three key factors: (1) differences in sampling design (haphazard vs. spatially balanced selection), (2) the method of extrapolation (simple vs. model-based), and (3) variation in detectability aligned with differences in survey methods used between studies. These discrepancies were particularly evident at the Tasman Peninsula, for which towed video surveys more intensively surveyed the 15–25 m depth range where urchins are currently more prevalent



in, whereas our study provided comprehensive estimates throughout the depth range in this region, including deep-water habitat where urchins are not presently found. While this likely led to lower predicted barren cover for the whole region, estimates within the same depth range were more comparable between studies (0.85% barren cover between 5 and 40 m), though they still differed by a factor of 4. An additional confounding factor is that we sampled within the GIMP, a no-take marine park with evidence of resilience to urchin invasion due to a larger number of predators, which was shown to have notably less barrens than adjacent nearby fish locations, which averaged around 5.5% barren cover at the time when assessed using the AUV-based survey design (Perkins et al., 2020). Clearly, sampling design matters, and future studies need to weigh up the merits of approaches that provide robust regional estimates versus approaches that can adequately assess the trajectory of change at a local scale but may provide less reliable estimates of overall density and barren cover.

Randomising sample locations is pivotally important when wanting to make an inference about a focal population/region (e.g. Smith et al., 2017; Tillé & Wilhelm, 2017). We applied a flexible probabilistic sampling design that captures the properties of randomisation but allows for specific strata to be more intensively sampled (in our case reef habitat). This generally results in estimates that are less variable, and more reasonable, than if they were calculated using simple random sampling (Stevens & Olsen, 2004). By contrast, previous urchin studies in Tasmania have relied on haphazardly or strategically selected sample sites as part of their sampling strategy. For example, Ling and Keane (2018) replicated surveys in sites initially chosen by Johnson et al. (2005) that may have repeat sampled sites at locations known for high densities of urchins and barren habitat cover, a strategy which may be sufficient to track change through time at those specific transects, but will not be generalizable to the broader region, and will likely overestimate urchin densities and habitat barren (Smith et al., 2017). While this study is not intended as a method comparison paper, it does demonstrate the usefulness of probabilistic sampling designs for providing robust estimates of density and cover where such estimates are important. Similar approaches have also been successfully applied for the quantification of the spatial extent continental shelf habitats (Lawrence et al., 2015) and the sessile biota they support (Monk et al., 2016), quantification of the size of vulnerable marine ecosystems on seamounts (Williams et al., 2020), and the inventory and description of the distribution of demersal fish communities (Hill et al., 2018). Perhaps one of the biggest advantages of the probabilistic sampling designs is that it allow for better

representativeness and spatial coverage to be maintained should sites need to be added or to be dropped due to weather conditions or technical failures (Foster et al., 2018). Additional research contrasting the performance of this approach with conventional approaches would provide better insight into the advantages of BAS.

We also demonstrate the importance of considering variations in seabed structure attributes when generating region-wide estimates of urchin density and barren cover. Our study found that these estimates varied across depth and orientation of the seabed (i.e. northness). The importance of seabed structure attributes in mapping the distributions of mobile and sessile marine organisms are well established (e.g. Monk et al., 2018; Moore et al., 2010; Perkins et al., 2020; Wilson et al., 2007), but have rarely been used in fisheries studies (Rubec et al., 2016). Previous quantitative estimates of urchin density and barren cover in Tasmania have not had the advantage of utilising extensive MBES datasets to incorporate the influence of subtle and important variations in seabed structure into estimates, and instead have relied on available mapped reef extent and bathymetry to extrapolate urchin densities within transects to broader regions (Ling & Keane, 2018). This simpler approach allows urchins and barren habitat within an area to be tracked throughout time but assumes that reefs are uniform and seabed characteristics surveyed on transect are representative of the greater region, which can lead to under- or over-inflated estimates.

What constitutes an acceptable level of accuracy and precision in the models will vary depending on context and the specific study objectives. We considered any model with an accuracy (explained deviance) of 0.4–0.6 as moderately sufficient (Maravelias et al., 2012) and models with an explained deviance of >0.6 to have relatively high explanatory power (Castella et al., 2001), whereas we considered CV values <0.3 as adequately precise (Peel et al., 2013). Previous studies have reported similar ranges in accuracy for other marine biota, including fish (0.14–0.55; Galaiduk et al., 2017), sessile invertebrates (0.04–0.68; Monk et al., 2018) and physical habitat structures (0.30–0.88; Torriente et al., 2019). That being said the accuracy and precision of most models in our study could be improved and would benefit from incorporating additional biotic variables, such as the extent of macroalgae canopy cover and occurrence of lobster, which have been linked to urchin density and barren formation (Johnson et al., 2005; Ling, 2008; Ling et al., 2009). Additional oceanographic variables such as small-scale temperature-related changes could further improve model accuracies and precision as they have been linked to urchin recruitment (Banks et al., 2010; Banks et al., 2007; Colwell & Rangel, 2009; Harley et al., 2006; Ling et al., 2008). At present these datasets are not at sufficient

spatial resolution to substantially improve model accuracies. It should also be noted that scales in which ecological processes operate at are important when generating covariates for such models (Lecours et al., 2015). For example, previous studies have advocated for multiscale analyses (Lecours et al., 2017; Monk et al., 2011; Wilson et al., 2007) so that important factors operating at different scales are not obscured. We did not apply a multiscale approach in our study due to errors in the positional data from USBL system precluding us from coupling the imagery and mapping data at fine scales. Hence, seabed terrain attributes were instead obtained by averaging along each transect, which has potentially further lower model accuracies by smoothing fine-scale relationships influencing urchin distribution (Lecours et al., 2017).

Another key issue when deriving robust estimates of urchin density using stereo footage from ROVs is optimising detection rates. Although variations of detection rates were not explicitly accounted for in this study, they could in part explain the differences in density, but not barren cover estimates with previous studies. The presence of kelp canopy in shallow waters (<20 m) or within high-profile cracks and crevices in the rocky substrata are likely to lead to variations in the detection rates of urchins, with detectability also changing depending on the survey equipment used (Monk, 2014). For example, SCUBA-divers detected ~40% more urchins in dense canopy beds when compared to AUVs (Ling et al., 2016). Variations in detection rates between sampling gear has been documented in other marine species, including fish, crustaceans, and epibenthic communities (Andaloro et al., 2013; Ayma et al., 2016; Chimienti et al., 2018; Schramm et al., 2020). The range measurement associated with stereo-video annotation data from a ROV could be used to improve population estimates through distance sampling. For example, the *dsm* r-package that implements a GAM-based approach to calculate spatially explicit estimates of organism abundance and importantly associated uncertainty could be used (Miller et al., 2013). Where data allows, the application of such analysis approaches are worthy of further investigation to better refine accurate assessment of urchin density while explicitly accounting for variation in detection rates.

The management of continued increases in urchin densities and barren formations in Tasmania is an ongoing issue. Systematic culling in north-eastern Tasmania has been shown to be effective at controlling urchins within small spatial areas, but can be expensive and logistically difficult at larger scales (Tracey et al. 2014). The use of commercial divers to cull urchins during normal fishing activity offers a more cost-effective solution, however Sanderson et al. (2016) suggests this method is primarily driven by fishing yield and has limited effectiveness on

population control when compared to establishing a dedicated fishery for urchins. Our model estimates and associated spatial predictions could be used to prioritise management responses aimed at reducing urchin numbers while simultaneously maximising fishery quotas for the burgeoning urchin fishery (Cresswell et al., 2019). For example, management could identify reefs that have commercially viable densities of urchins yet predicted low levels of barren cover to maximise fisheries quota, while further reducing the likelihood of barren formation. This management strategy could also have benefits for the fishery, as Blount et al. (2017) suggests improved urchin quality (gonad quality) can be achieved by controlling the formation of high urchin densities. Similar management approaches have been used elsewhere to improve the gonad yield of the red sea urchin (*Strongylocentrotus franciscanus*) as well as increase kelp cover and distribution (Andrew et al., 2002).

We demonstrated the utility of ROVs equipped with stereo video, based on probabilistic sampling design, to quantitatively estimate urchin density and barren cover at a regional scale. Although precise estimates (i.e. repeatable) of urchin density and barren cover were possible when ROV observations are combined with MBES-derived seafloor structure attributes, we found that model accuracies could be further enhanced, perhaps with addition of other covariates. Seasonal and temporal replication is also recommended to determine how urchin densities and barren habitat, once established, persist through seasons and annually. We advocate that the information generated here can be used to spatially target efforts to reduce urchin numbers via mechanisms such as spatially targeted quotas for burgeoning urchin fishery. Finally, we advocate the use of probabilistic sampling designs such as those presented here when developing and implementing monitoring programs due to their flexible nature and ability to maintain good spatial coverage should additional samples need to be added or to be dropped due to changes in weather conditions or equipment failures. Importantly, where gaining a robust understanding of the region-wide extent of major habitat perturbations such as the formation of urchin barrens is important to management, non-selective spatially balanced designs offer a solution to generality of inferences from monitoring programs. However, for such designs to be effective, they first must be underpinned by a similar investment in prior mapping effort, on which to base appropriate spatial sampling.

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