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

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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.

Key words: Artisanal fishing, shark finning, Ecuador, Marine Protected Areas, Bayesian belief network

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 Mendley, 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 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 (Jentoff, 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 minutes 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 than individual species themselves, for many nodes in the network (Table 2). This allowed for reduced data requirements, but still provided sufficient resolution to give an indication of the importance to various ecosystem services. These functional groupings of species were considered within the Machililla national park. We also considered different habitat types outside of the reserve which may act as reservoirs or nursery grounds for different species, and subsequently could affect populations in Machililla. By considering community similarity between these different types of habitat (mangroves, sandy beaches, rocky coastline < 5m deep, and reefs > 10 m but not in Machililla national park, although some reefs were in newly established marine reserves) we parametrised the network to demonstrate how protected areas would affect and would be affected by changes to these habitats. Finally we included nodes that demonstrated human activity and/or ecosystem services that would affect coastal regions, or be affected by changes in the region. The exact definition of what the probability of each node represents is provided in detail in Table 2.

As such, the scope of the model is to predict how a number of different management options could affect the diversity and functioning of the Machalilla marine reserve as well as coastal systems not in reserves. The outputs are expressed both in terms of population trends of major functional groups of marine organisms, but also in terms of predictions to economic and ecosystem services and ultimately the economic role of coastal marine systems.

2.2 Data collection

For field studies, sites were selected largely based on accessibility from land, or from SCUBA charter or tourist boats. Locations of study sites, including their classifications in the models are provided in Figure 1. Sites around the town Santa Elena were situated just outside the Puntilla de Santa Elena marine reserve and due to accessibility to sites, several surveys were conducted in the recently established El Pelado reserve. Because El Pelado was also a marine reserve (although recently established), we conducted analyses on the data that compared Machililla to these other marine reserve sites and sites of similar depth and benthic structure, but not in marine reserves.

Species composition and relative abundance were collected during snorkelling or SCUBA dives. Data collection was observational, rather than following strict transects or timed counts. In this case, observational data collection was sufficient as detailed quantification of population sizes was not required, given the use of functional groups in the model rather than individual species. However, data were collected to a minimum of family level, and mainly to species level, using photographs to identify species not immediately recognisable underwater. The DAFOR scale was used to indicate abundance, and allowed for rapid data collection, but with an ability to compare community structure between different locations and habitat types. DAFOR was applied separately to fish in different feeding guilds, hence common piscivores would contribute equally to our measure of community structure as common planktivores, despite the number and biomass of piscivores being lower. Such an approach therefore allows a more sensitive approach to examining differences in communities in different habitats, especially if some functional groups differ between habitats.

To assess similarity and hence possible connectedness (as nursery grounds for adult of planktonic movement between areas) between environments, species lists for within the reserves, from other reefs not in reserves (but of similar depth and substratum type), from rocky coasts and from sandy coasts were compiled along with DAFOR scoring for each habitat type. The DAFOR classification was then converted to numbers (10, 6, 4, 2, 1 respectively) and the sum of scores for species was added for each of the habitat types, to assess the most important species in the communities. Those species with a combined score ≥ 12 (i.e. at least occasionally sighted in several of the habitats studied) were then converted into percentage of the community in each habitat type (following similar procedures to that in Stafford et al., 2014) and these percentages used to assess similarities between communities using bootstrapped PCA analysis (Stafford et al., 2012).

While observational, *ad-hoc*, data collection methods may not provide as robust results as systematic surveys, their use in this study is justified by both ecological sampling theory and recent research. Firstly, this study aims to provide simple management advice for marine environments where data are limited, and the Bayesian belief network models are developed with limited data in mind, emphasising uncertainty in the approach – hence systematic surveys may not be available for many regions where this technique could be applied. The comparisons between habitats (e.g. marine reserves to non-reserves) conducted in this study only use the most abundant species in each habitat to draw values of connectivity between habitats for use in the models, which in turn only model functional group, rather than species responses. In terms of ecological sampling theory, abundant species are recorded rapidly with low sampling effort (Southward, 1978), and high numbers of replicates are normally only required to detect the rarest species. As such, observational surveys, placing species abundance into an ordinal scale (DAFOR) are unlikely to bias collection of common species, but may be subjective to greater stochasticity in recording rare species (depending on whether a species was encountered or not). Indeed, results from citizen science surveys indicate that common species can be accurately recorded using observational and *ad-hoc* collected data (Silvertown 2009; Stafford et al., 2010), and that *ad-hoc* sightings data can be as good as more rigorously collected 'scientific' data in addressing many ecological questions (Higby et al., 2012; Stafford et al., 2013). As such, observational data were used in the current study to rapidly achieve the desired results.

Fish markets where artisanal fishery catches were sold were visited to identify the commonly landed species. Artisanal fisheries are the only fisheries allowed within 8 miles of the coast, hence these catches represent what is directly taken from the coastal areas.

2.3 Determining parameters of models

BBNs are designed to incorporate belief about systems. As such some degree of subjectivity in parameterisation of the network is unavoidable. In some ways, this can be seen as a strength of the approach, as it allows stakeholders to modify the network dependent with their beliefs and/or motives. However, to help eliminate subjectivity in the initial model we present in this paper, during initial parameterisation all sets of parameters used in the model were discussed and agreed between a group of at least three authors of the study. In addition, as many parameters as possible were informed by data. For example, communities which did not differ significantly from each other at the 95% confidence level were considered to have a higher connection (edge value) for most groups of species. Those habitats that differed more generally had lower confidence edge values, although in some cases, some taxonomic groups were similar between habitats, despite the overall community varying. Mangroves could not be surveyed in this study due to inaccessibility and lack of visibility in the water, and alternative strategies for sampling would provide very different results from visual surveys. Previous studies from the Caribbean have shown strong connectivity between mangrove fish and coral reef communities, where the distance between mangroves and reefs is relatively small (i.e. several miles; Mumby et al., 2004). However, such studies have not been conducted in Ecuador and distances between the largest mangroves in the south of the country and the coral reefs of the reserve were much larger than in these studies (hundreds of miles), although other closer and smaller mangrove systems did exist. Hence a connectivity of between 0.6 and 0.7 were used for different functional groups (with higher values of 0.7 indicating connections with typical reef fish, which were examined in the Caribbean study). Such figures encapsulate the best evidence available (with values of 0.6 indicating high levels of uncertainty), and hence support the concept of using BBNs in data poor environments, as they can combine evidence from multiple and disparate sources.

The parameters used in the current study are provided in Table 1. The working model is provided as supplementary material to this paper as a macro enabled Microsoft Excel file. Changes to prior values from those presented in Table 1 and the supplementary material are clearly indicated when presenting case studies of the simulations. Where different parameters to those used in the supplementary material were used for particular sections of the results, these changes are clearly indicated.

3. Results

3.1 Overview of results

Marine communities > 10 m in depth, regardless of their position in long established or recently established marine reserves or outside any reserve and did not demonstrate significant differences in community composition (Figure 2). Overall the benthic structure of hard reefs comprised of rock or boulders covered in soft corals and algae. Hard, stony corals are mostly slow growing boulder corals such as those found within the families Poritidae, Faviidae and Agaracidae. These hard corals were mainly in the shallower areas (~10 - 15m). Other sessile invertebrates were a variety of sponges, hydroids and bryozoans. Fish communities were dominated by invertivores, although very large numbers of planktivores such as chromis could also regularly be seen in many locations. There was a noticeable distinct lack of top predators, including sharks but also larger teleost fish such as jacks and tuna.

Communities on shallower reefs (coastal subtidal rock < 5 m) were broadly similar, to those of deeper reefs, with a key difference being the absence of the abundant soft corals in shallower water, which were clearly present in the deeper water sites. Sponges were common at several sites with high suspended sediment levels. Fish communities were also similar, but with an absence of the bigger reef fish such as parrot fish, resulting in clear differences in community structure at the 95% confidence interval level, but not being significantly different from these reef habitats at the 99 % confidence level (Figure 2).

Sand bottomed communities were significantly different to rock communities (Figure 2). Small or juvenile rays were frequent, as were shoals of juvenile pelagic species. However, some common similarity occurred between species present on sandy communities and on deeper reef systems (i.e. porcupine fish, chromis) being common at all sites (Table 3). Birds, including pelicans, boobies and tropical birds were common over sand communities.

While it is expected that community differences would occur between many of these habitats, common species can give an indication of connectivity of habitats, and these data were used in the parameterisation of the BBN (see methods).

Many small fish markets were mainly supplied by the artisanal fishing industry common across Ecuador. Typically pelagic and predatory species were targeted (e.g. mackerel and tuna), although smaller pelagic fish and demersal fish associated with sandy benthos were also present (e.g. mullets). While the majority of boats landing to these fish markets were artisanal (typically manned by one or two fishers and around 8 m in length), these boats were numerous, with an estimated 16,000 vessels in ports in Ecuador according to 2008 figures (Lemay and Llaguno, 2008).

3.2 Key model predictions

All probabilities in the following model predictions are presented for nodes in the network increasing. A probability of > 0.5 indicates an increase is more probable than not, with higher numbers indicating stronger probabilities of increases. Probabilities < 0.5 indicate that the node is more likely to decrease, and the probability of decreasing can be found by:

$$1 - P_{increasing.}$$

Establishing greater controls on shark finning (in the model, altering the prior for shark finning increasing to 0.1) resulted in a high probability of increase of megafauna and top predators (both > 0.8, both of these groups included shark species – Table 4). Reef fish may show a decline in population size due to trophic interactions, but other biodiversity demonstrated little probability of change (< 0.05 change from 0.5), as did the effect on the majority of habitat types. An increased probability of fishing (as a result of boats being diverted from specifically targeting sharks) was predicted (0.59), and tourism showed a slight probability of improvement (0.53). The economy (from the marine and coastal areas) showed a decline (0.32), although this would largely be related to the illegal economy (selling of shark fins) as the process of intentionally catching sharks for finning is prohibited. The legal economy would be likely to increase due to fishing and tourism increases.

A second scenario involved decreasing all fishing activities (reducing fishing to 0.3 and shark finning to 0.1 – full results presented in Table 4). Such a management change would have a big effect on all functional animal groups in the reserve (Table 4). Sand, rock and fringing reefs also showed increased probability for improved biodiversity at these sites. However, a negative effect is predicted for the economy, if these activities are reduced (0.32). This value for the economy is identical to the previous scenario, just involving reduction is shark finning, and this lack of further reduction to the economy is related to a higher likelihood of increase in tourism (0.58).

To investigate whether a reduction in fishing and shark finning income could be offset by tourism, the previous example (reduced fishing and reduced shark finning) was investigated with a probability of increased tourism of 0.8 (full results given in Table 4). The probability of the economy contracting was reduced from previous scenarios, but still likely to contract (0.42) and the biodiversity improvements were not as great as previously identified. Increases to subtidal species were found in all cases, with birds and structural organisms showing declines (bird nesting sites can be greatly affected by tourism, and structural organisms such as coral may suffer from increased damage unless activities such as diving are carefully regulated). Biodiversity or ecosystem health of other habitat types was likely to improve.

Ecotourism in the marine environment is poorly developed and regulated in Ecuador (see discussion), so a further scenario, with adjusted 'edge' probabilities for the interactions of tourism and other network nodes was developed. This involved improved regulation of activities such as whale watching (probability of increasing with an increase in tourism from 0.4 to 0.45), removal of the link between top predators and tourism (through sport fishing practices), change in the effect of tourism on bird populations (from 0.4 to 0.45), and of tourism on structural organisms (from 0.2 to 0.4), finally, an increase in the money into the economy from tourism, from 0.7 to 0.9. These changes resulted in almost no change to the economy (0.49), but increases to all marine organisms and connected habitats (except birds $p = 0.50$, structural organisms $p = 0.46$ – full results in Table 4).

Finally, if no action was taken within the country, but external events resulted in increased recruitment of biodiversity (the 'external' node increased to 0.8), improvements would be found across the board. Biodiversity would increase (generally > 0.6 for all groups and habitats – full results in Table 4). Tourism, fishing and shark finning would all increase (in the absence of other regulation) and the overall economy would likely improve (0.54). However, a decrease in recruitment from external sources would have an equally detrimental effect (Table 4). Populations of the groups would decrease, health of habitats would decrease and the effects on ecosystem services and economy would suffer, with probabilities of decreasing equal to the probabilities of increasing in the reciprocal case.

4. Discussion

The coastal regions of mainland Ecuador are rich in biodiversity and possess abundant marine life. However, reef systems, both inside or outside of marine parks show that the systems are far from undisturbed. Sharks and other large predatory fish are largely absent from many locations, and only small skipjack tuna were observed during any survey. Invertivore reef fish and planktivorous fish such as chromis were abundant, as were green turtles. Humpback whales were clearly present close to dive sites, with many sightings from boats, and being frequently audible when diving.

Typically, those fish which were absent are also those of high commercial value for food (large pelagics such as tuna) or for other purposes such as shark finning. These species are often highly migratory, and as such may be fished in deeper waters, or in non-protected areas, although small scale fishing was observed within Machililla, and is not a prohibited activity under the park's legislation, and fishing for pelagic fish is encouraged over benthic trawling (INEFAN, 1998).

In general, previous studies investigating effectiveness of MPAs have shown that fish biomass increases, and this can be especially noticeable in higher trophic level species (often those targeted by fisheries; Sciberras et al., 2013; Guidetti et al., 2014). However, most high profile studies are conducted on large reserves with strict restrictions on fishing, and largely only report fish biomass increases (Costello and Ballantine, 2015). Several studies have questioned the effectiveness of marine reserves to protect biodiversity in general, especially when fishing is allowed in the protected areas (Edgar 2011; Costello and Ballantine, 2015). A recent systematic review of the effectiveness of MPAs also found that no take areas were most effective in enhancing fish biomass, but there was significant variability in effectiveness of other marine protected areas which allowed fishing. Smaller reserves were generally less effective, and regulations on which fish could be targeted also played a role (Sciberras et al., 2013). In the current study, high tropic level fish were low in abundance in all areas, protected or otherwise, indicating problems with the marine reserves and the marine environment in general. These problems could be related to size of reserves, those in the current study are generally small, but most likely the lack of regulation of activities in the reserves, for example, high levels of artisanal fishing are not prohibited (INEFAN, 1998). We can be confident that high trophic level fish were not abundant anywhere in our surveys, and that the community structure of the most common species was unaffected by the reserves. However, given our survey design, we cannot say with certainty that protected areas did not benefit rare species, as such, we may not have fully evaluated the potential of marine reserves in protecting biodiversity in this study.

As an industry, fishing is important in Ecuador. Official figures show it is worth $540 million to the economy per year (Lemay and Llaguno, 2008). Artisanal fishing boats are numerous, and are integral to the fishing communities. Removing such industry with no alternative of replacement would be difficult, as coastal areas of Ecuador are also the areas of highest poverty (Gravez et al., 2013). Policy shifts have recognised the importance of social, political and cultural issues around the establishment of marine reserves (Teran et al., 2006), and participatory management from stakeholders has been developed for some reserves (Gravez et al., 2013). However, in general, legislation is poorly formed and poorly enforced. For example, there is no clear regulation of fishing effort in the legislation for Machililla (INEFAN, 1998). Regulation of fishing effort, both inside and outside of reserves, is determined annually by the Undersecretary for Fisheries Resources, but it is reported that there are few resources to enforce these regulations (Gravez et al., 2013). Due to allowing fishing in reserves, and altering catch regulations each year, it is difficult to detect illegal fishing activity, or for the majority of concerned stakeholders to know what is allowed.

The predictions from the model, however, clearly show that regulation and enforcement of fisheries, and of illegal practices such as shark finning, would greatly improve biodiversity, both within the reserve, but also in many of the surrounding areas. Increases from marine ecotourism can somewhat buffer the effects of these decreases in traditional activity, but actively increasing tourism, without proper regulation, would decrease the gains in biodiversity achieved through restricting fishing and finning. At present, much of the marine tourism present does not demonstrate ecological credentials, and safety concerns over some aspects are likely preventing growth of higher income activities.

It should be noted, however, that well developed ecotourism can only mitigate for the modelled loss of economic provision from fisheries, it is unlikely to provide higher income in the short-term. As such, to bring about such a change in employment would require understanding of the fact that levels of income from fishing may be unsustainable in the long term, as fish stocks, and particularly income from shark fins, will decline in future years as populations decrease, indeed, evidence from recent news stories suggests that demand for shark fin has markedly declined in recent months, and as such, money from these activities will also rapidly decrease (Whitcraft et al., 2014).

While ecotourism is abundant in the Ecuador owned Galapagos Islands, the number of visitors is rapidly increasing raising concerns from many conservationists, some of which are very long founded (de Groot, 1983; Mejía and Brandt, 2015). However, many charismatic species in the Galapagos are also present in mainland Ecuador; for example, blue- and red-footed boobies, frigate birds and marine life such as manta rays, eagle rays and turtles. In addition, the mainland has a seasonal abundance of humpback whales, often seen from the shore, or from boat trips a short distance from the shore. Coral is also more abundant in mainland Ecuador, either through numerous sea fans or in some cases reef building corals (e.g. in shallow waters surrounding Isla de Plata and Isla de Salango in the Machililla national park). Large areas of protected mangroves are also present, giving opportunities for viewing birds and in some cases, dolphins. Mainland Ecuador, however, is lacking in high trophic level predators, such as sharks, large tuna and other large pelagic fish frequently seen in the Galapagos. While oceanographic features contribute to the abundance of large predators in the Galapagos, the reason for their almost total absence off mainland Ecuador is related to overfishing and shark finning, which has practically removed sharks from shallow reef areas over the last 15 years (Techera and Klein, 2014).

On the mainland, the majority of marine ecotourism is poorly run, although there are some operations providing first rate services in terms of health and safety and ecological sensitivity. For example, while there are restrictions on boats for whale watching (e.g. remaining a distance 100 m as a regulation for Machalilla national park), in practice most boats not only go much closer than this, but frequently several boats will chase a small group of whales for around one hour. The negative effects of poorly controlled whale watching activities have been well documented (Christiansen and Lusseau, 2014). Of equal importance is the fact that when whales were observed from land, or from boats not concerned with whale watching (or those keeping a good distance from the whales), far more diverse behaviour could be seen, such as fin and tail slapping and breaching. As such, proper regulation of whale watching could create a higher value activity, but with fewer negative ecological effects on the whales.

SCUBA diving is another example of how ecotourism could become more high value and less environmentally damaging. From the authors' experiences, in many locations, SCUBA diving did not seem to be restricted to those with diving certification. Small boats operated without any safety equipment (such as oxygen cylinders) or essential spare parts (O-rings to prevent leaks from cylinders). Dive briefings were very short or non-existent, and always in Spanish with no means of translation. Dive guides showed little respect for the marine environment, kicking corals with their fins, and chasing or antagonising larger fauna such as sea-turtles or moray eels to obtain photographs. Adherence to decompression limits was also frequently ignored, and safety stops frequently cut short or not conducted. In some areas of Ecuador, although SCUBA diving was offered as an activity, the offered activity was in fact diving with a surface supply hose (used by local octopus fishermen). The equipment appeared homemade, with the air intake close to the compressor exhaust, and had such activities normal required considerable further training beyond the normal SCUBA qualifications. Increasing safety concerns would allow for increased cost of provision (indeed, the better established SCUBA operations, with improved environmental awareness and safety procedures are significantly more expensive).

These high value activities may be better suited to international tourists, or wealthy residents, who may have more disposable income. The Galapagos Islands are now very popular with international eco-tourists. Flights to Galapagos only originate from Ecuador, so considerably more could be made of the eco-tourism potential of the mainland as part of an Ecuador trip. Tourists visiting mainland sites in addition to the Galapagos would not only provide good income for the country as a whole, but would mean the development, farming and water supply pressures facing Galapagos could be reduced if fewer visitors were there at once (Benitez-Capistros et al., 2014).

A simple fix to providing higher value and lower environmentally damaging activities would be to ‘buy in’ operators, either internationally (from the US or Europe, for example), or nationally, from the big cities. However, such an approach is unlikely to benefit conservation of the marine environment. Reductions in allowable fish catches and removal of the highly profitable shark finning trade would mean that local communities will lose valuable economic income. These communities are already the poorest in Ecuador (Gravez et al., 2013). To ensure successful outcomes of these conservation measures it is necessary to both ensure that money from tourism stays in the local community and also that people’s roles in the community are still evident after the change in emphasis. Artisanal fishing is an activity that has occurred for generations in many of these communities, and although the predictions of models do not need to eliminate artisanal fishing practices, to create better and healthier marine ecosystems, they do need to be reduced. Providing an alternative job of similar social standing would be important in ensuring such a transition can occur.

Clearly, transitions which require education, language skills and restructuring of employment will take time to occur. Understanding this will take time is vital in successful implementation. The ability to create greater income at a national level by simply increasing demand for tourism is not a sustainable approach and will harm, rather than benefit wildlife, especially if reductions in harmful practices such as shark finning and overfishing do not occur. Slow changes also mean that rather than having to retrain for jobs, the younger generation can be more involved in tourism than some of the traditional practices, and the older generation can continue with their traditional roles.

However, conservation measures do still need urgent implementation. For example, a clamp down on shark finning is essential as soon as possible. Although this is an illegal activity in Ecuador (Franciso-Fabian, 2001), it has been well documented in news stories in recent months; with loopholes in the law and lack of enforcement by officials allowing the exploitation of sharks (Jacquet et al., 2008). Partly this will allow populations of sharks to begin to increase in the area (creating more demand for tourism activities such as diving in the longer-term), but also the low number of sharks is leading to increased levels of illegal activity in other areas, such as the Galapagos; one of the few remaining areas with healthy shark populations (Schiller et al., 2014).

While the results of the model suggest it is possible to decrease harmful fishing, and install better eco-aware tourism practices which will balance the coastal economy and enhance wildlife, external events are also important. Some external events may be beyond the control of a single country, but others, such as the careful maintenance of the Galapagos marine reserve, may play an important role in ensuring mainland Ecuador's biodiversity continues to flourish. Equally, although not quantified here, further damage to the mainland ecosystems (again, especially in relation to highly migratory species such as sharks) could have negative impacts on external communities including neighbouring countries and the Galapagos.

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Table 1. Interactions in the Bayesian belief network. Each row interacts with certain columns, light grey squares with numbers > 0.5 indicate a positive interaction, so if the probability of the row begins to increase, then the probability of the column increasing will also increase along with Bayesian inference. Dark grey squares with values < 0.5 indicate a negative interaction, and if the probability of the row increasing becomes higher, then the probability of the column increasing will decrease. The numbers of the columns correspond to the numbers of each row. 1-7 (no highlighting) are populations of organisms in different functional groups inside Machalilla National Park. 8-12 (light grey highlighting) represent biodiversity and abundance of organisms in habitats which may be connected to Machalilla. 13 – 17 (dark shading) represent processes and industries which may contribute to income to coastal communities. 18 (black shading) is the overall effect on the coastal economy.

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**
1 Megafauna										0.8			0.8	0.6			0.7	
2 Top Predators			0.2							0.8			0.8	0.7			0.7	
3 Reef Fish		0.75		0.3	0.2		0.6	0.6	0.7	0.8	0.8		0.8	0.8				
4 Prey Species	0.6	0.7	0.8			0.8	0.6	0.7	0.8	0.8	0.8			0.65				
5 Invertebrates			0.8	0.7		0.6	0.7	0.55	0.7	0.8	0.7		0.7	0.7				
6 Birds			0.3	0.2				0.8	0.8	0.7	0.6		0.8					
7 Structural organisms			0.7	0.8	0.8				0.7	0.8			0.8					
8 Sand			0.6	0.7	0.6	0.8			0.7	0.7	0.6		0.6	0.8				
9 Rock			0.8	0.8	0.8	0.8	0.7	0.7		0.8	0.7			0.8				
10 Fringing Coral		0.7	0.9	0.9	0.9	0.7	0.9	0.7	0.8		0.8		0.7	0.7				
11 Mangroves		0.6	0.8	0.8	0.6	0.65		0.7	0.8	0.8			0.6	0.6				
12 External	0.8	0.8	0.65	0.65	0.65	0.7	0.65	0.7	0.7	0.7	0.6							
13 Tourism	0.4	0.4				0.4	0.2							0.7			0.3	0.7
14 Fishing		0.2	0.1	0.3	0.25			0.2	0.2	0.2								0.8
15 Aquaculture			0.4	0.4	0.4	0.4	0.3	0.45	0.3	0.3	0.2			0.3				0.8
16 Development								0.3	0.2	0.2	0.1							0.7
17 Shark finning	0.2	0.1												0.4				0.7
18 Economy													0.75	0.4	0.6	0.8	0.2	

Table 2. Definitions of the nodes in the BBN

Node	Definition
Megafauna	Large species with high ecotourism potential including whales, turtles, manta rays and whale sharks. Excluding reef shark species
Top predators	Sharks (excluding whale sharks), big pelagic predators such as tuna and other game fish. Large groupers
Reef fish	Typical coral reef fish such as large angel fish, snappers, small groupers, parrotfish, trumpet fish etc
Prey species	Smaller shoaling reef fish. E.g. Chromis
Invertebrates	Crabs, lobsters, starfish. Excluding structure building inverts such as coral
Birds	Birds which feed on marine fish or inverts. Pelicans, frigate birds, boobies etc.
Structural organisms	Algae, sea fans and coral which increase structural complexity

Sand	Overall biological richness of sand bottomed habitats (based on number of species and abundance)
Rock	Overall biological richness of rocky coastline, defined as above. Refers to shallow rocks < 10m deep and close to shore
Fringing reef	Overall biological richness of reefs which are not in protected areas
Mangroves	Overall biological richness of mangrove systems
Fishing	Total fishing effort. By default this assumes fishing occurring in protected areas at typical levels, but some simulations exclude illegal fishing and these are clearly indicated in results
Tourism	Tourism based around marine activities such as diving and whale watching. Default parameters assume no changes in current tourism practices
Aquaculture	Development of aquaculture projects in coastal areas

Development	Building work for accommodation, tourism or infrastructure
Shark finning	Fishing purposefully for elasmobranchs to sell shark fins
External	External influences (e.g. recruitment, dispersal of species) on marine systems from countries outside of continental Ecuador
Economy	Economic output from coastal marine ecosystems

Table 3. Common fish and other mobile fauna used to classify different habitat types. Species are ranked by the overall score, depending on abundance in different habitats. Scoring per habitat is on the DAFOR scale (d = dominant (10), a = abundant (6), f = frequent (4), o = occasional (2), r = rare (1)). Those with scores ≥ 12 (calculations defined in methods) were used in statistical analysis of classification.

Family	**Common Name**	**Scientific Name**	**Feeding**	**Machililla**	**Other Reserves**	**Non-Reserve Reefs**	**Rock**	**Sand**	**Score**
Pomacanthidae	Scissor-tailed chromis	*Chromis atrilobata*	Planktivores	d	d	d	d	o	42
Diodontidae	Balloonfish	*Didon holocanthius*	Invertivore	d	d	d	a	f	40
Chaetodontidae	Threebanded Butterflyfish	*Chaetodon humeralis*	Invertivore	d	d	d	a	r	37
Diodontidae	Porcupinefish	*Diodon hystrix*	Invertivore	d	a	a	a	a	34
Blennidae	Panamic Fanged Blenny	*Ophioblennius steindachneri*	Invertivore	f	f	f	f	f	20
Labridae	Cortez rainbow wrasse	*Thalassoma lucasanum*	Invertivore	f	f	f	a	o	20
Chaetodontidae	Barberfish	*Johnrandallia nigriostris*	Invertivore	a	a	a	o	r	21
Pomacanthidae	King Angelfish	*Holacanthus passer*	Invertivore	f	a	a	o		18
Dasyatidae	Stingray	Dasyatidae	Invertivore	a	f	f	o	f	20
Serranidae	Flag Cabrilla	*Epinephalus labriformis*	Piscivore	a	a	a	r		19
Serranidae	Serrano	*Serranus psittacinus*	Piscivore	a	a	a	r		19
Balistidae	Orangeside Trigger	*Sufflamen verres*	Invertivore	a	a	a	r		19
Serranidae	Pacific Creole Fish	*Paranthias colonus*	Planktivore	f	f	f	o	r	15
Cheloniidae	Green turtles	*Chelonia mydas*	Herbivores	a	f	f	r	o	17
Serranidae	Panamic Graysby	*Cephalopholis panamensis*	Piscivore	f	f	f	r		13

Balistidae	Blunthead Trigger	*Pseudobalistes naufragium*	Invertivore	f	f	f	r		13
Scaridae	Bumphead parrotfish	*Scarus perrico*	Herbivores	f	f	f	r		13
Aulostomidae	Trumpetfish	*Aulostomus chinensis*	Piscivore	f	f	f			12
Fistulariidae	Cornetfish	*Fistularia commersonii*	Piscivore	f	f	f			12
Octopodidae	Octopus	Octopodidae	Invertivore	o	f	f	r		11
Tetradonitdae	Guineafowl Puffer	*Arothron meleagris*	Invertivore	o	o	f	r	r	10
Carangidae	Steel Pompano	*Trachinotus stilbe*	Planktivore	o		f	f		10
Blennidae	Sabertooth blenny	*Plagiotremus azaleus*	Piscivore	o	o	o		o	8
Mullidae	Mexican goat fish	*Mulloidichthys dentatus*	Invertivore	o	o	o		o	8
Monacanthidae	Vagabond Filefish	*Cantherhines dumerilii*	Invertivore	o	o	o	r		7
Chaetodontidae	Duskybarred Butterflyfish	*Chaetodon kleinii*	Invertivore	o	o	o	r		7
Muraenidae	Fine Spotted Moray	*Gymnothorax dovii*	Piscivore	o	o	o	r		7
Monacanthidae	Scrawled Filefish	*Aluterus scriptus*	Invertivore	o	o	o			6
Ostraciidae	Pacific Boxfish	*Ostracion meleagris*	Invertivore	o	o	r	r		6
Tetradonitdae	Stripebelly Puffer	*Arothron hispidus*	Invertivore	o	o	o			6
Tetradonitdae	Longnose puffer	*Sphoeroides lobatus*	Invertivore	o	o	o			6
Muraenidae	Panamic green moray	*Gymnothorax castaneus*	Piscivore	o	o	o			6
Lutjanidae	Blue and Gold Snapper	*Lutjanus viridus*	Piscivore	o	o	o			6

Lutjanidae	Pacific dog snapper	*Lutjanus novemfasciatus*	Piscivore	o	o	o			6
Serranidae	Pacific Mutton Hamlet	*Alphestes immaculatus*	Piscivore	o	o	r			5
Scombridae	Black Skipjack	*Euthynnus lineatus*	Piscivore	r	r	o			4
Scorpaenidae	Stone Scorpionfish	*Scorpaena plumieri mystes*	Piscivore	r	r	r		r	4
Labridae	Spinster wrasse	*Halichoeres nicholsi*	Invertivore	r	r	r	r		4
Carangidae	Almaco Jack	*Seriola rivoliana*	Piscivore		r	r			2
Synodontidae	Lizardfish	Synodontidae	Piscivore	r	r	r			3
Syngnathidae	Pacific Seahorse	*Hippocampus ingens*	Invertivore		r	r			2
Myliobatidae	Eagle ray	*Aetobatus ocellatus*	Invertivore	o	r			r	4
Mobulidae	Manta Ray	*Manta biostris*	Planktivore	o	r	r			4
Ophichthidae	Tiger snake eel	*Myrichthys tigrinus*	Invertivore	r				o	3
Torpendinidae	Peruvian torpedo ray	*Torpedo peruana*	Invertivore	r				r	2
Hydrophiinae	Sea snake	Hydrophiinae	Piscivore					r	1

Table 4. Results of the different management scenarios from the Bayesian belief network. Bold values represent a probability of the node increasing of ≥ 0.55. Light grey values indicate a probability of the node increasing ≤ 0.45 (or of the node decreasing of ≥ 0.55).

	Reducing shark finning ($p_{increase}$ = 0.1)	Reducing shark finning and fishing effort ($p_{increase}$ = 0.1 and 0.3 respectively)	Reducing shark finning and fishing effort and actively increasing tourism ($p_{increase}$ = 0.1, 0.3 and 0.8 respectively)	As column to left, but enhancing high value tourism	No changes to management but external increase in diversity and populations ($p_{increase}$ = 0.8)	No changes to management but external decrease in diversity and populations ($p_{increase}$ = 0.2)
Megafauna	**0.81**	**0.79**	**0.64**	**0.66**	**0.72**	0.28
Top predators	**0.85**	**0.78**	**0.68**	**0.78**	**0.71**	0.29
Reef fish	0.36	**0.68**	**0.68**	**0.68**	**0.61**	0.39
Prey species	0.51	**0.61**	**0.60**	**0.60**	**0.61**	0.39
Invertebrates	0.52	**0.62**	**0.61**	**0.62**	**0.62**	0.38
Birds	0.51	**0.57**	0.46	0.50	**0.66**	0.34
Structural organisms	0.49	**0.57**	0.33	0.46	**0.62**	0.38
Sand	0.49	**0.65**	**0.64**	**0.64**	**0.65**	0.35
Rock	0.49	**0.66**	**0.64**	**0.65**	**0.66**	0.34
Fringing reef	0.54	**0.67**	**0.65**	**0.66**	**0.67**	0.33
Mangroves	0.49	**0.58**	**0.57**	**0.57**	**0.60**	0.40
Fishing	0.50	0.50	0.50	0.50	**0.80**	0.20
Tourism	0.53	**0.58**	**0.81**	**0.82**	**0.60**	0.40
Aquaculture	**0.59**	0.36	0.36	0.36	**0.57**	0.43
Development	0.45	0.46	0.48	0.50	0.51	0.49
Shark finning	0.36	0.37	0.45	0.49	0.52	0.48
External	0.12	0.12	0.09	0.09	**0.56**	0.44
Economy	0.32	0.32	0.42	0.49	0.54	0.46

Figure 1. Location of some of the key marine reserves in Ecuador, and the positions of the survey sites in this study. Data on marine reserve location modified from Marine Conservation Institute (2015).

Figure 2. Differences in community structure in different habitats assessed by bootstrapped PCA. a) differences at 95% confidence level. b) Differences at 99 % confidence level. Overlap indicates no significant difference between communities at given confidence level.

Supplementary material. The working Bayesian belief network is provides as a macro enabled Microsoft Excel file. The parameters of the model are identical to those used in the majority of scenarios in the manuscript and equate to those provided in Table 1. Source code for the model can be seen by looking at the VBA macro associated with this file.

Figure1

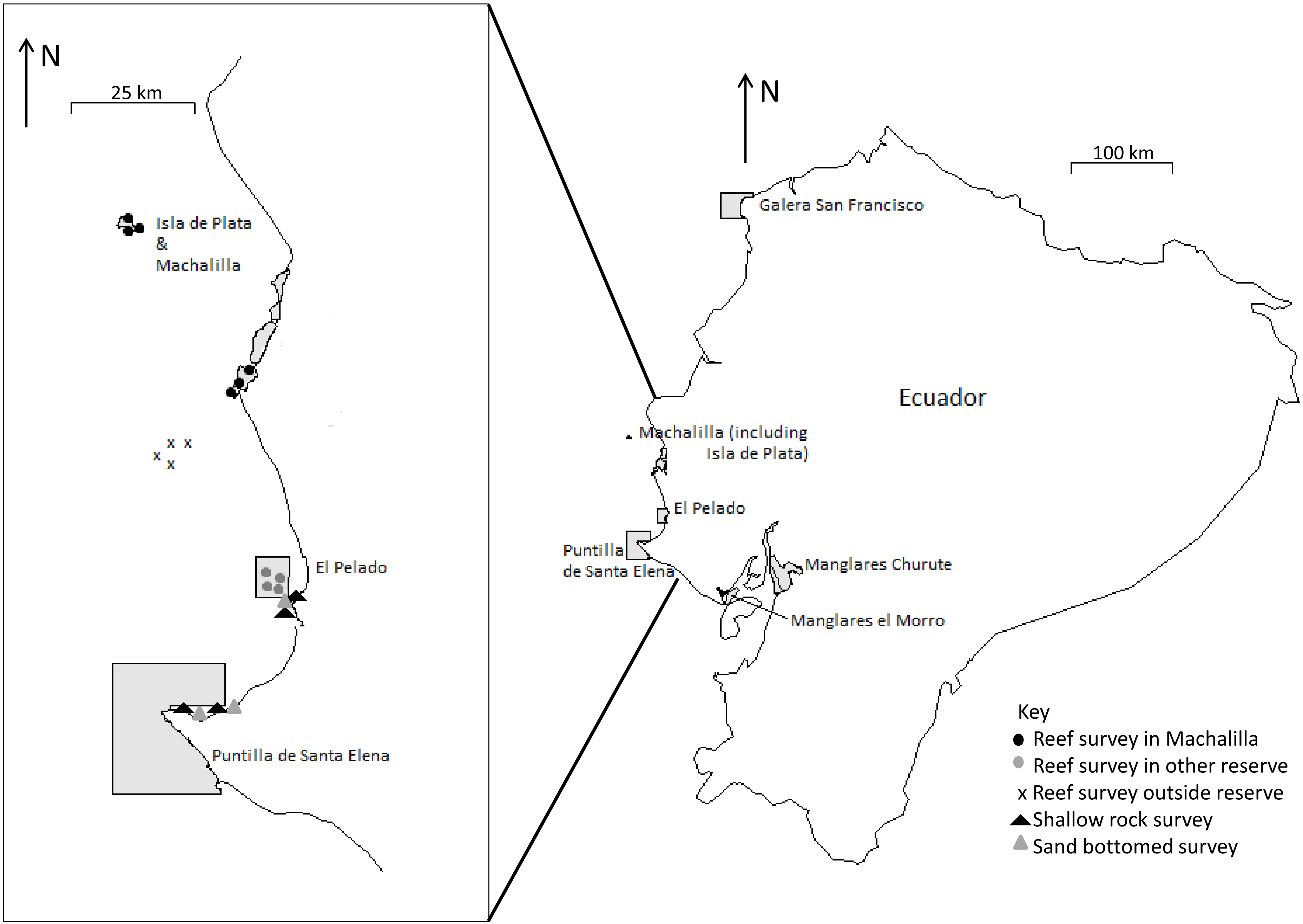
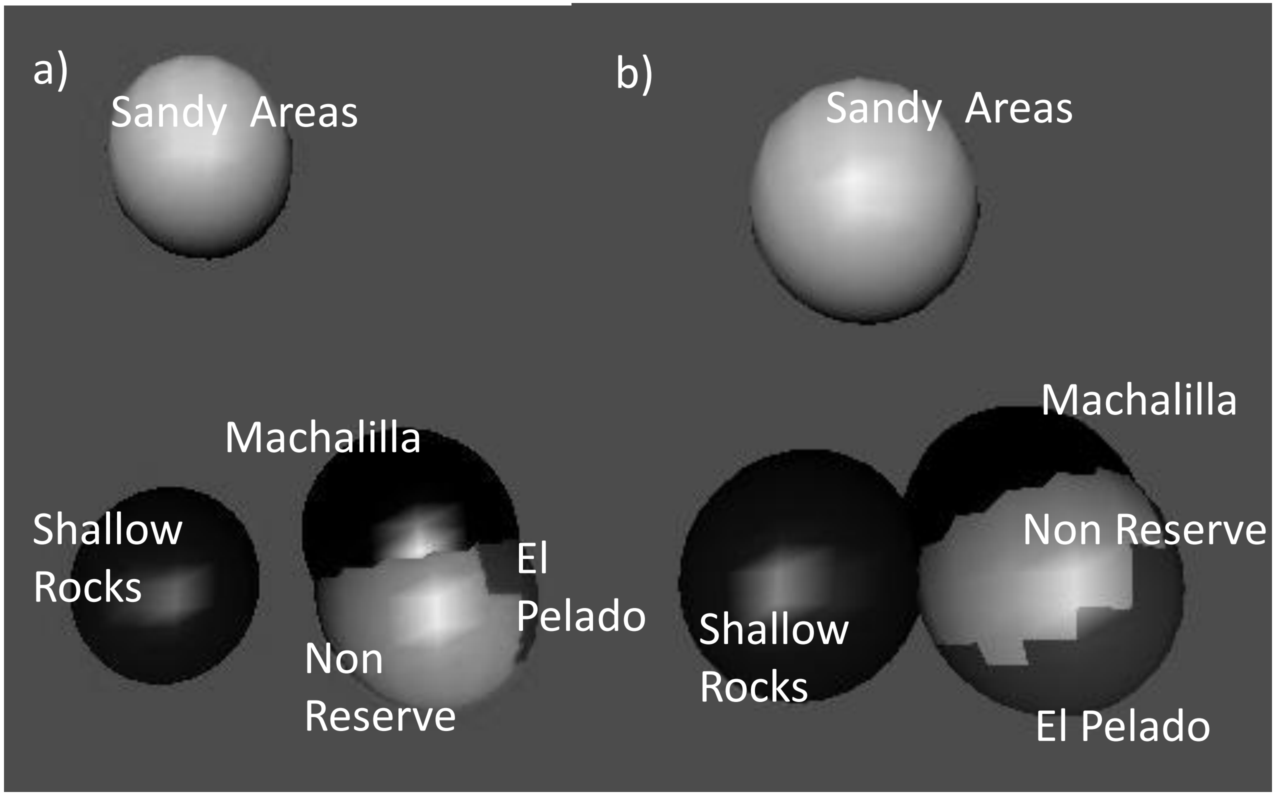

N
25 km
Isla de Plata
&
Machalilla
x x
x
x
El Pelado
Puntilla de Santa Elena
N
100 km
Galera San Francisco
Ecuador
Machalilla (including
Isla de Plata)
El Pelado
Puntilla
de Santa Elena
Manglares Churute
Manglares el Morro
Key
Reef survey in Machalilla
Reef survey in other reserve
x Reef survey outside reserve
Shallow rock survey
Sand bottomed survey

Figure2


a)
Sandy Areas
Machalilla
Shallow
Rocks
El
Pelado
Non
Reserve
b)
Sandy Areas
Machalilla
Non Reserve
Shallow
Rocks
El Pelado