Comparison of local knowledge and researcher-led observations for wildlife exploitation assessment and management

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Summary

The use of local knowledge observations to generate empirical wildlife resource exploitation data in data-poor, capacity-limited settings is increasing. Yet, there are few studies quantitatively examining their relationship with those made by researchers or natural resource managers. We present a case study comparing intra-annual patterns in effort and mobulid ray catches, derived from local knowledge and fisheries landings data at identical spatio-temporal scales in Zanzibar (Tanzania). The Bland-Altman approach to method comparison was used to quantify agreement, bias and precision between methods. Observations from the local knowledge of fishers and those led by researchers showed significant evidence of agreement, demonstrating the potential for local knowledge to act as a proxy for, or complement, researcher-led methods in assessing intra-annual patterns of wildlife resource exploitation. However, there was evidence of bias and low precision between methods, undermining any assumptions of equivalency. Our results underline the importance of considering bias and precision between methods, as opposed to simply assessing agreement, as is commonplace in the literature. This case-study demonstrates the value of rigorous method-comparison in informing appropriate use of outputs from different knowledge sources, thus facilitating the sustainable management of wildlife resources and the livelihoods of those reliant upon them.

Introduction

Since the formation of modern natural resource management institutions, the majority of wildlife resource exploitation assessments have been derived either from observations or formal declarations, typically made by those specifically employed as researchers or natural resource managers (from here, 'researchers'). This has been the case for fisheries management, where such methods have been championed by fisheries science organisations, like the International Council for the Exploration of the Seas (ICES) formed in 1902. The types of methods used by ICES have been exported globally, being used as the model for other fisheries management bodies (Rozwadowski 2002). These now established methods for resource management generally rely on data-heavy sampling and complex statistics; a substantial barrier when time, financial capacity, or personnel expertise are limited.

If we were to go back roughly 100 years, such intensive methods were not common. Instead assessments we founded on the knowledge of those using natural resources, such as in Canadian (Murray et al. 2008) and Scottish (Thurstan and Roberts 2010) fisheries. Although local knowledge (LK), based on both the observations and experiences of those not directly employed as researchers (Stephenson et al. 2016), has attracted academic - and some bureaucratic - interest as an information source for resource management. To date, there is a lack of quantitative evaluations of the relationship between LK and researcher-led observations.

Since recording LK is generally considered a cheap but effective process (Neis et al. 1999; Anadón et al. 2009; Rist et al. 2010), the use of LK observations to assess various aspects of data-poor and capacity-limited fisheries is increasingly common (e.g. Moore et al. 2010; Pilcher et al. 2017). Such situations are perhaps most evident in the fisheries of low and middle income regions, making the use of LK in these particularly attractive. Additionally, LK observations may be advantageous in documenting unusual or illegal events, which researcher-led observations are liable to miss (Peterson and Stead 2011; Slater et al. 2014). Conversely, LK is vulnerable to interviewee subjectivity and bias, be it malicious or malign, for example through provision of misleading information or biases in cognitive recall. Yet, ignorance of LK has, in some cases, resulted in fisheries mismanagement (Johannes et al. 2000).

Despite uncertainties in both LK and researcher-led observations there are few studies that cross-examine their outputs. The majority have been restricted to evidencing agreement (e.g. Anadón et al 2009; Rist et al. 2010; Daw et al. 2011) and fail to assess bias and precision among methods. Evidence for agreement between LK and researcher-led observations is mixed (Anadón et al. 2009; Rist et al. 2010; O'Donnell et al. 2012), although LK is generally considered a useful indicator of long-term trends (Stead et al. 2006; Daw et al. 2011; O'Donnell et al. 2012). The use of LK to assess shorter temporal ranges, such as intra-annual trends, has received relatively limited attention since a number of earlier publications outlined how knowledge accumulated in real-time, over the shortest timescales, may be amongst the most unique knowledge possessed by fishers (Fischer 2000; Knapman 2005; Hind 2012). Yet, intra-annual trends are often important in the formulation of management strategies.

The aim of this study is to assess the capability of LK observations to provide data for improved sustainable resource management in data-poor and capacity limited settings. Further, the case-study presented, which assesses intra-annual patterns in small-scale fisheries effort and catch is, to our knowledge, the first of its kind. Thus, it also facilitates an initial assessment of the potential use of LK observations as a proxy for researcher-led observations in data-poor and capacity-limited situations at intra-annual timescales.

Methods

Trained observers from the then Ministry of Livestock and Fisheries (now Ministry of Agriculture, Natural Resources, Livestock and Fisheries) collected researcher-led observations of fisheries effort (active vessels per day) and landed catch (individuals per day) of mobulid rays, *Mobula sp*. (n=161), from bottom-set and drift gillnets, longlines, and handlines at small-scale fisheries landings sites in Zanzibar (n=8) (Fig. 1); 147 simultaneous days were observed over a complete 12-month period between June 2016 and 2017. In order to account for lunar-driven patterns in fishing effort and species availability, monitored days were selected using a stratified-random approach; the year was divided into lunar months which were subdivided into four lunar phases (new moon, first quarter, full moon, third quarter) and three sampling days randomly generated within each lunar phase. Landing sites were selected to account for the following criteria: the prevalence of longline and gillnet gears (the primary gear threats to rays); geographic spread (maximising geographic coverage and potential links to species availability); and logistical constraints (e.g. sites needed to be accessible by road) (Temple et al. 2019). Resultant data were linearly scaled to monthly totals.

LK observation data were collected using a modified Rapid Bycatch Assessment (RBA) interview (e.g. Moor et al. 2010; Alfaro-Shigueto et al. 2018) in September 2017. The RBAs targeted fishing vessel captains in the same small-scale fisheries landing sites, covering the same gears and temporal period (n=204, captains=99). The RBAs recorded declarations of average days fished per month (on an annual level), months in which fishing occurred, average mobulid catch per month (on an annual level), and months in which catches occurred. A minimum of three, or a quarter of the known vessels, whichever was largest, RBAs were conducted for each gear type at each site in order to achieve a representative sample. RBAs were carried out in Swahili by co-author Jiddawi, who is a native speaker. Interviewees were selected opportunistically, avoiding multiple crew members from the same vessel. The RBAs lasted approximately 20 minutes. Interviewees were informed of both the motivation and the intended use of the data collected, anonymity, the right to decline answering any question and the right to end the interview at any stage. Verbal consent was sought before the RBA was undertaken. The RBAs were not facilitated with either monetary or material motivation.

Statistical Analysis

All analyses were carried out using the R statistical software package v3.6.0 (R Core Team 2019). We used the Bland-Altman approach (Bland & Altman 1999; Bland & Altman 2003) to compare intra-annual patterns (measured as a proportion of annual total) of fisheries effort and catch observations. Agreement was assessed using binomial generalised linear mixed models (GLMMs) with site treated as a random effect for both slope and intercept (R package *lme4*). Subsequently, bias was assessed by modelling the relationship between the means of methods and the difference between methods using linear mixed effect models (LMEs) with site treated as a random effect for both slope and intercept (R package *lme4*).

The precision of methods relative to one another was described by the exact limits of agreement (LOA), equivalent to the 95% mean confidence interval of the differences between methods (Carkeet & Goh 2018). Both GLMM and LME models were weighted using the RBA sample size, reflecting increased confidence in data derived with higher sample sizes.

Results

The GLMM for intra-annual patterns in fishing effort showed a significant, but relatively weak, relationship between LK and researcher-led observations (Z=2.04, p=0.042, r^2c=0.006) (Fig. 2a) and found no evidence for any interacting effect of gear type on the relationship between methods (ANOVA, χ^2=0.801, p=0.992). As there was sufficient evidence of a positive relationship between method outputs for fisheries effort, assessments of bias and precision were undertaken. LMEs demonstrated a significant deviance from the null model (ANOVA, χ^2=37.181, p<0.001), indicating a significant bias between method outputs, and found no significant interacting effect of gear type on the bias between methods (ANOVA, χ^2=6.12, p=0.410). The RBA surveys produced higher fishing effort estimates than observer data at low mean effort and the inverse at high mean effort (Fig. 2b). LOAs, once bias was accounted for, were estimated at ±3.67% (95%CI 3.37-4.03%) of annual effort in any given month (Fig. 2b).

The GLMM for intra-annual patterns in fisheries catches showed a significant, but relatively weak, relationship between methods (Z=3.49, p<0.001, r^2c=0.101) (Fig. 2c). As there was sufficient evidence of a positive relationship between methods for fisheries catches, assessment of bias and precision was undertaken. LMEs demonstrated a significant deviance from the null model (ANOVA, χ^2=15.5, p<0.001). The results indicate the presence of significant bias between methods for mobulid ray catch, with RBA surveys producing higher catch estimates than observer data at low mean catches and the inverse at high mean catches (Fig. 2d). LOAs, once bias was accounted for, were estimated at ±22.4% (95%CI 19.3-27.0%) of annual mobulid catch in any given month (Fig. 2d).

Discussion

We found a positive relationship between LK and researcher-led observations of intra-annual patterns in fisheries effort and catches. This suggests that both approaches may act as a proxy for, or complement, one another when assessing such harvest effort and wildlife resource exploitation data. This outcome provides support for the expanded use of LK as an assessment tool with which to support the sustainable management of wildlife resource exploitation, particularly in data-poor and capacity limited situations. Indeed, by demonstrating a real-world application, it strengthens representations already being made in the specific context of fisheries management for greater integration of fishers' local knowledge (often termed 'fishers' knowledge') into scientific assessments (Soto 2006; Hind 2012; Hind 2015; Stephenson et al. 2016). However, the analyses also highlight the importance of considering bias and precision between LK and researcher-led observations, in order to facilitate informed interpretation of their outputs. The significant bias and low level of precision between LK and researcher-led observations evidenced in this study, undermines any baseline assumptions of equivalency, in spite of the general evidence for method agreement. Understanding and accounting for factors that drive inequivalences (which may be both generalised and/or case specific) between LK and researcher-led observations is an important step in supporting the decision making for sustainable wildlife resource exploitation.

Equivalency between LK and researcher-led observations is a particularly important consideration here because natural resource management is an activity where it is readily identified that epistemic communities have formed around shared and coordinated knowledge bases, which they have then brokered. As communities are empowered through governing institutions prioritising their knowledge in the policy making process, they essentially determine which knowledge is used in management (Hass 1989). Epistemic communities have typically been dominated by researchers, because firstly, their approaches have typically aligned with governing agendas of doing what is perceived as good by citizens, and secondly, it has suited governments to refer to a single group as this creates economies-of-scale and results in quicker arrival at consensus (Weale 1992). Natural resource management has been little different. Knowledge of those beyond epistemic communities remains what might be considered 'subjugated' (Foucault & Ewald 2003), integrated only at the discretion of the research community, as is the case for fisheries management (Jentoft 2005). Gaining perceived equivalence of utility in the eyes of researchers, or at least reaching such levels, is the most likely path to LK actually being used in management (Soto 2006; Hind 2012).

Perhaps the most important factor to consider, then, is simply - are LK and researcher-led observations measuring the same thing? Such disparities have been seen in studies compiling knowledge from various sources (e.g. Jennings & Polunin 1995; Daw et al. 2011), where differences in selectivity and spatio-temporal coverage undermine equivalency. The same spatio-temporal disparities have even been promoted as a chance to manage at scales seen as desirable, but at which it has not yet been possible based solely on data derived from researcher-led observation (Griffin 2009; Hind 2012). With regard to the present study, there are a number of factors potentially contributing to a lack of equivalency between LK and researcher-led observations. Discards, loss of catch at sea, and secreted landings inevitably create underestimate in fisheries landings observation data but could feature in LK observations. Underestimates are potentially most prevalent for those catches most difficult or dangerous to bring aboard, especially in gears that are not suited to their capture, and for illegal or heavily regulated catches, which may be discarded or hidden for fear of prosecution. Further, fishers often land catches at sites other than their home port, depending on local market conditions and demand for specific catches (Temple unpub. data.). This may result in site-specific under- and over-representation of some catches from LK. Lastly, the migratory nature of some fisheries in this (Wanyonyi et al. 2016) and other regions means fishers may be active in other fishing grounds when activity from their home port is low. Greater consideration for, and disaggregation of, these and similar potential factors may help improve the equivalency of LK and researcher-led observations and/or improve the informed interpretation of their outputs relative to one another.

The efficacy of both LK and researcher-led observations in representing reality is another important consideration. For example, it is probable that the efficacy of researcher-led observations will vary with the overall level of observer competence (e.g. level of training provided), individual observer competence, and the nature of the landing sites themselves (e.g. size, layout, and level of formal organisation). Similarly, researcher-led observation efficacy likely varies among components of the catch. For example, smaller specimens are perhaps less likely to be observed if they are mixed with bulk landings of similarly sized catch, and rare or infrequent catches may become underrepresented with only a small number of missed observations. Conversely, the efficacy of LK observations may be affected by survey design and biases in human memory recall. For example, the RBA questionnaire used in the present study derives catch and effort data from average monthly levels, alongside months of occurrence, an approach that likely supresses the magnitude of monthly variability. Human recall is generally improved for events that are particularly unusual or emotive (e.g. unusually poor fishing conditions, catches of unusual size, volume, value or rarity) and/or that display prominent and consistent temporal trends (Matlin 2004; Hirst et al. 2009). Such events may be more easily recalled by fishers and may therefore be over-represented relative to other less memorable events. As a result, LK observations of fisheries effort and catches may be partially obscured at the fishery-level. High variability among fisher declarations, which was evident here, may also partially obscure catch and effort patterns at the fishery level (O'Donnell et al. 2012). Mobulid rays display traits that could potentially increase their memorability (e.g. unusual body form, large size, high value, distinct seasonality, and relative rarity) and this might be expected to increase the reliability of LK observations, if it were the case. Agreement between LK and researcher-led observations for species which are not memorable to fishers might be expected to result in lower agreement among methods, a potential effect that should to be considered in future sampling methodologies.

The current use and continued iterative refinement of both LK and researcher-led observation methods is an ongoing challenge for researchers and managers of natural wildlife resource exploitation. Yet method comparison studies are uncommon and they rarely consider bias and precision (e.g. Anadón et al. 2009; Rist et al. 2010; Daw et al. 2011). We believe that the concurrent use and thorough cross-examination of outputs from these methodologies will be valuable to future methodological developments and current usage of method outputs, and support moves to integrate LK into mainstream research and management of natural resources (Stephenson et al. 2016). Assessment of agreement, the identification of bias, and quantification of precision allow for a greater understanding of the variable structure of the relationship among methods. Thus, comparative studies can better facilitate the identification of method shortcomings or disparities and thus improve method refinement and contextualisation. Most importantly, comparative studies stand to inform the appropriate use of LK, established, and novel method outputs. This is a vital step in ensuring the appropriate application of method outputs to the sustainable management of wildlife resources and the livelihoods and wellbeing of those dependent upon them. The findings herein contribute to the wider discourse on how LK can help countries improve progress towards achieving United Nations Sustainable Development Goals targets.

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Conflict of Interest

None

Ethical Standards

All data used in this study were collected in line with national and institutional laws and requirements. Ethical approval for the study was sought, and granted, from both Newcastle University, UK and the University of Dar es Salaam, United Republic of Tanzania as appropriate. RBA interviewees were informed of both the motivation and the intended use of the data collected, anonymity of their responses, the right to decline answering any question and the right to end the interview at any stage were assured. Verbal consent was sought before the RBA was undertaken. The RBAs were not facilitated with either monetary or material motivation.

Literature Cited

Alfaro-Shigueto J, Mangel JC, Darquea J, Donoso M, Baquero A, Doherty PD, Godley BJ (2018) Untangling the impacts of nets in the southeastern Pacific: Rapid assessment of marine turtle bycatch to set conservation priorities in small-scale fisheries. *Fisheries Research* 206: 185-192.

Anadón JD, Giménez A, Ballestar R, Pérez I (2009) Evaluation of Local Ecological Knowledge as a Method for Collecting Extensive Data on Animal Abundance. *Conservation Biology* 23: 617-625.

Bland JM, Altman DG (1999) Measuring agreement in method comparison studies. *Statatistical Methods in Medical Research* 8: 135-160.

Bland JM, Altman DG (2003) Applying the right statistics: analyses of measurement studies. *Ultrasound in Obstetrics & Gynecology* 22: 85-93.

Carkeet A, Goh YT (2018) Confidence and coverage for Bland–Altman limits of agreement and their approximate confidence intervals. *Statistical Methods in Medical Research* 27: 1559-1574.

Daw TM, Robinson JAN, Graham NAJ (2011) Perceptions of trends in Seychelles artisanal trap fisheries: comparing catch monitoring, underwater visual census and fishers' knowledge. *Environmental Conservation* 38: 75-88.

Fischer J (2000) Participatory research in ecological fieldwork: a Nicaraguan study. *Finding our sea legs: linking fishery people and their knowledge with science and management* 41-54.

Foucault M, Ewald F (2003) *" Society Must Be Defended": Lectures at the Collège de France, 1975-1976*. Macmillan.

Griffin L (2009) Scales of knowledge: North Sea fisheries governance, the local fisherman and the European scientist. *Environmental Politics* 18: 557-575.

Haas PM (1989) Do regimes matter? Epistemic communities and Mediterranean pollution control. *International organization* 43: 377-403.

Hind EJ (2012) Last of the hunters or the next scientists? Arguments for and against the inclusion of fishers and their knowledge in mainstream fisheries management.

Hind EJ (2015) A review of the past, the present, and the future of fishers' knowledge research: a challenge to established fisheries science. *Ices Journal of Marine Science* 72: 341-358.

Hirst W, Phelps EA, Buckner RL, Budson AE, Cuc A, Gabrieli JDE, Johnson MK, et al. (2009) Long-term memory for the terrorist attack of September 11: Flashbulb memories, event memories, and the factors that influence their retention. *Journal of Experimental Psychology: General* 138: 161.

Jennings S, Polunin NVC (1995) Biased underwater visual census biomass estimates for target-species in tropical reef fisheries. *Journal of Fish Biology* 47: 733-736.

Jentoft S (2005) Fisheries co-management as empowerment. *Marine Policy* 29: 1-7.

Johannes RE, Freeman MMR, Hamilton RJ (2000) Ignore fishers' knowledge and miss the boat. *Fish and Fisheries* 1: 257-271.

Knapman P (2005) Participatory Governance in Inshore Fisheries Co-Management in England and Wales. In: *Participation in Fisheries Governance*, ed. TS Gray, pp. 163-178. Dordrecht: Springer Netherlands.

Matlin MW (2004) *Cognition*. Hoboken, NY, USA: Wiley.

Moore JE, Cox TM, Lewison RL, Read AJ, Bjorkland R, McDonald SL, Crowder LB, et al. (2010) An interview-based approach to assess marine mammal and sea turtle captures in artisanal fisheries. *Biological Conservation* 143: 795-805.

Murray GD, Neis B, Palmer CT, Schneider D (2008) Mapping cod: fisheries science, fish harvesters' ecological knowledge and cod migrations in the Northern Gulf of St. Lawrence. *Human Ecology* 36: 581-98.Neis B, Schneider DC, Felt L, Haedrich RL, Fischer J, Hutchings JA (1999) Fisheries assessment: what can be learned from interviewing resource users? *Canadian Journal of Fisheries and Aquatic Sciences* 56: 1949-1963.

O'Donnell KP, Molloy PP, Vincent ACJ (2012) Comparing Fisher Interviews, Logbooks, and Catch Landings Estimates of Extraction Rates in a Small-Scale Fishery. *Coastal Management* 40: 594-611.

Peterson AM, Stead SM (2011) Rule breaking and livelihood options in marine protected areas. *Environmental Conservation* 38: 342-352.Pilcher NJ, Adulyanukosol K, Das H, Davis P, Hines E, Kwan D, Marsh H, et al. (2017) A low-cost solution for documenting distribution and abundance of endangered marine fauna and impacts from fisheries. *PLoS ONE* 12: e0190021.

R Core Team (2019) *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria.

Rist J, Milner-Gulland EJ, Cowlishaw GUY, Rowcliffe M (2010) Hunter reporting of catch per unit effort as a monitoring tool in a bushmeat-harvesting system. *Conservation Biology* 24: 489-499.

Rozwadowski HM (2002) *The sea knows no boundaries: a century of marine science under ICES*. University of Washington Press and University of Columbia Press.

Slater MJ, Napigkit FA, SM Stead (2013) Resource perception, livelihood choices and fishery exit in a Coastal Resource Management area. *Ocean & Coastal Management* 71: 326-333.

Stead S, Daw T, Gray T (2006). Uses of fishers' knowledge in fisheries management. *Anthropology in Action* 13: 77-86

Soto CG (2006) Socio-cultural barriers to applying fishers' knowledge in fisheries management: An evaluation of literature cases. School of Resource and Environmental Management-Simon Fraser University.

Stephenson RL, Paul S, Pastoors MA, Kraan M, Holm P, Wiber M, Mackinson S, et al. (2016) Integrating fishers' knowledge research in science and management. *Ices Journal of Marine Science* 73: 1459-1465.

Temple AJ, Wambiji N, Poonian CNS, Jiddawi N, Stead SM, Kiszka JJ, Berggren P (2019) Marine megafauna catch in southwestern Indian Ocean small-scale fisheries from landings data. *Biological Conservation* 230: 113-121.

Thurstan RH, Roberts CM (2010) Ecological meltdown in the Firth of Clyde, Scotland: two centuries of change in a coastal marine ecosystem. *PLoS ONE* 5: e11767.

Wanyonyi IN, Wamukota A, Mesaki S, Guissamulo AT, Ochiewo J (2016) Artisanal fisher migration patterns in coastal East Africa. *Ocean & Coastal Management* 119: 93-108.

Weale A (1992) *The new politics of pollution*. Manchester University Press.

Figure Legend

Fig. 1. Locations of landing sites in Zanzibar where both local knowledge and researcher-led observations were recorded for fishing effort and mobulid catch between June 2016 and June 2017.

Fig. 2. Relationships between estimates of fishing effort and mobulid catch derived from local knowledge (LK) and researcher-led observations: a) regression line derived from binomial generalised linear mixed model for fisheries effort, b) Bland-Altman plot showing significant bias between observations and the limits of agreement between observations for fisheries effort, c) regression line derived from binomial generalised linear mixed model for mobulid catch, d) Bland-Altman plot showing significant bias between observations and the limits of agreement between observations for mobulid catch.