Chapter II - Sampling design comparison

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# 1 Introduction

Monitoring the marine environment is becoming increasingly important as we try to understand and manage the effects of local and global stressors on species and habitats (Borja and Elliott 2021). Reducing the effects of local stressors on the environment can slow down the effects of climate change in the medium-term on marine communities (Brown et al. 2013; Gissi et al. 2021). For example, effective fisheries management could reduce the climate risk of many species of marine fish (Cheung et al. 2018), and Marine Protected Areas (MPAs) and No-Take Zones (NTZs) could provide resilience to climate-driven shifts of deleterious species (Ling and Johnson 2012). As a result, correctly identifying the benefits of MPAs (or lack thereof) is critical to advise whether objectives are being met and ecosystems are adequately protected from local stressors (Miller and Russ 2014).

Countless methods are used to monitor MPAs, depending on the main aim of the MPA.Correctly attributing biological trends to the conservation intervention under study has been the object of discussion in the literature for decades. Miteva, Pattanayak, and Ferraro (2012) reviews conservation instruments (including protected areas) and urges “better theory, better methods, better data” are needed to evaluate conservation interventions, as monitoring often makes conclusions doubtful. Ferraro and Pattanayak (2006) adds to this that programme evaluation quantity cannot overcome quality. The presence of controls, the number of temporal and spatial replicates etc., despite being fundamental ecological considerations (Underwood 1991), appear in very few studies of MPA effectiveness (Osenberg et al. 2006). As a result, poor monitoring design make it difficult to answer many (or any) questions relating to the effect and performance of MPAs on biological metrics (Hayes et al. 2019).

In Australia, Baited Remote Underwater Video (BRUVs) are widely used to monitor the effects of marine management like Sanctuary Zones (NTZ denomination specifically in Australia) on abundance and size of communities and target species (Harasti et al. 2018; Kelaher et al. 2014; Malcolm et al. 2018). BRUVs present advantages compared to other common methods, including the ability to sample areas inaccessible by divers and more sensitive detection of changes in fish communities compared to other census methods (Schramm et al. 2020). The ability to use footage for a variety of research questions, including fish community, (Watson et al. 2007) size, (Malcolm et al. 2018; Watson et al. 2009) and habitat (Scott et al. 2022) make BRUVs a popular choice for managers. Like any spatial sampling, the choice of where to deploy sampling units like BRUVs depends on the research question of interest (Foster et al. 2018). Nevertheless, very few BRUV research articles report their spatial sampling design, or why sites were chosen for deployment at all (Whitmarsh, Fairweather, and Huveneers 2017).

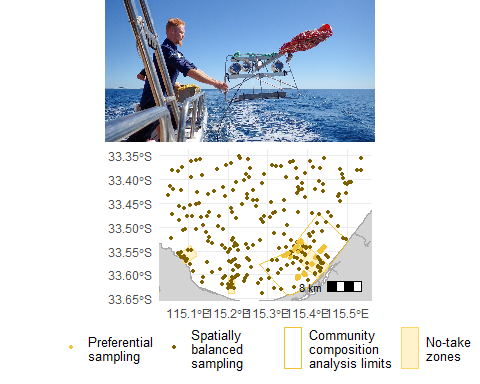
Spatial sampling design choice has been extensively discussed in other methodologies, identifying large differences in estimates and outcomes depending on the sampling method used. For stock assessments of fished species Cheng et al. (2024) and underwater visual census of reef fish Smith et al. (2011), discussions as to how to distribute samples across a study area are common. Certain sampling designs like preferential designs are known to produce strong biases in abundance estimates and other trends found in ecology Conn, Thorson, and Johnson (2017). However, while many discussions bring some form of stratification into optimal designs, the outcomes could depend on the chosen metric, and with environmental shifts over time like temperature (Zhao et al. 2018). One study exploring BRUV sampling design specifically looked at a large spatial scale (>1000km) for stock assessment purposes, not specifically looking at MPAs (Switzer et al. 2023). With the variety of spatial sampling design performances in other methodologies, and the paucity of explorations of BRUV spatial sampling designs, questions remain as to the validity of conclusions of BRUV studies, especially in the context of MPAs.

In Wadandi Country (south-west Western Australia), recreational fishing pressure has lead to the depletion of emblematic fish species like Djubitj/West Australia Dhufish (*Glaucosoma hebraicum*) and Yijarup/Pink Snapper (*Chrysophrys auratus*) (Ryan, Smallwood, and Lai 2022; Gaynor, Kendrick, and Westera 2008). Despite the implementation of an NTZ network between Waatern/Geographe Bay and Taalinup/Augusta, the effectiveness of these measures may not match the response of target species found in other NTZs (Alós and Arlinghaus 2013; Taylor and McIlwain 2010; Harasti et al. 2018, 2019). The area is monitored by BRUVs, inside and outside NTZs, with two different spatial sampling designs aiming to assess MPA performance and effectiveness. This context represents a perfect case study for the comparison of these two sampling designs to assess fish communities in NTZs.

This preliminary study aims to compare two BRUV spatial sampling designs at a local (<30km) scale, to understand how the detection of single and multiple species differ between a preferential and spatially balanced sampling design. Real BRUV data is used to (1.) compare community composition, and (2.) simulate sampling designs on a modelled spatial distribution model of a single species. Yijarup/Pink Snapper (*Chrysophrys auratus*) was chosen as a study species for (2.) as it is both a popular recreational fishing species (Ryan, Smallwood, and Lai 2022; Gaynor, Kendrick, and Westera 2008) as well as a species of interest to Wadandi Traditional Owners, who have intimate traditional knowledge of this species’ ecology and life history, cited in Wadandi cultural songlines (Davies et al. 2022).

# 2 Methods

## 2.1 Study site and BRUV deployment



(#fig:real\_BRUV\_location)(top) Wadandi Ranger Joe deploying a BRUV in Waatern/Geographe Bay. (bottom) Location of preferential and spatially balanced samples in Waatern/Geographe Bay, with the NTZs and boundary of the community composition analysis.

Stereo Baited Remote Underwater Video (BRUV) footage was collected February-April 2024 in Waatern/Geographe Bay, in Wadandi Country, southwestern Australia (Figure @ref(fig:real\_BRUV\_location)) as part of the state and federal Marine Park monitoring of Western Australia. This area represents a temperate ecosystem dominated by seagrass beds, sand and limestone reef (Galaiduk et al. 2018). The area contains NTZs gazetted in 2018, covering 51.77km2 of the bay. Deployments were conducted following standardised protocols for bait quantity and deployment times, used across Australia, which include calibration of the stereo systems before and after campaigns (Langlois et al. 2020). Wadandi Rangers and Cultural Custodian Elders were present and participated in the entirety of the BRUV deployments, informing the location of some of the preferential deployments. (Figure @ref(fig:real\_BRUV\_location))

BRUVs were deployed according to two sampling designs: preferential sampling and spatially balanced sampling:

* Spatially balanced sampling aims to distribute sampling units randomly within strata of an environmental variable. This sampling design accounts for spatial heterogeneity and spatial autocorrelation (Kermorvant et al. 2019). More details on this sampling design is available further in section @ref(sec:SD\_sim).
* Preferential sampling locations were sought out for the seabed complexity of the area, aiming to deploy BRUVs near highly complex sections, where many species tend to aggregate (Fernández et al. 2008). Deployments were either clustered around or aligned along seabed features (Figure @ref(fig:real\_BRUV\_location).

## 2.2 Data analysis

### 2.2.1 Annotation

BRUV footage was annotated by research assistants trained in temperate Australian bony and cartilaginous fish species identification, to obtain MaxN (the greatest number of individuals of a species observed at one time in a deployment), lengths of individuals (possible through the stereo-video system), as well as habitat data (Langlois et al. 2020). Quality assurance was conducted through species identification cross-checking with other annotators, as well as through CheckEM (<https://github.com/GlobalArchiveManual/CheckEM>), a purpose-built application to compare identified species with their life histories and geographical ranges.

### 2.2.2 Community composition analysis

The East Geographe NTZ (composed of National Park Zones and Sanctuary Zones) was used as a case study for the community composition analysis, as this was the only area of the campaign where the inside and outside of the zones were sampled by both sampling designs. Spatially Balanced sample points were subsampled (Figure @ref(fig:real\_BRUV\_location)) to match the Preferential sampling in and around the East Geographe NTZ for this part of the analysis. To investigate the difference in species composition between sampling designs, we conducted Principal Coordinate Analysis (PCoA; Bray-Curtis distances; vegan package, (Oksanen et al. 2013) and manyglm (negative binomial, adjusted p-values, mvabund package, (Wang et al. 2012)), a generalised linear model framework for multivariate abundance data. Sampling design (preferential or spatially balanced), status (NTZ or fished), and depth were included in the model, to account for their potential effects on community composition (Scott et al. 2022; Watson et al. 2007). Habitat was purposefully not included in the model as we considered it to be a sampling design choice: Bathymetry plays a major role in sampling design choice (see 2.2.3.3 below), and is also a strong contributor to habitat type and species distribution (Cameron et al. 2014). Assumptions of log-linearity, homoscedasticity and normality were confirmed prior to analysis using the plot.manyglm and qqnorm functions.

### 2.2.3 Simulation of single species detection

To compare the sampling designs’ ability to detect the effect of a NTZ on a single species, we conducted simulations of sampling designs on a modelled abundance distribution of Yijarup/Pink Snapper (*C. auratus*) around a simulated NTZ.

#### 2.2.3.1 Abundance distribution model

To build the abundance distribution model, several environmental variables were obtained from observed and external data:

To account for seabed morphology, 250m and 5m resolution bathymetry datasets from Geoscience Australia were tested as co-variates in the spatial distribution model. The terrain function of the raster package (Hijmans, Van Etten, and Cheng 2015) also allowed bathymetry derivatives (aspect and roughness for both datasets and detrended bathymetry for the 250m dataset) to be tested in the model. Both resolutions were included to test the potential different biological scales that may affect *C. auratus* abundance. Habitat information from BRUV data was used to fit the spatial distribution model. Reef, seagrass and sand were tested, as *C. auratus* abundance has been found to be higher near structured habitat like reef and kelp (Parsons et al. 2016; Terres et al. 2015).

The distribution of *C. auratus* was then modelled using BRUV abundance data (all samples were included, preferential and spatially balanced). Only mature individuals (>375mm, (Wakefield et al. 2015) were used in models, as these larger fish are targeted by recreational anglers, and tend to benefit from NTZ protection (Alós and Arlinghaus 2013; Taylor and McIlwain 2010). Obtaining the lengths of fish in a video is dependent on whether the fish are visible and positioned correctly in the video frame, which may not be possible, and we acknowledge that the number of measured mature individuals may not reflect a. the MaxN in that particular video or b. their abundance in the area. Rather, this spatial distribution model should be viewed as a hypothetical species rather than an accurate or true distribution of *C. auratus* in Waatern/Geographe Bay. Generalised Additive Models (using the FSSgam package in R, Fisher (2024)) were fitted to BRUV abundance data, using the habitat and bathymetry layers described above as co-variates. The relative importance of variables was assessed using the generate.model.set function to select the optimal model by ranking viable model sets according to Akaike Information Criterion (AIC) values (Fisher et al. 2018).

From this model, the distribution of *C. auratus* was predicted over the entirety of Waatern/Geographe Bay using the optimal model, and the predict.gam function of the mgcv package in R (Wood 2001). Predictions were made within the range of observed environmental variables, to avoid extrapolating abundance values. While bathymetry variables were already rasterised, no raster of habitat was deemed accurate or consistent enough to use for this distribution prediction. We therefore fitted Generalised Additive Models (using the FSSgam package in R, Fisher (2024)) of seagrass, sand and reef habitats from BRUV data to the bathymetry and bathymetry derivatives described above to create layers of predicted habitat to use in the *C. auratus* distribution prediction.

#### 2.2.3.2 Simulated NTZ

To test the ability of each sampling design to detect a true increase in *C. auratus* abundance in NTZs, a simulated NTZ was designed in Waatern/Geographe Bay (Fig. @ref(fig:example\_sampling\_design) ). The location and size of the NTZ was chosen to be representative and realistic compared to existing NTZs in the area. The chosen area encompasses a reef feature along the coastline, which is protected by the East Geographe NTZs further southwest of the bay.

The mean and standard errors of the predicted *C. auratus* abundance were increased uniformly by 80% within the NTZ to simulate a moderate ‘MPA effect’ on this recreational target species, as had been found for *C. auratus* by Harasti et al. (2018). We did not select another location to act as a ‘control’ site to compare to our ‘impact’ site (NTZ) as the nature of the iterated simulation (see below) acted as a replication of the ‘NTZ’ treatment.

#### 2.2.3.3 Sampling design simulation

The sampling design simulations were conducted in an area covering 223.5 km2, including 86.1 km2 of NTZ and ~68.7 km2 on either side of the simulated NTZ. This aims to mimic the subsample of BRUVs analysed for community composition. Areas shallower than 7m were excluded, as these are rarely targeted in real BRUV campaigns. Two sampling designs were simulated to emulate real BRUV sampling conducted in Waatern/Geographe Bay:

##### 2.2.3.3.1 a. Spatially Balanced Sampling Design

The spatially balanced sampling design was created by categorising the 250m detrended bathymetry raster layer into 3 strata (based on quantiles values, (@ref(tab:n\_sample\_table)), then randomly distributing a set number of sample points into each stratum. The number of sample points were decided to approach the number of samples collected in 2024 in a comparable area (@ref(tab:n\_sample\_table)). Sample points were placed using the Generalised Random Tessellation Stratified (GRTS) sampling function from the spsurvey package in R (Dumelle et al. 2023). The minimum distance between points was set to 500m to reduce spatial autocorrelation. Detrended bathymetry was considered to be most appropriate to use for the stratification as it encompassed the study site’s seabed features better than bathymetry, aspect or roughness. The GRTS sampling function was iterated 1000 times to obtain a set of simulated spatially balanced sampling designs.

##### 2.2.3.3.2 b. Preferential simulation

Samples were randomly distributed on the highest quantile of detrended bathymetry (Strata 3, Table @ref(n\_samples\_table)), to mimic the preferential approach of real sampling. The number of samples per strata were decided based on the number of samples from a comparable area in the BRUV campaign of this paper (Table @ref(n\_samples\_table)). The number of samples were then distributed using Generalised Random Tessellation Stratified (GRTS) sampling function from the spsurvey package in R (Dumelle et al. 2023). This method was iterated 1000 times to obtain a set of simulated preferential sampling designs.

#### 2.2.3.4 Analysis of simulation

Values of abundance were obtained from the latitude and longitude of the simulated points of each of the 2000 simulaions. To account for the biological variability of *C. auratus*, and the uncertainty associated with detecting *x* individuals consistently where *x* individuals are predicted to be, each sampling point could observe one abundance value within the Normal distribution of abundance of that raster pixel (rnorm function in base R). For each simulation, ratios of the mean abundance of *C. auratus* inside and outside the NTZ before the NTZ increase, and after the NTZ increase were calculated to assess the ability of each sampling design to detect changes in the NTZ (Equation 1):

# 3 Results

(#tab:n\_samples\_table)BRUV Sampling Results

Inside No-Take Area

Outside No-Take Area

Total

a. Real BRUVs collected

Preferential Sampling

18

29

47

Spatially Balanced Sampling (full data)

45

191

236

Spatially Balanced Sampling (subset data)

27

25

52

b. Simulated BRUVs (spatially balanced sampling design)

Strata 1 (0-50% quantile)

10

10

20

Strata 2 (50-80% quantile)

10

10

20

Strata 3 (80-100% quantile)

5

5

10

c. Simulated BRUVs (preferential sampling design)

Strata 3 (80-100% quantile)

25

25

50

A total of 283 BRUV samples were collected, including 47 preferential and 236 spatially balanced samples (Table @ref(tab:n\_samples\_table)). In all samples, 184 species were detected, including 56 species identified to the Genus or Family level and 128 species identified to the species level. Certain species with very similar appearance (=difficult to identify/differentiate with certainty) were grouped together to form a complex, and are considered as one species.

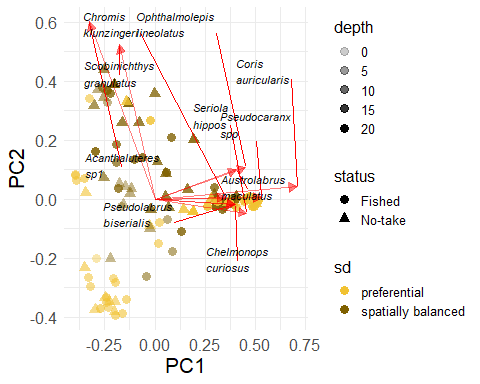
Within the yellow box specified in Fig. @ref(fig:example\_sampling\_design), 18 preferential samples were collected inside the NTZ, and 29 were collected outside. In the same area, 27 spatially balanced samples were collected inside the NTZ, and 25 were collected outside the NTZ.

## 3.1 Community composition analysis

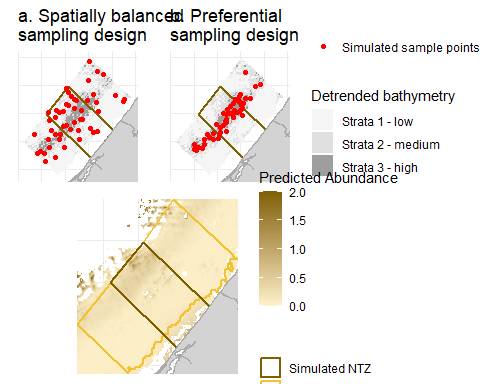
The PCoA and manyglm model highlighted differences in fish community composition between the two sampling designs (Figure @ref(fig:PCoA\_plot)). Community composition was significantly different between sampling design and across depths. The effect of NTZ on community composition was not significant overall, but was significantly different depending on the sampling design (Table @ref(tab:manyglm\_output\_table)).

Table 1: Manyglm model

| Variable | Res.Df | Df.diff | Dev | Pr(>Dev) |  |
| --- | --- | --- | --- | --- | --- |
| (Intercept) | 98 |  |  | NA |  |
| Sampling Design | 97 | 1 | 376.7 | **0.001** | \*\*\* |
| Status | 96 | 1 | 165.0 | 0.121 |  |
| Depth | 95 | 1 | 595.0 | **0.001** | \*\*\* |
| Sampling Design:Status | 94 | 1 | 118.3 | **0.046** | \* |



(#fig:PCoA\_plot)Principal Composite Analysis of community composition.



(#fig:example\_sampling\_design)Examples of a. Spatially balanced sampling design simulation and b. Preferential sampling design simulation around the simulated NTZ. C.Predicted abundance around the simulation area with 80% increase in C. auratus within the simulated NTZ.

## 3.2 Single species detection

(#tab:gam\_table)Generalised Additive model of *C.auratus* abundance

| Variable | Estimate | Std. Error | t value | Pr(>|t|) |
| --- | --- | --- | --- | --- |
| Detrended bathymetry (250m resolution) | 1.841810 | 1.965806 | 5.614574 | 0.0722129 |
| Sand (250m resolution) | 1.864994 | 1.972467 | 8.615501 | 0.0226893 |
| Seagrass (250m resolution) | 1.808135 | 1.962816 | 7.510075 | 0.0376987 |

From all the BRUV samples, a distribution model was fitted to bathymetry and habitat co-variates. The optimal model included sand, seagrass and detrended bathymetry as co-variates, all from the 250m resolution dataset. The final model explained 34.286857% of deviance in the data (Table???).

(#tab:sampling\_design\_mean\_abundance\_summary\_table)Mean abundance obtained from the 2000 sampling design simulations.

Before

After

SD

Fished

No-take

Fished

No-Take

Preferential

0.0935163

0.1069704

0.0937161

0.1935641

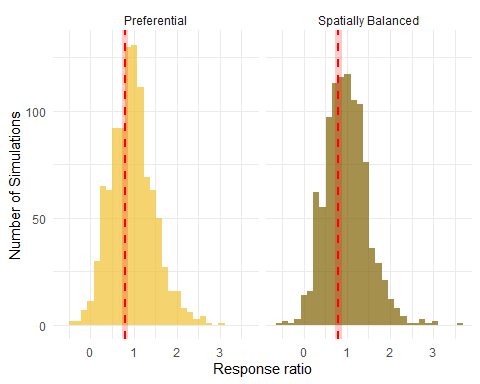
Spatially Balanced

0.1131049

0.1312991

0.1132655

0.2395948



(#fig:ratio\_barplot)Histogram of the difference between the abundance ratio inside/outisde after the NTZ increase, and the abundance ratio inside/outside before the NTZ increase. The red dashed line indicates the 1.8x true increase, and the red box indicates 10% either side of the true increase.

Simulations of the Spatially Balanced sampling design detected on average 2.12 times more fish inside the NTZ than outside (inside: 0.24, outside: 0.11), and correctly detected the 80% (+/- 10%) increase in abundance 6.2% of the time.

The Preferential sampling design detected on average 2.07 times more fish inside the NTZ than outside (inside: 0.19, outside: 0.09), and correctly detected the 80% (+/- 10%) increase in abundance 6.25% of the time.

# 4 Discussion

In this study, the performance of preferential and spatially balanced BRUV sampling designs are compared as MPA monitoring tools for single species detection and community composition at the local scale. We found significant differences in community composition between the two sampling designs, and across depth, but not between inside and outside the MPA. Sampling design simulations were run over a spatial distribution model of Yijarup/Pink Snapper to test the ability of preferential and spatially balanced sampling designs to detect an increase in abundance within an NTZ. The detection of the NTZ effect of a single species was poor, with <10% of sampling design simulations correctly detecting the 80% increased abundance within the NTZ. This study highlights the importance of sampling design choice for BRUV studies, and how both designs may under- or over-estimate the effects of NTZs on single target species such as Yijarup/Pink Snapper.

## 4.1 Community composition

Community composition is known in many fields of ecology to vary with the structure, type, and state of physical and biological gradients in the environment (Williams et al. 2019). Variation in community across depth is not surprising, with examples of temperate reef community composition studies and species ecology highlighting the depth preferences of different species Rees et al. (2021), and/or their affinity to depth-limited habitat (ex. seagrass-associated species may not venture deeper than seagrass meadows’ depth limits, Hutchinson et al. (2014), Kingsford and Carlson (2010)).

The significant difference in community composition between sampling designs is also not surprising, as the spacing and distribution of points within the sampling area is quite different between the two. The habitat association of communities, and the variety of habitat across the sampling area likely means that the preferential design sampled a restricted range of habitat, thus detecting this habitat’s community very well. In contrast, the spatially balanced design may have sampled a wider range of habitats, detecting the whole sampling area’s community well overall, but individual habitats’ communities less thoroughly. Alessi et al. (2023) found the same conclusions when simulating the performance of preferential and probabilistic sampling designs on vegetation communities. They found that preferential sampling detected ‘habitat specialist species’ better, but the probabilistic sampling design detected the richness and diversity of the study area better. This could be roughly summarised as a difference in number of samples per habitat, which alters the accuracy of estimates in population ecology (Stockwell and Peterson 2002). In multivariate community composition analyses like the present study, sample size is just as important, as trends for multiple species must be detected: Irvine, Dinger, and Sarr (2011) simulated the detection ability of different multivariate tests on community composition, and found that sample size had a large effect on trend detection, regardless of the community trend (increasing/decreasing and mixed trend) or the multivariate test.

As a result, a choice must be made between sampling a single habitat very well (like the preferential design) or capturing more habitats perhaps less well (like the spatially balanced design). In the context of MPA monitoring, the added challenge of placing samples so that the effect of the MPA can be assessed correctly complicates things. As the model suggests, sampling designs could perform differently when comparing community composition inside and outside an MPA (@ref(tab:manyglm\_output\_table)). On a local scale, habitats are heterogeneous and MPAs are generally placed to optimise conservation/fisheries outcomes (= on attractive and biodiverse features, not necessarily representing all habitats at this scale, Stevens and Connolly (2005)). Measuring the effect of an MPA on community composition using preferential sampling would require the same features to exist inside and outside the MPA, or an identical unprotected site to be sampled to pair with the MPA site, which is rarely possible, and rarely done in studies (Osenberg et al. 2006; Underwood 1991). Comparatively, spatially balancing samples allows for the categorisation of the area not in features, but in geomorphology (or other environmental variable) strata, which can be more even and fair in the distribution of samples, and allow the assessment of an MPA effect. (am i talking out of my ass)

## 4.2 Single species detection

A very small portion of simulations correctly detected the 80% increase in abundance within the NTZ (Figure @ref(fig:ratio\_barplot)). The difference in the modelled abundance ratio of Yijarup/Pink Snapper inside and outside the simulated NTZ was underestimated in 32.1% and overestimated in 55.4% of preferential simulations. Spatially balanced simulations underestimate this ratio in 30.6% of simulations and overestimated it in 57% of simulations. (Figure @ref(fig:ratio\_barplot)).

An element of concern in the performance of the sampling design simulations is the large portion of simulations (>50%) to overestimate the increase in abundance within the NTZ. A study by Ovando et al. (2021) criticises the use of response ratios to assess population effects of MPAs, as they find they are susceptible to inaccuracy. Their simulations find that a response ratio of 50% was produced in a majority of cases by a population effect of <10% (median 2.5%), greatly overestimating the MPA effect on populations. This study also describes the difficulty of detecting MPA effects when effect sizes are small. Even with an 80% increase in abundance within the simulated NTZ, 1.4% of preferential simulations and 1.3% of spatially balanced simulations do not find an increase in the NTZ (response ratio <0, Figure @ref(fig:ratio\_barplot)). The use of alternative metrics of MPA assessment is essential to correctly describe the performance of MPAs.

Several elements may have contributed to the sampling designs performing similarly: First, Waatern/Geographe Bay has very gentle geomorphology, with few striking features to create attractive habitat in the simulation area. Second, Yijarup/Pink Snapper in this study appears to be a spatially generalist species with even affinity to environmental variables in the simulation area, making the spatial distribution difference between the sampling designs close to irrelevant. The shallow depth of the simulation area may have also played a role in the low and even abundance of mature fish in the model (Heyns-Veale et al. 2016). In a species with stricter affinities to geomorphology and habitat, and a study site with more prominent geomorphology features and habitat differences, the predicted abundance of the species would be less even over the simulation area, and the difference between sampling designs is likely to be much bigger.

## 4.3 Implication

The challenge of monitoring MPAs to assess whether they provide substantial effects

### Recommendations

Alessi, N., G. Bonari, P. Zannini, B. Jiménez-Alfaro, E. Agrillo, F. Attorre, and A. Chiarucci. 2023. “Probabilistic and Preferential Sampling Approaches Offer Integrated Perspectives of Italian Forest Diversity.” *Journal of Vegetation Science* 34 (1): e13175.

Alós, J., and R. Arlinghaus. 2013. “Impacts of Partial Marine Protected Areas on Coastal Fish Communities Exploited by Recreational Angling.” *Fisheries Research* 137: 88–96.

Aubry, P., C. Francesiaz, and M. Guillemain. 2024. “On the Impact of Preferential Sampling on Ecological Status and Trend Assessment.” *Ecological Modelling* 492: 110707.

Borja, A., and M. Elliott. 2021. “From an Economic Crisis to a Pandemic Crisis: The Need for Accurate Marine Monitoring Data to Take Informed Management Decisions.” *Advances in Marine Biology* 89: 79–114.

Brown, C. J., M. I. Saunders, H. P. Possingham, and A. J. Richardson. 2013. “Managing for Interactions Between Local and Global Stressors of Ecosystems.” *PloS One* 8 (6): e65765.

Bryan, D. R., S. G. Smith, J. S. Ault, M. W. Feeley, and C. W. Menza. 2016. “Feasibility of a Regionwide Probability Survey for Coral Reef Fish in Puerto Rico and the u.s. Virgin Islands.” *Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science* 8 (1): 135–46.

Cameron, M. J., V. Lucieer, N. S. Barrett, C. R. Johnson, and G. J. Edgar. 2014. “Understanding Community-Habitat Associations of Temperate Reef Fishes Using Fine-Resolution Bathymetric Measures of Physical Structure.” *Marine Ecology Progress Series* 506: 213–29.

Cheng, W., C. Zhang, Y. Ji, Y. Xue, Y. Ren, and B. Xu. 2024. “Performance Evaluation of Spatially Balanced Sampling Designs in Fishery-Independent Surveys.” *Fisheries Research* 270: 106879.

Cheung, W. W. L., M. C. Jones, G. Reygondeau, and T. L. Frölicher. 2018. “Opportunities for Climate-Risk Reduction Through Effective Fisheries Management.” *Global Change Biology* 24 (11): 5149–63.

Conn, P. B., J. T. Thorson, and D. S. Johnson. 2017. “Confronting Preferential Sampling When Analysing Population Distributions: Diagnosis and Model‐based Triage.” *Methods in Ecology and Evolution* 8 (11): 1535–46.

Davies, H., W. Webb, I. Webb, T. Webb, D. Guilfoyle, S. Clohessy, K. Griffin, and T. Langlois. 2022. “The Cultural Seascape of Wadandi Boodja: The Cultural Values of Australia’s South-West Marine Parks.” <https://www.nespmarinecoastal.edu.au/wp-content/uploads/2023/12/Davies_Langlois-et-al_D3_The-Cultural-Seascape-of-Wadandi-Country_Sep2022_Report.pdf>.

Dumelle, M., T. Kincaid, A. R. Olsen, and M. Weber. 2023. “Spsurvey: Spatial Sampling Design and Analysis in r.” *Journal of Statistical Software* 105 (3): 1.

Fernández, T. V., G. D’anna, F. Badalamenti, and A. Pérez-Ruzafa. 2008. “Habitat Connectivity as a Factor Affecting Fish Assemblages in Temperate Reefs.” *Aquatic Biology* 1 (3): 239–48.

Ferraro, P. J., and S. K. Pattanayak. 2006. “Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments.” *PLoS Biology* 4 (4): e105.

Fisher, R. 2024. *FSSgam: Full Subsets Multiple Regresssion in r Using Gam(m4)*. <https://github.com/beckyfisher/FSSgam_package>.

Fisher, R., S. K. Wilson, T. M. Sin, A. C. Lee, and T. J. Langlois. 2018. “A Simple Function for Full‐subsets Multiple Regression in Ecology with r.” *Ecology and Evolution* 8 (12): 6104–13.

Fitzpatrick, B. M., E. S. Harvey, A. J. Heyward, E. J. Twiggs, and J. Colquhoun. 2012. “Habitat Specialization in Tropical Continental Shelf Demersal Fish Assemblages.” *PloS One* 7 (6): e39634.

Foster, S. D., J. Monk, E. Lawrence, K. R. Hayes, G. R. Hosack, and R. Przeslawski. 2018. “Statistical Considerations for Monitoring and Sampling.” In *Field Manuals for Marine Sampling to Monitor Australian Waters*, 1:23–41.

Galaiduk, R., R. Nanson, Z. Huang, S. Nichol, and K. Miller. 2018. “An Eco-Narrative of Geographe Marine Park – South-West Marine Region.” <https://www.researchgate.net/profile/Ronen-Galaiduk/publication/330409162_An_eco-narrative_of_Geographe_Marine_Park_South-west_marine_region_Report_to_the_National_Environmental_Science_Programme_Marine_Biodiversity_Hub_Geoscience_Australia/links/5c3e83f392851c22a3785da6/An-eco-narrative-of-Geographe-Marine-Park-South-west-marine-region-Report-to-the-National-Environmental-Science-Programme-Marine-Biodiversity-Hub-Geoscience-Australia.pdf>.

Gaynor, A., A. Kendrick, and M. Westera. 2008. “An Oral History of Fishing and Diving in the Capes Region of South-West Western Australia.”

Gissi, E., E. Manea, A. D. Mazaris, S. Fraschetti, V. Almpanidou, S. Bevilacqua, M. Coll, et al. 2021. “A Review of the Combined Effects of Climate Change and Other Local Human Stressors on the Marine Environment.” *The Science of the Total Environment* 755: 142564.

Harasti, D., T. R. Davis, A. Jordan, L. Erskine, and N. Moltschaniwskyj. 2019. “Illegal Recreational Fishing Causes a Decline in a Fishery Targeted Species (Snapper: Chrysophrys Auratus) Within a Remote No-Take Marine Protected Area.” *PloS One* 14 (1): e0209926.

Harasti, D., J. Williams, E. Mitchell, S. Lindfield, and A. Jordan. 2018. “Increase in Relative Abundance and Size of Snapper Chrysophrys Auratus Within Partially-Protected and No-Take Areas in a Temperate Marine Protected Area.” *Frontiers in Marine Science* 5: 334780.

Hayes, K. R., G. R. Hosack, E. Lawrence, P. Hedge, N. S. Barrett, R. Przeslawski, M. J. Caley, and S. D. Foster. 2019. “Designing Monitoring Programs for Marine Protected Areas Within an Evidence-Based Decision Making Paradigm.” *Frontiers in Marine Science* 6. <https://doi.org/10.3389/fmars.2019.00746>.

Heyns-Veale, E. R., A. T. F. Bernard, N. B. Richoux, D. Parker, T. J. Langlois, E. S. Harvey, and A. Götz. 2016. “Depth and Habitat Determine Assemblage Structure of South Africa’s Warm-Temperate Reef Fish.” *Marine Biology* 163: 1–17.

Hijmans, R. J., J. Van Etten, and J. Cheng. 2015. *Package "Raster"*.

Hutchinson, N., G. P. Jenkins, A. Brown, and T. M. Smith. 2014. “Variation with Depth in Temperate Seagrass-Associated Fish Assemblages in Southern Victoria, Australia.” *Estuaries and Coasts* 37 (4): 801–14.

Irvine, K. M., E. C. Dinger, and D. Sarr. 2011. “A Power Analysis for Multivariate Tests of Temporal Trend in Species Composition.” *Ecology* 92 (10): 1879–86.

Kelaher, B. P., M. A. Coleman, A. Broad, M. J. Rees, A. Jordan, and A. R. Davis. 2014. “Changes in Fish Assemblages Following the Establishment of a Network of No-Take Marine Reserves and Partially-Protected Areas.” *PloS One* 9 (1): e85825.

Kermorvant, C., F. D’Amico, N. Bru, N. Caill-Milly, and B. Robertson. 2019. “Spatially Balanced Sampling Designs for Environmental Surveys.” *Environmental Monitoring and Assessment* 191 (8): 524.

Kingsford, M. J., and I. J. Carlson. 2010. “Patterns of Distribution and Movement of Fishes, Ophthalmolepis Lineolatus and Hypoplectrodes Maccullochi, on Temperate Rocky Reefs of South Eastern Australia.” *Environmental Biology of Fishes* 88: 105–18.

Langlois, T., J. Goetze, T. Bond, J. Monk, R. A. Abesamis, J. Asher, N. Barrett, et al. 2020. “A Field and Video Annotation Guide for Baited Remote Underwater Stereo‐video Surveys of Demersal Fish Assemblages.” *Methods in Ecology and Evolution* 11 (11): 1401–9.

Ling, S. D., and C. R. Johnson. 2012. “Marine Reserves Reduce Risk of Climate-Driven Phase Shift by Reinstating Size- and Habitat-Specific Trophic Interactions.” *Ecological Applications: A Publication of the Ecological Society of America* 22 (4): 1232–45.

Liu, Y., Y. Chen, and J. Cheng. 2009. “A Comparative Study of Optimization Methods and Conventional Methods for Sampling Design in Fishery-Independent Surveys.” *ICES Journal of Marine Science* 66 (9): 1873–82.

Malcolm, H. A., J. Williams, A. L. Schultz, J. Neilson, N. Johnstone, N. A. Knott, D. Harasti, M. A. Coleman, and A. Jordan. 2018. “Targeted Fishes Are Larger and More Abundant in ‘No-Take’ Areas in a Subtropical Marine Park.” *Estuarine, Coastal and Shelf Science* 212: 118–27.

Miller, K. I., and G. R. Russ. 2014. “Studies of No-Take Marine Reserves: Methods for Differentiating Reserve and Habitat Effects.” *Ocean & Coastal Management* 96: 51–60.

Miteva, D. A., S. K. Pattanayak, and P. J. Ferraro. 2012. “Evaluation of Biodiversity Policy Instruments: What Works and What Doesn’t?” *Oxford Review of Economic Policy* 28 (1): 69–92.

Oksanen, J., F. G. Blanchet, R. Kindt, P. Legendre, P. R. Minchin, R. B. O’hara, and M. Oksanen. 2013. *Package "Vegan." Community Ecology Package*. Vol. 2.

Osenberg, C. W., B. M. Bolker, J. S. S. White, C. M. St Mary, and J. S. Shima. 2006. *Statistical Issues and Study Design in Ecological Restorations: Lessons Learned from Marine Reserves*.

Ovando, Daniel, Jennifer E. Caselle, Christopher Costello, Olivier Deschenes, Steven D. Gaines, Ray Hilborn, and Oliver Liu. 2021. “Assessing the Population‐level Conservation Effects of Marine Protected Areas.” *Conservation Biology* 35 (6): 1861–70. <https://doi.org/10.1111/cobi.13794>.

Parsons, D. M., D. Buckthought, C. Middleton, and G. MacKay. 2016. “Relative Abundance of Snapper (Chrysophrys Auratus) Across Habitats Within an Estuarine System.” *New Zealand Journal of Marine and Freshwater Research* 50 (3): 358–70.

Rees, M. J., N. A. Knott, M. L. Hing, M. Hammond, J. Williams, J. Neilson, and A. Jordan. 2021. “Habitat and Humans Predict the Distribution of Juvenile and Adult Snapper (Sparidae: Chrysophrys Auratus) Along Australia’s Most Populated Coastline.” *Estuarine, Coastal and Shelf Science* 257: 107397.

Ryan, K. L., C. B. Smallwood, and E. K. Lai. 2022. “Boat-Based Recreational Fishing in Western Australia 2020/21.” Western Australia: Department of Primary Industries; Regional Development.

Schramm, K. D., E. S. Harvey, J. S. Goetze, M. J. Travers, B. Warnock, and B. J. Saunders. 2020. “A Comparison of Stereo-BRUV, Diver Operated and Remote Stereo-Video Transects for Assessing Reef Fish Assemblages.” *Journal of Experimental Marine Biology and Ecology* 524: 151273.

Scott, M. E., S. B. Tebbett, K. L. Whitman, C. A. Thompson, F. B. Mancini, M. R. Heupel, and M. S. Pratchett. 2022. “Variation in Abundance, Diversity and Composition of Coral Reef Fishes with Increasing Depth at a Submerged Shoal in the Northern Great Barrier Reef.” *Reviews in Fish Biology and Fisheries* 32 (3): 941–62.

Smith, S. G., J. S. Ault, J. A. Bohnsack, D. E. Harper, J. Luo, and D. B. McClellan. 2011. “Multispecies Survey Design for Assessing Reef-Fish Stocks, Spatially Explicit Management Performance, and Ecosystem Condition.” *Fisheries Research* 109 (1): 25–41.

Stevens, Tim, and Rod M. Connolly. 2005. “Local-Scale Mapping of Benthic Habitats to Assess Representation in a Marine Protected Area.” *Marine and Freshwater Research* 56 (1): 111–23. <https://doi.org/10.1071/MF04145>.

Stockwell, D. R., and A. T. Peterson. 2002. “Effects of Sample Size on Accuracy of Species Distribution Models.” *Ecological Modelling* 148 (1): 1–13.

Switzer, T. S., S. F. Keenan, K. A. Thompson, C. P. Shea, A. R. Knapp, M. D. Campbell, and M. C. Christman. 2023. “Integrating Assemblage Structure and Habitat Mapping Data into the Design of a Multispecies Reef Fish Survey.” *Marine and Coastal Fisheries* 15 (4): e10245.

Taylor, B. M., and J. L. McIlwain. 2010. “Beyond Abundance and Biomass: Effects of Marine Protected Areas on the Demography of a Highly Exploited Reef Fish.” *Marine Ecology Progress Series* 411: 243–58.

Terres, M. A., E. Lawrence, G. R. Hosack, M. D. E. Haywood, and R. C. Babcock. 2015. “Assessing Habitat Use by Snapper (Chrysophrys Auratus) from Baited Underwater Video Data in a Coastal Marine Park.” *PloS One* 10 (8): e0136799.

Underwood, A. 1991. “Beyond BACI: Experimental Designs for Detecting Human Environmental Impacts on Temporal Variations in Natural Populations.” *Marine and Freshwater Research* 42: 569–87.

Wakefield, C. B., I. C. Potter, N. G. Hall, R. C. J. Lenanton, and S. A. Hesp. 2015. “Marked Variations in Reproductive Characteristics of Snapper (Chrysophrys Auratus, Sparidae) and Their Relationship with Temperature over a Wide Latitudinal Range.” *ICES Journal of Marine Science: Journal Du Conseil* 72 (8): 2341–49.

Wang, Y. I., U. Naumann, S. T. Wright, and D. I. Warton. 2012. “Mvabund–an r Package for Model-Based Analysis of Multivariate Abundance Data.” *Methods in Ecology and Evolution* 3 (3): 471–74.

Watson, D. L., M. J. Anderson, G. A. Kendrick, K. Nardi, and E. S. Harvey. 2009. “Effects of Protection from Fishing on the Lengths of Targeted and Non-Targeted Fish Species at the Houtman Abrolhos Islands, Western Australia.” *Marine Ecology Progress Series* 384: 241–49.

Watson, D. L., E. S. Harvey, G. A. Kendrick, K. Nardi, and M. J. Anderson. 2007. “Protection from Fishing Alters the Species Composition of Fish Assemblages in a Temperate-Tropical Transition Zone.” *Marine Biology* 152 (5): 1197–1206.

Whitmarsh, S. K., P. G. Fairweather, and C. Huveneers. 2017. “What Is Big BRUVver up to? Methods and Uses of Baited Underwater Video.” *Reviews in Fish Biology and Fisheries* 27 (1): 53–73.

Williams, J., A. Jordan, D. Harasti, P. Davies, and T. Ingleton. 2019. “Taking a Deeper Look: Quantifying the Differences in Fish Assemblages Between Shallow and Mesophotic Temperate Rocky Reefs.” *PloS One* 14 (3): e0206778.

Wood, S. N. 2001. “Mgcv: GAMs and Generalized Ridge Regression for r.” *R News*. <https://journal.r-project.org/articles/RN-2001-015/RN-2001-015.pdf>.

Zhao, J., J. Cao, S. Tian, Y. Chen, and S. Zhang. 2018. “Evaluating Sampling Designs for Demersal Fish Communities.” *Sustainability* 10 (8): 2585.