Effectiveness of community and volunteer based coral reef monitoring in Cambodia

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# Abstract

1. Globally, coral reef monitoring programmes conducted by volunteer-based organizations or local communities have the potential to collect large quantities of marine data at low cost. However, many scientists remain sceptical about the ability of these programmes to detect changes in marine systems when compared with professional techniques.
2. A limited number of studies have assessed the efficacy and validity of volunteer-based monitoring, and even fewer have assessed community-based methods.
3. This study in Cambodia investigated the ability of surveyors of different levels of experience to conduct underwater surveys using a simple coral reef methodology. Surveyors were assigned to four experience categories and conducted a series of six 20 x 5 m belt transects using five benthic indicator species.
4. Results indicate decreased variation in marine community assessments with increasing experience, indicating that experience, rather than cultural background, influences survey ability. This suggests that locally based programmes can fill gaps in knowledge with suitable on-going training and assessment.

# Key Words

Coral, Reef, Island, Monitoring, Survey, Conservation evaluation, Participatory monitoring, Citizen science

# Introduction

Within developing communities there is a clear need for coastal communities to have access to simple marine monitoring techniques that are able to assess reef health (Risk *et al.,* 2001). Such communities are invariably resource limited and the costs and logistical constraints associated with the effective monitoring of marine resources by professional scientists can be crippling to management efforts. Orthodox methods to monitor biodiversity and resource use are often costly and hard to sustain, especially in developing countries where financial resources are limited (Danielsen *et al*., 2005) and the results of such studies are rarely communicated to the local population (Uychiaoco *et al.,* 2005). As of April 2015, there were 11,333 recognized global Marine Protected Areas (MPAs) covering 2.12% of the oceans, with 0.94% classed as 'no-take' zones (Marine Conservation Institute, 2015). However, without reliable monitoring of these MPAs, no supportable management decisions can be reached, no diagnoses on the state of fisheries can be performed and no progress on the effects of management control can be made (FAO, 2002; Lunn and Dearden, 2006).

While there is a clear need for monitoring, it is less clear who should undertake it. In the past, reef monitoring was typically conducted by professional marine biologists: SCUBA divers trained in the scientific identification of marine organisms (Uychiaoco *et al.,* 2005), non-governmental organisations or foreign specialists (Léopold *et al.,* 2009). However, there is the general consensus that science has neither the financial resources or man power to meet the current demand (Hodgson 1999; Foster-Smith and Evans, 2003). Monitoring approaches using non-resident volunteers and local communities have been receiving increasing attention as a cost-effective way to collect data on the environment and/or involve local stakeholders in management (Léopold *et al.,* 2009; Mascia 2001). In the last 2-3 decades there has been a progressive shift towards involving foreign volunteers and local community members formally and more effectively in monitoring and management (Sheil and Lawrence, 2004; Wells and McShane, 2004; Danielsen *et al.,* 2005). Subsequently, the scope of coral reef monitoring systems has been expanded to include monitoring programmes specifically designed for volunteer divers such as the Reef Check programme (Hodgson, 1999) and the Global Coral Reef Monitoring Network (Hill and Wilkinson, 2004). Each of these programmes utilises variations of the same transect techniques to assess fish, invertebrate and substrate communities (for detailed methodological information see: Hodgson (1999) and Hill and Wilkinson (2004)). In most developing countries, local communities do not have access to SCUBA equipment and are therefore not able to be directly involved in monitoring with professionals and non-resident volunteers (Uychiaoco *et al.,* 2005).

Nevertheless, many members of the scientific community remain sceptical about the reliability of data collected by volunteers (Foster-Smith and Evans, 2003). Species and habitat monitoring data collected by local communities and external agencies has, in the past, been assumed to yield the same results as collected by scientists (Coleman & Steed, 2009) but there has only been minimal testing of this assumption (Danielsen *et al.,* 2014). This is particularly true in the marine environment, as a majority of studies attempting to determine the suitability of community based resource monitoring have focussed on terrestrial ecosystems such as forests (Constantino *et al.*, 2012; Funder *et al.,* 2013; Danielsen *et al*., 2014). In fact, previous studies using community members to assess fish populations have suggested that community members are more likely to either under- or over-estimate species counts (Bray and Schramm, 2001; Léopold *et al.,* 2009). Whilst there are many examples of how to integrate volunteers and community members into the collection of monitoring data, through interviews designed to assess community perceptions of fish populations (Yasué *et al.,* 2010) and catch assessments with behavioural observations (Lunn and Dearden, 2006), this study is focussed on the use of non-resident volunteers and local community members to conduct in-water assessments of reef resources.

Previous studies have compared the data collected by scientists with data from either volunteers (Mumby *et al.,* 1995, Darwall and Dulvy, 1996; Goffredo *et al.,* 2010) or local communities (Uychiaoco *et al.,* 2005). However, attempts at determining any differences between scientist-gathered data and data collected by both volunteers and local communities was not possible due to differences in the methodologies used by each group (Léopold *et al.,* 2009). Past studies have also varied in regards to the indicators and groups used. A majority of studies addressing the abilities of non-resident volunteers and community members to assess coral reefs have focussed on the assessment of fish populations (Uychiaoco *et al.,* 2005; Léopold *et al.,* 2009). While it makes sense to monitor local fish stocks, and fish species are often known or more identifiable to the local communities, the issue lies in the fact that a majority of fish species are motile: assessment of individuals’ ability to accurately quantify fish populations is inherently difficult when no scientist, volunteer or community member is necessarily seeing the same fish. While this is not an issue unique to coral reef surveys, it is understandably difficult to count individual fish in a three-dimensional space with target and non-target individuals moving in and out of the survey area, particularly for those with limited experience. Although it is important to consider the assessment of fish, particularly in areas where fish are potentially over-harvested by resource dependent communities, it is arguably more appropriate first to establish how accurately marine surveyors are quantifying marine communities before expanding studies to include groups which are more difficult to assess. Analysis of sedentary marine invertebrates and substrate indicators could offer an alternative means of coral health assessment, because the indicators in question do not move.

The aim of this study was to compare the abilities of coral reef surveyors with different levels of experience to assess benthic invertebrate and substrate indicators, to establish the potential application of volunteer and community-based monitoring to coral reef MPA management. More formally we tested the following hypotheses: (1) surveyor precision and accuracy will be greater in groups with more experience; (2) local community members will show the greatest variation in survey data; and, finally, (3) variation will decrease with the survey experience participants gained in this survey.

# Methods

Study Site

This study was conducted on the north-eastern side of Koh Sdach Island, in the Koh Sdach Archipelago, adjacent to the Botom Sakor National Park in Cambodia (Figure. 1a, b).

Study Participants

The study was conducted using a total of 16 participants allocated to four experience categories (four participants in each category) based on the following criteria:

* *Khmer* - Local community residents interested in conservation; able to swim and snorkel, but no experience of scientific surveys.
* *Non-resident volunteers with no experience* - International paying volunteers on a marine conservation programme with no survey training or prior experience.
* *Non-resident volunteers with experience* - International paying volunteers; fully trained with minimum of two months survey experience.
* *Scientists* - Minimum of a bachelor's degree in an environmental discipline, with one-year marine survey experience.

Participants aged between 18 and 40 (eight males and eight females), from mixed cultural and educational backgrounds, were selected based on their involvement in environmental projects on Koh Sdach and their adherence to the above criteria. Khmer participants were involved in environmental education and outreach programmes and expressed interest in trialling a new community-based monitoring programme. International volunteers were sourced through an international volunteer organisation, Projects Abroad, and were interested in gaining experience in marine monitoring techniques. All participants gave their time voluntarily as a means of gaining experience, and learning new skills.

Indicator Species

Due to the involvement of inexperienced surveyors, the number of indicators for this study was kept small rather than setting unrealistic targets for new surveyors, with a view to increasing it in subsequent studies.

Indicator species were required to meet all of the following criteria:

1. Regionally accepted as appropriate indicators of reef stress (Risk *et al*., 2001).
2. Conspicuous, easily identifiable and known to the local community members.
3. Diurnally sedentary, ensuring that individual organisms did not move out of the survey area between surveys.

A series of five benthic marine invertebrate species were selected, which fulfilled the above criteria and were capable of acting as indicators of stress in coral reef systems.

* Boring Sponge (*Cliona* sp.)
* Christmas Tree Worm (*Spirobranchus giganteus*)
* Feather Duster Worms (*Sabellastarte* sp.)
* Giant Clam (*Tridacna* sp.)
* Long-spine Sea Urchin (*Diadema* sp.)

Four of these indicators were boring species, which embed themselves directly into the coral structure. The rationale was that a stressed coral will be less able to defend itself against colonisation by these species so, the higher the concentration of these indicators, the more stressed the coral (Risk *et al.,* 2001).

Training

All participants underwent training during a one-hour workshop, where they were introduced to the indicator species through a series of photographs and drawings and to appropriate survey techniques and swim patterns. Practice ‘dry-run’ surveys were set-up using a series of photographs of both indicator and non-indicator species arranged around a transect line on the floor of the training room. Participants were encouraged to swim in an ‘S’ shape across the survey area, taking as long as they needed, and choose whether to count each species individually, or as many at once as they felt they could manage. A translator was used where required to ensure clear understanding by all participants.

Protocol

Six fixed-position belt transects (Hodgson, 1999; Hill and Wilkinson, 2004) were deployed in a 50 m2 area of shallow rocky reef (Figure 1c). Each survey was conducted over a 20 x 5 m belt transect (extending 2.5 m each side of the transect line), with a minimum distance of 5 m between each line. The maximum depth was 1.5 m ensuring participants were able to surface dive if necessary. Since the purpose of this analysis was to assess surveyor accuracy and not reef health, the proximity of transects was not a scientific consideration, but assisted with health and safety, considering the large number of inexperienced surveyors in the water.

Transects were secured below the water line to concrete posts used for pier construction, and the deployment bearing was recorded. All participants were briefed together (a translator was used to ensure the Khmer participants were fully informed).

Prior to the start of the participant surveys, a line point transect (Hodgson, 1999) was conducted with data points collected at 50 cm intervals, to determine the general condition of each survey area. Each participant surveyed the six belt transects using basic snorkelling and safety equipment. Surveys were carried out one by one and participants were asked to record the number of each indicator seen within the survey area. Following each transect, there was a rest period. This allowed participants to rest, data to be collated, and allowed time for the marine communities to resettle following any disturbance.

Data Analysis

Data were analysed using the Bray-Curtis index of similarity (Bray and Curtis, 1957) to create a rank similarity matrix and subsequently construct a Multidimensional Scaling (MDS) plot to look at overall trends in marine communities detected on each transect by the different experience groups. The MDS plots were further examined by the addition of percentage similarity contours, indicating the maximum similarity between individuals to aid visualization.

A Permutational Multivariate Analysis of Variance (PERMANOVA) was conducted using Bray-Curtis similarity coefficients (Bray and Curtis, 1957), calculated using square-root transformed data to stabilize variances (Anderson *et al.*, 2008). For the analysis, 999 unrestricted random permutations of residuals were used to generate P-values (Anderson *et al.*, 2008). It included the factor ‘experience level’ (Khmer, non-experienced volunteers, experienced volunteers and scientists) across the six transects used in the study. The results of the Bray-Curtis similarity index were then used to conduct a similarity percentage (SIMPER) analysis (Anderson *et al.*, 2008) to assess the percentage contribution of individual species to the similarity matrix produced by the Bray-Curtis index of similarity.

Based on the assumption that participants in the ‘scientist’ category are accurately counting the number of each species in each transect area (Mumby *et al.,* 1995; Uychiaoco *et al.,* 2005; Léopold *et al.,* 2009), additional analyses were conducted to assess the impact of experience on survey accuracy. These were calculated using the sum of the absolute deviations between the mean score of scientists for each species, and the species counts recorded by each surveyor in the remaining three experience categories (Eq. 1), plotted against the order in which the surveys were conducted.

$$\sum\_{}^{}\left(^{}/\_{}\sum\_{}^{}\left|\_{}\right.\left.\overbar{}\right|\right)$$

(Eq. 1)

Where *n* = 4, sp is the number of species recorded, *O* refers to the observer*, T* the order in which each transect was conducted, and *Sci* is the mean score for the scientist group at each specific transect. This analysis was supported by subsequent Spearman’s Rank Correlations for each non-scientist experience level to assess relationships between the sum of the absolute deviations and the order in which the surveys were completed.

# Results

Results indicated that there was less within-group variation with increasing experience (Figure 2), with participants in the scientist category showing the highest percentage similarity in counts of indicator species in each transect. Overall, results varied by transect, related to the individual substrate breakdown of each survey area (Table 1). However, there was evidence of particular observers’ consistently reporting species counts that varied from other participants’ (observer one and observer three, Figure 2). This was supported by an increase in similarity of detected communities between individuals with increasing experience. This indicated that individuals in the scientist group demonstrated the highest level of precision whereas participants in the Khmer produced the greatest variation between participants’ recordings. This partially supports the initial hypothesis, indicating that precision increases with experience. However, it was not possible to assess surveyor accuracy due to the variation present in each individual participant’s assessment of the five indicator species over the six transects. Similarly, the results support the hypothesis that there will be more variation in the data collected by the local community members.

Results from the PERMANOVA vary from transect to transect (Table 2). Overall there is evidence of significant differences in the indicators detected by participants in each experience group. However, the results also indicate many instances where there were no significant differences in counts of the five indicator species between experience groups.

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There were no significant differences between assessments of the five indicator species made by the local community members and those made by the non-experienced volunteers on any of the transects (Table 1). This suggests that the differences are not due to any cultural background, but are due to experience.

The SIMPER analysis (Tables 3 and 4) suggested that the variation in the community composition assessments was caused by two species, long-spine sea urchins (*Diadema* sp.) and feather duster worms (*Cliona* sp.), irrespective of surveyor experience. These were also the most abundant species on each transect.

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Subsequent analysis assessing the impact of the experience gained during this study on survey accuracy was conducted using the mean scores of participants in the Scientist experience category as a baseline (Figure. 3). There was a decrease in variance in the non-experienced international volunteers and experienced international volunteers, while the Khmer volunteers showed increase variation in results from the scientist baseline. Subsequent Spearman’s Rank correlations showed no significant relationship between the sum of residuals for participants in the Khmer and non-experienced volunteer categories and the order in which surveys were conducted, suggesting that the six surveys of experience obtained did not lead to more accurate counts of the five indicator species. However, a significant negative correlation was detected between the survey order and the sum of residuals in the experienced volunteer category (rs (6) = -0.886 *P* = 0.019). This shows that there was less variation in the data collected by the experienced volunteers and the mean scientist scores with increasing survey experience. This disproves the third hypothesis that variation would decrease with the experience gained in this study.

# Discussion

There is a consensus that science has neither the manpower nor the financial resources to meet the demands that are being placed upon it (Hodgson, 1999; Foster-Smith and Evans, 2003). Participatory monitoring aims to bring together conservationists and members of the public to collect data about changes in nature (Staddon *et al*., 2014) and, past literature has suggested that involving local communities in monitoring programmes could assist with the immense amounts of monitoring required to assess trends in natural resources around the world (Sodhi and Ehrlich, 2010; Danielsen *et al.,* 2014). However, species and habitat monitoring conducted by local communities has, in forest communities been assumed to yield the same results as data collected by scientists (Coleman and Stead, 2009; Danielsen *et al.,* 2014), but there has been little empirical testing of this (Danielsen *et al*., 2014), particularly in coral reef environments. This study has shown that, irrespective of experience groups, individuals can make different assessments of the abundance of each indicator species within the survey area. Examination of the MDS plots (Figure 2) indicated less variation in the non-experienced volunteer category, which was not apparent with the local community members.

The PERMANOVA analysis (Table 2) was used to determine significant differences between experience groups in terms of the counts made of the five indicator species in each of the six study areas. Results varied between experience groups and across transects. Of the 36 comparisons made, 25 showed no significant differences in the assessments of the five indicator species. However, this does not mean that the participants in each group are collecting similar counts for each of the transects. The only comparison which remained non-significant across all transects was the Khmer and non-experienced volunteers. While this does not infer whether one group is more or less able than the other, the results do indicate that there are no significant differences in the assessments of indicators in each group, suggesting that it is the lack of experience in these two groups causing the results rather than cultural background. Significant differences in assessments made of the five indicator species between the other experience groups varied across the six transects however, this was not consistent. It is possible that this variation is as a result of differences in the general health and composition of each of the study areas (Table 1). Despite each of the surveys being conducted within a small area, there were notable differences in the status the reef beneath each transect as shown in the substrate summary (Table 1).

For the purposes of this study, it was not important for the survey sites to be similar in terms of reef composition but these results suggest that surveyor ability may vary under different reef conditions, particularly as the presence and absence of indicators may be dependent on the underlying substrate. Such assumptions are reasonable in the context of this study since some of the indicator species used here bore into the coral colony (i.e. feather duster worms, giant clams and boring sponge) or graze on macro-algae (i.e. long-spine sea urchin). As a result, there may be differences in the relative precision of surveyors in areas that show higher or lower abundances of live coral, dead coral and algae etc. Previous studies on surveyor accuracy have suggested that variables such as depth can have an impact on surveyor performance (Mumby *et al.,* 1995). As such, it would also be wise to determine if other variables such as reef composition impact surveyor ability.

While it has been understood for some time that data derived from locally-based monitoring typically demonstrates higher variance than professional studies (Leopold *et al.*, 2009), past studies assessing the survey accuracy and ability of volunteers conducting marine surveys have used scientists or ‘experienced surveyors’ as a control (Mumby *et al.,* 1995; Darwall & Dulvy, 1996; Uyachioco *et al.,* 2005; Léopold *et al.,* 2009; Goffredo *et al.,* 2010). This suggests that the data collected by scientists is generally considered to be accurate. The variation in species count data collected by scientists suggests that while more precise, there is no way to determine accuracy. The variation seen within the scientist group (Figure 2) suggests that the individual scientists are detecting different abundances of each of the indicator species. Therefore, assessing the accuracy of any surveyors or technique is difficult. One alternative would be to develop simulated survey areas, which are constructed in such a way that the abundance of key indicators is known. Conducting surveys in a simulated area such as this, which mimics a lab environment would facilitate reliable studies of surveyor accuracy. This would also facilitate further research aimed at determining whether the indicator counts and subsequent community assessments made by individuals of different survey experience are accurate and reliable enough to detect changes in marine systems over time and, similarly, which indicators are most suitable.

Aspects of this and other studies dependent on data collected by scientists as an accurate baseline may not be reliable; but certain assumptions about the accuracy of the data collected by scientists were necessary for this analysis. In the absence of an accurate and reliable baseline, it was necessary to consider the data collected by scientists as an accurate assessment of each indicator species. This assumption has formed the basis of a number of similar assessments determining the suitability of community members and volunteers to collect marine survey data (Mumby *et al*., 1995; Uychiaoco *et al.,* 2005; Léopold *et al*., 2009). However, the results of the MDS analysis (Figure 2) show that this is not necessarily the case. The clustering of the scientist group markers suggests increased precisions within the group, but it is not possible to determine accuracy. At present, there is no meaningful alternative to using scientists in this manner, but results from this and similar studies should be considered with this assumption in mind. Subsequently, when considering the results of the effect of six surveys’ worth of experience, it was also necessary to assume that the scientists’ abilities remained constant throughout the six surveys.

When assessing the potential impact of six surveys worth of experience on surveyor accuracy, there lack of significant correlation between the participants in the two categories with no experience. This, combined with the significant negative correlation showing improvement in relative accuracy for the participants in the experienced volunteers category indicates that experience is a key factor in determining the success of individual surveys. It is clear that the six surveys’ worth of experience was no enough to provide accurate results; however, the underlying reasons could vary. Considering all participants had a minimum of high-school level education and, aside from cultural distinctions, the biggest difference between participants in the Khmer category and the other groups was being taught to swim formally. Each of the international volunteers had enrolled on a programme that involved learning to SCUBA dive and all scientists were already certified divers and so met the swimming. Contrastingly, the Khmer participants had limited experience in the water, and had not received any formal training. While the Khmer participants were able to swim and meet the safety requirements of the study, there was a distinct difference in the apparent comfort in the water. In-water comfort is a key factor in this analysis. Regardless of time taken to complete the transects, and both the number and duration of rest periods, conducting six surveys in a single day can be tiring; particularly if spending extended periods in the water is new, and this could explain the apparent decrease in survey ability in the Khmer group as more surveys were conducted: exhaustion increases and subsequently participants may not have been paying as much attention to what they were counting or were hurrying to complete the task.

For the international volunteers, those in the non-experienced category showed a small improvement with experience. Although this was not statistically significant, the data suggests that improvement may continue to increase with experience, as seen with the experienced volunteers. While the current study has not provided data on how long it would take for the non-experienced international volunteers to gain beneficial experience, conducting similar assessments throughout the training of new volunteers could indicate whether current training programmes are sufficient to elicit reliable monitoring data and, similarly, are appropriate for both international volunteers and local community members. There may be a subsequent need to establish alternative, more culturally- and locally-adapted training programmes.

There are numerous studies suggesting that volunteers (Darwall and Dulvy, 1996; Thomas, 1996; Mumby *et al.,* 1999; Goffredo *et al.* 2010) and community members (Constantino *et al.,* 2012; Danielsen *et al.,* 2014) can gather the monitoring data which is the same as professional scientists. However, a majority of these studies are based on work conducted in the terrestrial biome. Marine studies have typically shown greater variation in survey ability, particularly with reference to variables such as depth (Mumby *et al.,* 1999) and duration of survey (Darwall and Dulvy, 1996). It is likely that aspects of the variation seen in this and previous studies, are related the inherent difficulties with conducting surveys underwater, and as discussed above, the comfort of surveyors, whatever their experience or academic qualification. Particularly when undertaking new tasks.

The purpose of this study was not to design a new programme for monitoring coral reef resources, but does suggest traits of techniques, individuals and indicators which can be developed further, with the aim of developing a set of appropriate methodologies. While this analysis represents an initial assessment of the potential of community and volunteer based monitoring (and there is much more to be done) it does offer the opportunity to assess and fine-tune the techniques and indicators used before continuing to establish their overall suitability and potential. The indicators selected in this study were chosen primarily because for their ease of recognition, their acceptance as regionally specific coral reef stress indicators (Risk *et al.,* 2001), and their diurnally sedentary nature. The latter of these requirements was necessary to ensure that individual organisms did not move out of the survey area between surveys and subsequently lead to inaccuracies and biases. However, for the purposes of on going studies and assessing the health of coral reefs, this is not a requirement. What is important is that surveyors are able to recognise the species in question and, that recognition is not dependent on extensive training or experience.

Likewise, this study did not aim to determine the suitability of these indicators for detecting change in coral reef environments. However, it was important to have a justification for indicator selection in order for participants to understand the change in marine communities and the reasons behind what they are being asked to do. The results from the SIMPER analysis showed that typically three of the five indicator species used were responsible for the variation in species counts both between transects (Table 4) and between experience groups (Table 3). Feather duster worms and long-spine sea urchins were shown to consistently affect results, with occasional differences also being attributed to counts of giant clams. Both feather duster worms and long-spine sea urchins were the most abundant in the study when considering the mean scientist data. It is possible that the variation in species counts across all participants could be as a result of the high abundance, leading to a natural difference in numbers because of the way each participant counts. However, in the case of the feather duster worms it is possible that the feeding habits of the species caused confusion. Feather duster worms are a boring species of worm, which embed themselves into the body of a coral, and exude feathery protrusions external to the coral in order to feed. The worm is able to retract these parts if threatened and will often do so in response to disturbance in water. While all participants were shown photographic examples of the worms, both with the feathery protrusions extended and retracted, in the latter case it is much more difficult to detect the presence of individuals. Therefore, it is possible that observers miss-counted the abundance of this species; there is also the risk that due to reduced comfort in the water, and associated splashing, that the feather duster worms retracted in the presence of surveyors, further complicating accurate presence/absence and abundance assessments. Similarly, while long-spine sea urchins are typically more active at night, there is a certain amount of variance in this behaviour between individual species (Tuya *et al.,* 2004). As such, there is the possibility that some individuals moved in and out of survey areas during the day, which could causes difficulties and biases in the accurate assessment of abundance.

Conducting more technical assessments is economically and logistically very complex in rural environments, particularly in developing countries such as Cambodia. This is due primarily to a lack of expertise and infrastructure to enable such analyses. Locally-based monitoring schemes can help bridge this gap, reinforcing existing community-based resource management programmes and fostering changes in the attitudes of local communities towards more sustainable environmental resource management (Danielsen *et al*., 2005).

The results of this study suggest that surveyor precision increases with experience marine surveys but assessing surveyor accuracy was outside its scope. As discussed previously, this could be conducted through the development of simulated survey areas mimicking lab-based studies. A similar approach could also be used to determine the potential impact of reef composition (with regards to the abundance and percentage cover of key substrate groups) on the accuracy and precision of surveyors, in addition to the impact of specific indicator species, and any life-history or behavioural traits which may encourage biases in abundance assessments.

Considerations for future monitoring programmes looking to involve community members or volunteers should consider the comfort and confidence of surveyors in the water. Using snorkelling equipment requires practice, particularly in areas with waves and currents. This can make conducting marine surveys difficult for those with limited experience. In some situations, it may also be necessary to provide suitable swimming training, to build comfort in the water especially if members of the community will be conducting surveys unsupervised. Therefore, it is important to assess the needs of the communities and individuals conducting the surveys prior to establishing monitoring programmes. Similarly, prospective participants should be trained and assessed on their species identification and survey skills, carrying a pass mark appropriate to the level of training and requirements of the monitoring programme. This could be difficult to enforce within some volunteer programmes, due to a lack of experience and knowledge of on-the-ground staff but training and assessment prior to data collection should be encouraged and supported.

The appropriate selection of indicator species is also a key consideration and may vary from site to site. This will depend on the needs of different management and monitoring programmes, the equipment available and the skills and knowledge of those collecting the data and conducting any necessary training. The results from this study highlight how important it is for indicator species to be easily recognisable, but to also consider the ecological traits of the species involved, as certain behaviours may lead them to be harder to identify, and therefore effect the relative accuracy of abundance assessments. Further investigation of the effects seen in the SIMPER analysis is essential to inform the correct assignment of indicators for research programmes and should incorporate the development of survey areas where the abundance of key indicators is known prior to surveys. While indicators were assigned to this programme based on their ease of identification and regional presence, care must be taken to adopt appropriate indicators to projects which are capable of informing both research and management questions, particularly when using less experienced surveyors (Soule, 1988) and including more indicator species. The selection of future indicator species should be made carefully, with consideration for the ease of identification, robust scientific knowledge on the species and geographic distribution (Hodgson, 1999). However, previous research has concluded that, in order to support long-term, locally based monitoring efforts, it is necessary to maintain simplicity and local relevance wherever possible (Danielsen *et al.,* 2005). As such, it would also be beneficial to integrate knowledge on the behaviour of each indicator species.

Furthermore, effort should be made to disseminate the results of findings to members of the local communities (Uychiaoco *et al.,* 2005). Involving community members in the collection of data could facilitate effective communication of findings, especially in areas of low literacy, and limited understanding of scientific principles. It may also help to foster effective conservation decision making through capacity building, increased environmental education, and trust between scientists, policy makers and local resource dependent communities (Mascia, 2001). It has also been suggested that community-based monitoring programs have the ability to empower local community members while also supporting conservation efforts (Constantino *et al.,* 2012).

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Table 1 Percentage cover of key substrate indicators collected to establish general state of each transect area.( Tr = Transect).

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | Tr. 1 | Tr. 2 | Tr.3  | Tr. 4 | Tr. 5 | T5. 6 |
| Live Coral | 5 | 37.5 | 37.5 | 25 | 0 | 40 |
| Dead Coral | 20 | 17.5 | 25 | 7.5 | 80 | 40 |
| Non-living Substrates | 67.5 | 45 | 37.5 | 60 | 15 | 17.5 |
| Algae | 7.5 | 0 | 0 | 7.5 | 0 | 0 |
| Other | 0 | 0 | 0 | 0 | 5 | 2.5 |

Table 2. PERMANOVA analysis identifying the differences in community composition assessment made by individuals in different experience groups (NS = non-signficant, df = degrees of freedom).

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Transect | Comparison | *t* | df | P*(perm)* |
| 1. | Khmer, Non-Exp Volunteer | 0.57 | 6 | NS |
|  | Khmer, Exp Volunteer | 2.42 | 6 | 0.03 |
|  | Khmer, Scientist | 0.62 | 6 | NS |
|  | Non-Exp Volunteer, Exp Volunteer | 2.42 | 6 | 0.03 |
|  | Non-Exp Volunteer, Scientist | 1.45 | 6 | NS |
|  | Exp. Volunteer, Scientist | 5.68 | 6 | 0.02 |
| 2. | Khmer, Non-Exp Volunteer | 1.46 | 6 | NS |
|  | Khmer, Exp Volunteer | 1.27 | 6 | NS |
|  | Khmer, Scientist | 1.40 | 6 | NS |
|  | Non-Exp Volunteer, Exp Volunteer | 1.02 | 6 | NS |
|  | Non-Exp Volunteer, Scientist | 3.11 | 6 | 0.04 |
|  | Exp. Volunteer, Scientist | 2.42 | 6 | 0.03 |
| 3. | Khmer, Non-Exp Volunteer | 0.81 | 6 | NS |
|  | Khmer, Exp Volunteer | 2.33 | 6 | 0.02 |
|  | Khmer, Scientist | 1.98 | 6 | 0.04 |
|  | Non-Exp Volunteer, Exp Volunteer | 2.00 | 6 | NS |
|  | Non-Exp Volunteer, Scientist | 1.46 | 6 | NS |
|  | Exp. Volunteer, Scientist | 1.66 | 6 | NS |
| 4. | Khmer, Non-Exp Volunteer | 1.77 | 6 | NS |
|  | Khmer, Exp Volunteer | 0.88 | 6 | NS |
|  | Khmer, Scientist | 2.07 | 6 | 0.03 |
|  | Non-Exp Volunteer, Exp Volunteer | 1.13 | 6 | NS |
|  | Non-Exp Volunteer, Scientist | 0.89 | 6 | NS |
|  | Exp. Volunteer, Scientist | 1.11 | 6 | NS |
| 5. | Khmer, Non-Exp Volunteer | 1.55 | 6 | NS |
|  | Khmer, Exp Volunteer | 1.42 | 6 | NS |
|  | Khmer, Scientist | 1.44 | 6 | NS |
|  | Non-Exp Volunteer, Exp Volunteer | 1.86 | 6 | 0.04 |
|  | Non-Exp Volunteer, Scientist | 1.56 | 6 | NS |
|  | Exp. Volunteer, Scientist | 2.50 | 6 | 0.03 |
| 6. | Khmer, Non-Exp Volunteer | 1.62 | 6 | NS |
|  | Khmer, Exp Volunteer | 1.70 | 6 | NS |
|  | Khmer, Scientist | 1.46 | 6 | NS |
|  | Non-Exp Volunteer, Exp Volunteer | 1.56 | 6 | NS |
|  | Non-Exp Volunteer, Scientist | 1.32 | 6 | NS |
|  | Exp. Volunteer, Scientist | 2.27 | 6 | 0.03 |

Table 3. Species identified in SIMPER analysis as being the main causes of change up to a threshold of 90% in counts of indicator species between transects. SIMPER analysis recorded similarity of 72.83%.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Transect | Species | Mean Abundance | Mean Similarity | Similarity St. Dev. | Percentage Contribution | Cumulative Percentage |
| 1. | Feather Duster Worm | 2.41 | 29.01 | 2.27 | 39.83 | 39.83 |
|  | Long-spine Sea Urchin | 2.34 | 22.45 | 1.14 | 30.83 | 70.66 |
|  | Giant Clam | 1.41 | 14.44 | 1.33 | 19.83 | 90.48 |
| 2. | Long-spine Sea Urchin | 6.11 | 38.95 | 3.91 | 48.01 | 48.01 |
|  | Feather Duster Worm | 3.81 | 28.16 | 3.47 | 26.94 | 74.95 |
|  | Giant Clam | 3.26 | 17.62 | 2.82 | 21.72 | 96.67 |
| 3. | Long-spine Sea Urchin | 5.58 | 37.40 | 4.87 | 50.78 | 50.78 |
|  | Feather Duster Worm | 4.49 | 23.33 | 2.34 | 31.68 | 82.46 |

Table 4. Species identified in SIMPER analysis as being the main causes of change up to a threshold of 90% in counts of indicator species between experience levels. SIMPER analysis recorded similarity of 55.80%.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  Group | Species | Mean Abundance | Mean Similarity | Similarity St. Dev. | Percentage Contribution | Cumulative Percentage |
| Khmer | Long-spine Sea Urchin | 3.93 | 32.66 | 1.97 | 58.54 | 58.54 |
|  | Feather Duster Worm | 2.48 | 15.58 | 1.33 | 27.91 | 86.45 |
|  | Giant Clams | 1.21 | 4.45 | 0.62 | 7.97 | 94.42 |
| Non Exp. Vol. | Long-spine Sea Urchin | 5.70 | 42.82 | 1.89 | 62.34 | 62.34 |
|  | Feather Duster Worms | 2.39 | 17.97 | 1.12 | 26.17 | 88.51 |
|  | Giant Clam | 1.38 | 7.23 | 0.89 | 10.52 | 99.03 |
| Exp. Vol. | Long-spine Sea Urchin | 5.44 | 34.80 | 1.65 | 44.31 | 44.31 |
|  | Feather Duster Worm | 3.49 | 21.71 | 1.95 | 27.65 | 71.96 |
|  | Sponge | 1.79 | 12.11 | 1.43 | 15.42 | 87.38 |
|  | Giant Clam | 1.80 | 9.33 | 0.94 | 11.89 | 99.27 |
| Scientist | Long-spine Sea Urchin | 5.47 | 43.88 | 2.33 | 50.19 | 50.19 |
|  | Feather Duster Worm | 3.66 | 30.40 | 2.59 | 34.77 | 84.96 |
|  | Giant Clam | 1.69 | 10.78 | 1.09 | 12.33 | 97.29 |