Ocean & Coastal Management 121 (2016) 60–69

Contents lists available at ScienceDirect

Ocean & Coastal Management

journal homepage: www.elsevier.com/locate/ocecoaman

An integrated evaluation of potential management processes on marine reserves in continental Ecuador based on a Bayesian belief network model

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ARTICLE INFO

Article history:
Received 14 August 2015
Received in revised form
19 December 2015
Accepted 19 December 2015
Available online 30 December 2015

Keywords:
Artisanal fishing
Shark finning
Ecuador
Marine protected areas
Bayesian belief network

ABSTRACT

Evaluating potential effects of conservation and management actions in marine reserves requires an understanding not only of the biological processes in the reserve, and between the reserve and the surrounding ocean, but also of the effects of the wildlife on the wider political and economic processes. Such evaluations are made considerably more difficult in the absence of good ecological data from within reserves or consistent data between reserves and the wider marine environment, as is the case in much of mainland Ecuador. We present an approach to evaluate the effects of a wide range of possible management processes on the marine ecology of the Machalilla National Park, as well as that of the surrounding marine environments (including recently established reserves) and related socio-economic pressures. The approach is based on Bayesian belief networks, and as such can be used in the presence of sparse data from multiple and disparate sources. We show that currently there are no observable benefits of marine reserves to reef and fish community structure, and that high value (normally predatory) fish, which are sought by fishers and shark finners are frequently absent from reef systems. We demonstrate that there is broad similarity in ecological communities between most shallow marine systems, in or out of marine reserves, and predict there can be a strong effect from actions outside the reserve on what is present within it. We also show that establishing a stronger link between (responsible) ecotourism and the marine environment could reduce the need for income in other more destructive areas, such as fishing and particularly shark finning, and discuss ways that high value, low impact eco-tourism could be introduced.

1. Introduction

Ecological management advice is normally provided based on ample data from the systems being studied. For example, total allowable catches for fisheries are based on the number of previous years' catches and estimates of recruitment (Lassen and Medley, 2001). However, while advice is normally provided in this context of ample data, decisions themselves can frequently seem uninformed by the science, and bear little relationship to the initial advice given (Daw and Gray, 2005). Part of this problem comes from the multiple demands placed on policy makers, legislators and politicians beyond the scientific predictions of the biological population or community; where political, economic and other priorities must be incorporated in the decision making (Daw and Gray, 2005; Beddington et al., 2007).

Recently, concepts such as ecosystem services and other socio-economic indicators have become part of any applied ecologist's

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http://dx.doi.org/10.1016/j.ocecoaman.2015.12.010
0964-5691/

vocabulary, yet the links between biological community structure and the sociological, political and economic processes are much more poorly understood than the links between components of the biological communities (Raffaelli and Frid, 2010; Silvertown, 2015).

Predictive models, such as those used in fisheries sciences, are data intensive, and can only be optimally parametrised by specialist scientists, often by mutual discussion and agreement in intensive working groups (Hilborn and Walters, 2013). Many stakeholders distrust the lack of transparency of the models and the predictions they produce (Jentoft, 2000), and legislators, bureaucrats and politicians are equally unfamiliar with the science. Claims of stakeholders such as fishers are combined with those of scientists in establishing final quotas and other protective measures (Beddington et al., 2007), and due to poor understanding of the scientific processes, this can lead to unsustainability of quota numbers.

The complexities outlined above, including the common use of data intensive models and the need to link community ecology to ecosystem services and socio-economics, would appear to indicate that management of marine communities for which there were few data would be virtually impossible. However, simple models can often provide sufficient information to meet many policy goals, and may require fewer data to parameterise (Stafford et al., 2015).

The marine ecosystem of mainland Ecuador is relatively unstudied, despite its high diversity and high abundance of charismatic megafauna such as manta rays, whale sharks and humpback whales which are attracted to the coast each year (Gabor, 2002). There are a large number of marine reserves and national parks serving as protected areas, although it is known that enforcement of restrictions in the parks are often poor (Gravez et al., 2013). However, some parks which conform more to the UN governance standards for MPAs are appearing to show greater benefits (Gravez et al., 2013). The exact nature of restrictions within parks can also be confusing, with unclear guidelines on what activities are legal and which are restricted or prohibited, or what levels of fishing are permitted, although fishing is largely restricted to artisanal fisheries, rather than larger industrial vessels within reserves (INEFAN, 1998).

International studies, as indexed in the Web of Knowledge database, of the coastal marine ecosystems of continental Ecuador are sparse (for example, 'Machalilla National Park' returns only six studies related to the marine environment, mostly on humpback whales). Literature on marine reserves has been collated (Hurtado et al., 2010), but there is no standard form of data collection or presentation from the reserves, making comparisons difficult between areas.

Recent reports have demonstrated illegal and unsustainable fishing practices; such as shark finning, have been occurring throughout the mainland Ecuador coast (for example 200,000 fins were seized in the port of Manta in May 2015). However, the country's tourism industry also promotes the biodiversity of the country, although much of the focus of marine biodiversity is placed on the Galapagos Islands (e.g. Halpenny, 2003). This is despite the mainland having many large species of megafauna, especially in the May to October period.

This study uses observational data, based on SCUBA dives with commercial operators and additional snorkelling surveys, as well as existing data to parameterise a modified Bayesian belief network (as presented in Stafford et al., 2015). This allows for rapid and simple surveys, compared to more structured systematic survey methods, but still collects useable data. The network integrates community interactions within the Machalilla national park at a broad scale, but also considers the interaction of the reserve with the wider network of nearshore or shallow marine habitats in Ecuador and beyond. It also integrates biological community dynamics with socio-economic concerns, such as tourism and fishing. This allows an integrated management strategy to be formulated for the region, which can exploit economic advantage while limiting damage to biodiversity, especially within the marine parks. Given the simplicity and transparency of the user interface of the model (Stafford and Williams, 2014), we envisage that such models could become useful management tools in a large number of coastal ecosystems worldwide.

2. Methods

The methods first present the concept of the Bayesian belief network approach for model construction, and an overview of the model. This provides context for the data collection and analysis sections. This methods section then describes how the data collected were transformed into parameters for the model.

2.1. Bayesian belief network model overview

A Bayesian belief network model (BBN) was used as the basis of the predictions in this study. The BBN is modified from traditional BBNs as detailed in Stafford et al. (2015). BBNs consist of a series of connected nodes, which have a probability of existing in a number of fixed states. For example, a node could represent the population size of a species, and it could be in two fixed states: *Increasing* or *Decreasing*. The probabilities of both states would sum to 1. Prior probabilities of each state of each node can be defined, for example, if evidence suggested a species was likely to decrease (i.e. a fishery for that species was commencing) then it would be possible to set the prior values accordingly.

Nodes are interconnected by edges. Each edge indicates a certainty and direction that one node may affect another. For example, if species A was connected to species B then it could be specified that; If species A was increasing (with a probability of 1), then it is 80% certain that species B will decrease (probability of 0.8). As absolute certainty (probability of 1) is unlikely, the network uses Bayesian inference to calculate the probability of species B decreasing, given the calculated probability of species A increasing.

Modifications to BBNs as detailed in Stafford et al. (2015) allow functionality important to ecosystem dynamics to be incorporated, including: 1) intuitive reciprocal interactions to be included in the network (i.e. as required by interspecific competition or both bottom up and top down tropic interactions). 2) Reduced use of prior knowledge. This means only targeted species or groups need to have priors assigned. Non-targeted species, which may be indirectly affected by a change in management practice do not need priors assigned (or more accurately, priors can remain 0.5 for both increasing and decreasing). This avoids 'double accounting' presented in some BBNs, as the belief in what will happen to non-targeted species or nodes will already be incorporated in the probabilities of the network 'edges'. 3) Interactions are considered individually rather than collectively. For example, if both Species A and Species B predate on Species C, the model would only require estimates of Species A on species C and species B on species C, rather than the combined effect of predation. This allows for easier parameterisation of the network from existing data, or less subjectivity if parameters are informed by expert opinion. 4) The BBN is presented in a simple user interface, using Microsoft Excel. Tests have shown that students entering university education are able to build and parameterise these networks using this interface with around 30 min training (Stafford and Williams, 2014). Hence they have wide potential to be understandable and transparent to multiple stakeholders.

The structure of the BBN used in this study is shown in Table 1. In this study, we used broad scale functional groups of species, rather