Prioritising invasive species control actions: evaluating effectiveness, costs, willingness to pay and social acceptance.

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Abstract

At a global scale, island ecosystems are recognised as high priority for biodiversity conservation, with introduction of invasive species a significant threat. To investigate prioritisation of control of invasive species on islands, we conducted a cost effectiveness analysis of donkey control on Bonaire, Caribbean Netherlands. To be successful, prioritisation of conservation actions must take account of the ecological, economic and social aspects. Further improvements may be seen where impacts can be measured across ecosystem boundaries, and where management actions can be tied to funding sources. We modelled the expected ecological impacts of three potential control options, estimated costs of each option, and connected this to the willingness of beneficiaries to fund such projects. Finally we surveyed experts to understand the social acceptability of donkey control. Of the control options, eradication is predicted to have the highest ecological impacts in both the dry-forest and coral reef, and to be most cost-effective over the long term. Costs of all control options were within user willingness to pay. Social acceptability was highest for fencing, and lowest for lethal control. Though eradication offers the highest ecological benefits, we suggest that lower initial costs and higher social acceptability make fencing the more appropriate choice for Bonaire in the immediate future. In this way we illustrate the importance of considering economic and social impacts alongside the ecological in environmental conservation, and present an integrated application for prioritising conservation policy choices.

Keywords: environmental management; cost-effectiveness analysis; invasive species; willingness to pay; funding; island conservation

1. Introduction

Invasive species present a significant threat to ecosystems worldwide. This is particularly the case on islands, where species have been isolated from competition or predation pressure, and thus are less able to withstand invasions when they occur (Dawson et al., 2015; Martins et al., 2006). Understanding the impacts of invasive species and the tools available for their control is important for prioritising environmental conservation actions. While evaluations of the cost-effectiveness and social acceptability of alternative control options are becoming more widespread, studies drawing these together with potential funding mechanisms remain scarce. Given the large impacts of invasive species on islands, further gains in environmental conservation may also be observed where such prioritisation is able to consider impacts across ecosystem boundaries (e.g. terrestrial to marine).

Eradication is widely regarded as the most environmentally effective solution to control damage by invasive species (Cruz et al., 2009; Donlan et al., 2003). However high economic costs or social concerns may make eradication a less-preferable or indeed an inappropriate action. Alternatives to eradication, such as lethal (Saunders et al., 2010) and non-lethal (Reiter et al., 1999) population control, or restricting species movement, often need to be considered (Cruz et al., 2009; Donlan et al., 2002).

Prioritising actions to tackle ecological degradation caused by introduced species requires prediction of environmental states both with and without action, to identify the added environmental value of proposed initiatives (Maron et al., 2013). This can be challenging, partly due to the long time scales involved with recovery (Shwiff et al., 2013). In addition, the highly specific spatial and temporal variation associated with costs and benefits of environmental conservation (Armsworth, 2014; Balmford et al., 2003; Cullen, 2013) limits the transferability of studies between locations. Invasive species control is associated with high economic costs, while environmental management remains chronically underfunded (Armsworth, 2014; Boyd et al., 2015; Bruner et al., 2004). Prioritisation of environmental conservation, and invasive species in general, has drawn upon risk analysis (Harwood, 2000), decision analysis (Maguire, 2004), multi-criteria analysis (Liu et al., 2011; Mendoza and Martins, 2006) and return on investment analysis (Boyd et al., 2015), among others, to incorporate multiple uncertainties, objectives and stakeholders involved in prioritising conservation actions. However the high data needs of such methods presents a barrier to many projects. As such we present here an initial step towards prioritisation of conservation actions, and the analysis presented in this paper may inform the basis of a more in-depth prioritisation plan.

This paper is the last in a series of papers investigating the impacts and control of invasive grazing species on the island of Bonaire in the Caribbean Netherlands (12° 10’ N 68° 17’ W). Previous work has modelled the relationship between ecosystem characteristics and natural variation in invasive species densities, estimating a negative relationship between grazing pressure by donkeys and vegetation ground cover (Roberts, 2017). We demonstrate how these models can be utilised to estimate the impacts of alternative management strategies (in this case donkey control) on ecosystem characteristics. We draw on models developed in Roberts et al 2017b, which estimate a positive relationship between terrestrial vegetation and coral reef health, to illustrate the impacts that invasive species control can have across ecosystem boundaries. Though estimating costs of invasive species control is fraught with difficultly (de Brooke et al., 2007; Donlan and Wilcox, 2007; Martins et al., 2006), inclusion of even broad cost estimates have been shown to be valuable to prioritising conservation actions (Boyd et al., 2015). We therefore estimate the costs of actions and relate these to predicted environmental impacts from Roberts et al 2017 & 2017b to assess the cost-effectiveness of each control option.

Conservation actions are limited by restricted funding (Bruner et al., 2004). Since the persistence of conservation programs is more likely where they are self-financed (Whitelaw et al., 2014), user fees have the potential to greatly improve conservation gains. As alternative conservation actions are expected to have varied environmental outcomes, user willingness to pay should vary across actions. Quantifying the willingness to pay of those who benefit from conservation actions, using a payment mechanism deemed acceptable by users, provides valuable information for availability of funding, and therefore the long-term economic sustainability of the project. In Roberts et al 2017a we estimated willingness to pay of SCUBA divers for control of terrestrial invasive species, where this would be expected to improve reef health. In this paper we use those estimates to calculate willingness of SCUBA divers to pay for the coral reef improvements predicted to arise from the alternative donkey control strategies.

Finally, addressing social concerns has been recognised as of high importance for successful invasive species control (Guerrero et al., 2010; McLeod et al., 2015). Failing to account for social acceptability of actions can lead to unforeseen costs and delays, public opposition, and cancellations of management actions (Frank et al., 2015; Lodge and Shrader-Frechette, 2003; Moon et al., 2015). We therefore present an initial overview of the social acceptability of each donkey control strategy, and discuss further work needed before any action can be implemented.

2. Methods

Drawing together the four criterion needed for prioritising conservation actions (conservation effectiveness (Roberts, 2017; Roberts et al., 2017b); economic costs; willingness to pay of beneficiaries (Roberts et al., 2017a), and social acceptance), we analyse options for invasive species control options, and make recommendations for future management in our study site. This approach is particularly applicable to sites where data and expertise for formal risk analysis, feeding into multi-criteria analysis, is not available. The process followed in this paper is summarized in Fig 1.

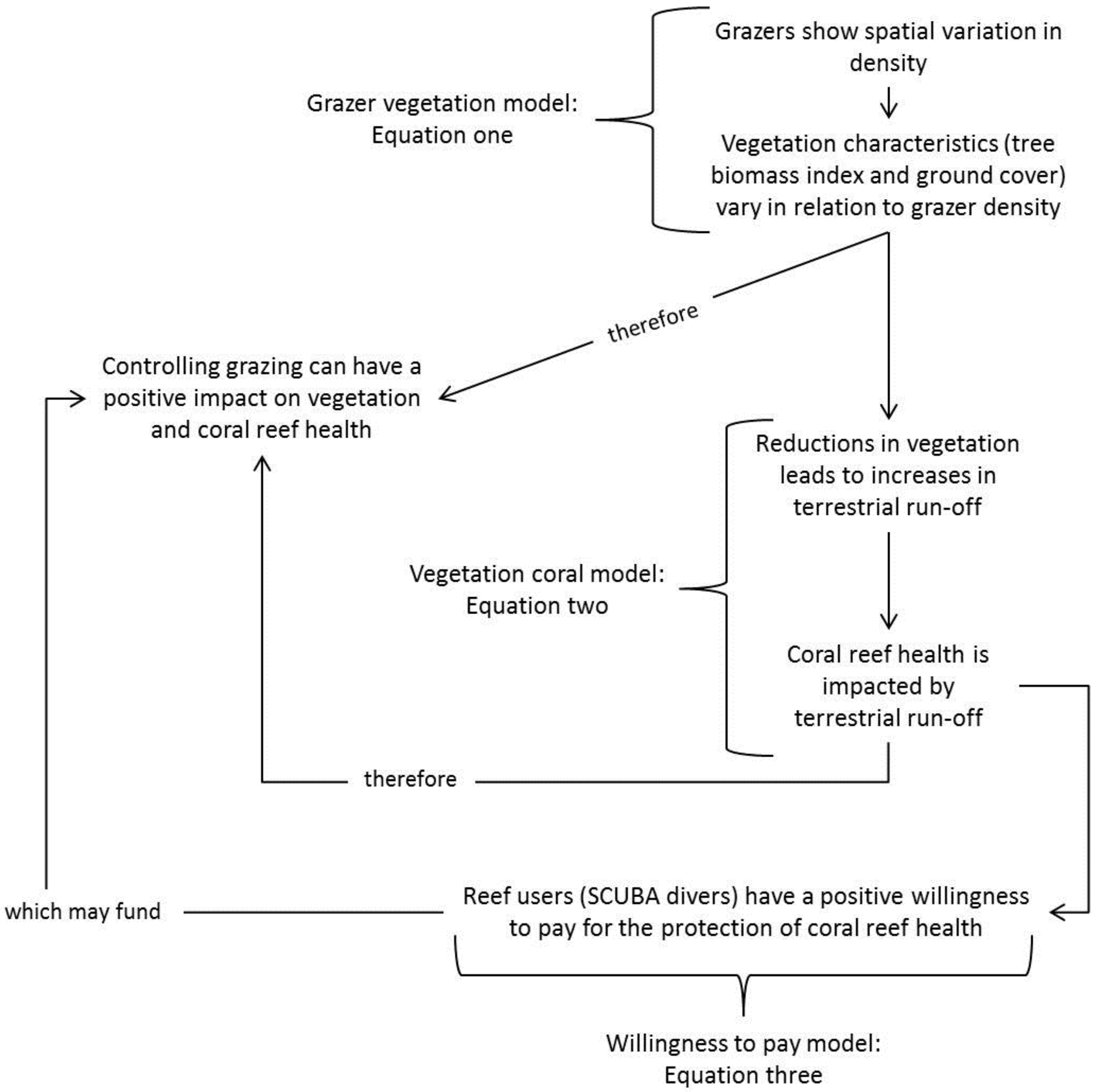

Fig 1 Map to indicate relationship between vegetation, coral reef, potential diver funding and controlling of grazing

2.1 Study system

The island of Bonaire, Caribbean Netherlands, is a highly-regarded SCUBA diving destination, and has an extensive marine conservation program (Steneck et al., 2015). However the island has a long history of terrestrial degradation, as invasive goats, donkeys and pigs were introduced for farming as early as the 16th Century (Westermann and Zonneveld, 1956). Today all three species have established feral populations (goats: 30,000 (Cado van der Lelij et al., 2013), donkeys: 1000 (unpublished data), pigs <1000 (unpublished data), whilst goats continue to be farmed, with privately-owned goats allowed to roam free alongside the feral population. As a result, Bonaire's dry forest is now characterised by only a few surviving trees and by low levels of vegetation ground cover (Freitas et al., 2005). Low vegetation cover is associated to increased sediment run-off, due to reduced root systems, which otherwise anchor soils (Álvarez-Romero et al., 2011; Maina et al., 2013; Mateos-Molina et al., 2015). Increased sediment levels adversely impact the coral reefs surrounding Bonaire. Increased suspended sediment is associated to reduced light levels, which slows coral growth rates (Pollock et al., 2014), reduces structural stability (Erftemeijer et al., 2012) and disrupts coral (Jones et al., 2015) and fish (Wenger et al., 2014, 2011) development and recruitment. Nutrient levels are also increased, which promote macroalgal growth and smothers hard corals (De'ath and Fabricius, 2010). Settling sediment can lead directly to coral mortality, as well as restricting feeding polyps, altering coral morphology (Erftemeijer et al., 2012), promoting disease (Weber et al., 2012) and disrupting fish communities (Goatley and Bellwood, 2012). Further disruption to recruitment is seen as juvenile corals struggle to establish on high sediment substrates (Jones et al., 2015). Such damage to coral reef system decreases its attractiveness to divers. Consequently, terrestrial degradation is recognised as threatening Bonaire's marine ecosystems (Slijkerman et al., 2011; Wosten, 2013), a situation which is common with many other coral reef systems worldwide.

2.2 Control options

Options for mitigating the ecological damages due to over-grazing by donkeys, goats and pigs were identified through communication with local stakeholders (Bonaire Island Government; Bonaire conservation organisation, Echo; National Park Authority STINAPA). Three management strategies were considered:

1. Fencing of designated nature areas (**Error! Reference source not found.**);
2. Lethal control of feral donkey populations (reducing populations but not eliminating them);
3. Eradication of feral donkey populations.

Due to the high densities of goats recorded across the island it was not possible to model the impacts of goat control, as no variation in goat grazing pressure was observable. Conversely pig densities were too low across the island to enable modelling of pig impacts . For these reasons we have considered only donkey control within this study.

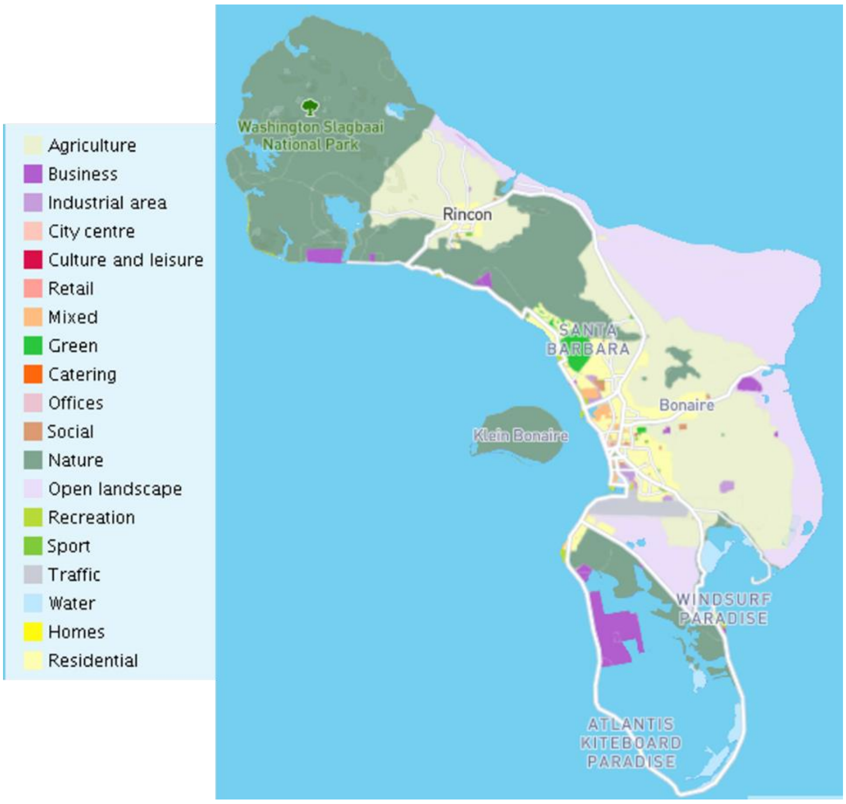

Fig 2 Bonaire Zoning Plan, showing nature areas in dark green. (Openbaar Lichaam Bonaire, 2011)

2.3 Quantifying grazer impacts on vegetation health

Vegetation characteristics anticipated to impact reef health were identified as tree biomass and percentage ground cover (Aguirre-Muñoz et al., 2008; Rojas-Sandoval et al., 2014). These characteristics were estimated within 101 quadrats of 100m^2, randomly located, stratified by landscape type. Due to low densities of donkeys point counts were not possible, therefore donkey densities were estimated from transect counts, with a density index calculated from the number of donkeys observed at a given location, divided by the number of visits to that location. Kernel density estimation was then used to extrapolate this data to create a density map across the island, from which estimated density at each point could be extracted. General linear models were used to estimate the relationship between donkey density and tree biomass (estimated from height and diameter, no attempt to estimate belowground biomass was made) or vegetation ground cover (data log transformed). Vegetation ground cover was estimated to be negatively impacted by dry season donkey density. Tree biomass did not show any variation with variables modelled (Appendix A).

We calculated the predicted impacts on ground cover of each grazer control strategy. To calculate ground cover for fencing estimates were first made for median and zero donkey density. Weighted means of these estimates were used to calculate ground cover for fencing from zero to 41% of island area (0ha – 1,208ha, area covered by nature areas which when fenced will have a donkey density of zero). Ground cover following donkey control and eradication was estimated from zero to maximum donkey density (max donkey density index = 18). Estimates of ground cover if no action were taken were estimated using median donkey density. Median density was used because grazer populations on Bonaire are well established, and therefore likely at equilibrium within the ecosystem. Sensitivity of models to errors associated with the estimates was tested through repeating calculations using the upper and lower 95% confidence intervals for donkey density impact. For full explanation of methods and results see (Roberts, 2017).

Due to low spatial variation in both goat and pig densities we were not able to model their impacts on vegetation, and therefore concentrate on donkey impacts only. This limits the outputs of our model in two ways. When considering removal of multiple species, such as would be the case in fencing, we are able to estimate only the benefits arising from donkey control, likely underestimating impacts. Conversely when estimating impacts of donkey eradication we are not able to incorporate potential for goats or pigs to fill the niche, and may therefore over estimate impacts (though that a relationship is observed between ground cover and donkey density at the current goat and pig densities suggests that some reduction in grazing would be observed with the removal of donkeys alone).

2.4 Quantifying vegetation impacts on coral reef health

Coral reef characteristics predicted to be affected by sedimentation rates were identified through a review of the literature as: coral cover (at 5m and deeper than 5m) (Erftemeijer et al., 2012; Jones et al., 2015; Pollock et al., 2014); visibility (Mateos-Molina et al., 2015; Risk, 2014); and fish community (abundance; species richness; and diversity) (Goatley and Bellwood, 2012; Wenger et al., 2014, 2011). A full explanation of methods and results can be found in Roberts et al. 2017b, and we will give only a brief overview here. Visibility and coral cover data were mapped using citizen science data collection, with fish data collected from the REEF fish database (REEF, 2016). Vegetation characteristics were measured at 101 sites across Bonaire, and average vegetation ground cover and tree biomass estimated for each watershed. General linear models were then used to estimate the impacts of vegetation characteristics on each of the coral reef characteristics measured. Coral cover below 10m depth was the only model to show a significant relationship to watershed characteristics. A positive relationship was found between coral cover and vegetation ground cover, interacting with tree biomass to show a larger positive impact when tree biomass was high. Tree biomass showed a negative relationship to coral cover, with high impacts when ground cover was low. Coral cover was also positively impacted by distance from town and presence of a salina on the watershed, and negatively impacted by the site being shore accessible to divers, and adjacent to urban areas (Appendix B).

We estimated changes in coral cover for each grazer control option. For all calculations, tree biomass and distance from urban areas were input as median values, and sites treated as shore accessible. Ground cover was entered using the estimates calculated above. To enable comparison to environmental condition with no control (Maron et al., 2013) coral cover was estimated using median ground cover estimates.. Due to the unbounded nature of the model, estimates of coral cover arising from donkey control were estimated beyond the possible range for coral cover. Cover reported in figures is restricted to between 0 and 100%. Sensitivity of the model to errors associated with the estimates were tested through repeating the calculations for upper and lower 95% confidence intervals of ground cover.

2.5 Economic costs and grazer control strategies

Economic costs are estimated only for material and labour involved in donkey control. Only government owned 'nature areas', covering 41% of the island (1,208ha, **Error! Reference source not found.**), are considered for fencing, because these are the only areas in which farming is currently prohibited, and could therefore be effectively fenced. As the donkey population is feral, reducing the population does not impose financial losses on individuals. Costs could not be calculated for loss of grazing for free ranging goats associated with the establishment of fenced areas.

Costs for fencing were adapted from budgets for a fencing project begun by Echo on Bonaire in 2016. This included materials, labour, transport, and administration costs. Labour and material costs were scaled up proportionally with the size of the project, whilst infrastructure and administration costs increased at 10% of proportional costs. An additional 10% was added to each budget to reflect underestimation of costs in initial budgets (S. Williams & L. Schmaltz, pers. comm.).

Control and eradication costs were initially estimated using costs reported in the literature. A search of Web of Science for: eradication and ungulate or goat or donkey or pig returned 81 relevant papers, of which six reported costs (Cruz et al., 2009; Holmes et al., 2015; Martins et al., 2006; Massei et al., 2011; McCann and Garcelon, 2008; Melstrom, 2014). Costs for control were estimated using median cost per hectare, and repeated using the lower and upper quantile.

Following communication with industry experts (Chad Henson, Island Conservation), Bonaire specific costs were also calculated. Costs were estimated for a two year long program using only ground hunting (including corrals and dogs), and for a 14 month long program with the additional use of helicopter for two months. Costs of confirming eradication were estimated for 6, 12, and 24 month programs. Control costs were estimated as a proportion of the total eradication costs. Full cost estimates can be found in Appendix D. It is important to note that even when considering a single control option, variations in costs occur depending on exact design of control efforts, particularly where and when actions are concentrated (Baker and Bode, 2016). Because we have not considered such cost variations here the values presented should be recognised as estimates only, and a full cost analysis would be needed to design the most appropriate control schedule.

2.6 Funding grazer control strategies

Choice experiments (Grafeld et al., 2016; Hanley et al., 2003; Train, 2009) were used to estimate the maximum willingness of SCUBA divers to pay for terrestrial grazing control, where this would be expected to improve reef health. Divers valued improvements in coral cover (ranging from under 25% to over 75%), visibility (25-100ft), and reduced fish decline (5%-35%) through an increased annual user fee. Prior to completing the survey divers were provided with information cards explaining that coral in Bonaire is declining, and that sediment run-off is one of the causes of this decline. Cards (Appendix C) explained that one way to reduce sediment run-off would be to control grazing by invasive species, though lethal control or restricting movements. Participants were then asked if they would be willing to pay an increased fee in principle to fund this action, before moving on to the choice experiment.

Within the choice experiment we did not include details of other, more direct, actions which could also improve coral cover. Bonaire already has a well established marine conservation program, the main body of which is run by STINAPA, the national park authority, and is funded by the existing dive fee of $25. Actions funded by this fee includes a lionfish hunting program, patrols to enforce fishing restrictions, and coral reef monitoring, and therefore would continue to be funded alongside any terrestrial conservation actions. As such the willingness to pay estimates presented here are applicable only to control of invasive grazing species, and cannot be used to trade off a broader set of alternative options for coral reef conservation.

Divers were sampled using a convenience sampling strategy, as no central record of divers exists to enable random sampling. Divers were approached at shore-accessible dive sites, and at dive centres. Sample size was 299, with a response rate of 72%. Analysis using latent class modelling, which groups respondents into 'classes' with similar preferences, indicated three classes in terms of preferences for coral reef improvements. We found a positive preference for reef health improvements for the majority of respondents.

Model estimates from the latent class analysis were used to estimate willingness to pay for the improvements in coral cover predicted to arise from each grazer control strategy, assuming a linear relationship between willingness to pay and coral cover[1]. These improvements fell within the range of attribute levels presented in the choice experiment. Coral cover coefficients were divided by cost coefficients to estimate willingness to pay for each percentage point improvement in coral cover. Maximum willingness to pay of divers for potential environmental improvements was calculated by multiplying this willingness to pay for a single percentage point improvement by predicted improvements arising from each control strategy (estimated coral cover from models above, minus 46% as estimated mean current coral cover) (Appendix C). For full explanation of methods and results see (Roberts et al., 2017a).

To provide insight on what financial resources this stated willingness to pay could provide for environmental management measures, individual willingness to pay for any specific predicted environmental quality change was multiplied by the number of dive tags sold annually (2015 estimate: 89,460 (Statistics Netherlands, 2015; STINAPA Bonaire, 2010), minus the $25 fee already paid to run the marine park. The current $25 fee was removed as it is already allocated to existing actions, such as marine park patrols, and therefore would not be available to cover costs of donkey control. The variability in funding potential was illustrated through repeating

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[1] To assess linearity in the relationship between coral cover and willingness to pay this model was also estimated using dummy variables, results present in Table 4, Appendix C. These results show a positive willingness to pay for very high coral cover in class one, and all increases in coral cover for class two. Because the willingness to pay for improved coral cover estimated from these models was higher than that estimated using the linear model, the results of the linear model are used throughout the study, as the most conservative estimate.

estimates using the upper and lower 95% confidence intervals of preference parameters for improvements to coral cover. We note that, should the environmental improvements represented in the choice experiment actually occur, then the number of dive visitors per year could easily rise: we have not tried to quantify this effect in our calculations of available funding.

2.7 Social acceptability of control options

Though social acceptability of control options is central to selecting the most appropriate action, the potentially sensitive nature of controlling grazing species on Bonaire meant that conducting such as survey without an established plan for moving forward with control risked damaging future control efforts. Therefore the social acceptability survey described here is designed only to provide a very broad overview of acceptability, and a full survey would be required as part of any donkey management put in place.

Social acceptability of grazer control options were estimated through scores assigned by five experts in invasive species control on Bonaire (Bonaire Ministry of Economic Affairs; Bonaire Department of Nature and the Environment; Echo; and the lead author of this study). Experts scored each strategy, and no grazer control, for social acceptability to five local stakeholders (Conservation NGO; Government; Goat farmer; Pro-donkey group; and tour organisers), from 0 to 2:

0 – This group has no opposition to this strategy;

1 – This group has some opposition to this strategy which must be taken into account, but the project could feasibly commence within the next 6 months;

2 – This group has large opposition to this strategy, which would prevent the project from beginning within the next 6 months.

Scores for each strategy were taken as the mean.

3. Results

Full donkey eradication was predicted to improve median ground cover from the current estimate of 4% to 18%, compared to an estimate of 14% for fencing (lower estimate: 13%; upper estimate: 15%, likely underestimate as do not include impacts of excluding goats and pigs) (Fig 3). Donkey control was estimated to improve median coral cover to 100% compared to cover of 46% estimated for median donkey density, while fencing predicted increases in coral cover to 85% (Fig 4). These estimates all lie within the range of ground and coral cover recorded on Bonaire (Min ground cover = 0%, max ground cover = 75%. Min coral cover = Under 25%, max coral cover = Over 75%). Donkey control impacts exceeded the maximum possible values for coral cover, therefore figures present only those impacts between 0 and 100%. To account for uncertainty in model estimates relationships were also considered using the upper and lower bounds of donkey density estimates.

The costs of fencing for the total area designated for nature (1,208ha) was estimated at $1,120,378 (NPV, 2% discount rate over 10 years), with an estimated lifetime of ten years before replacement.

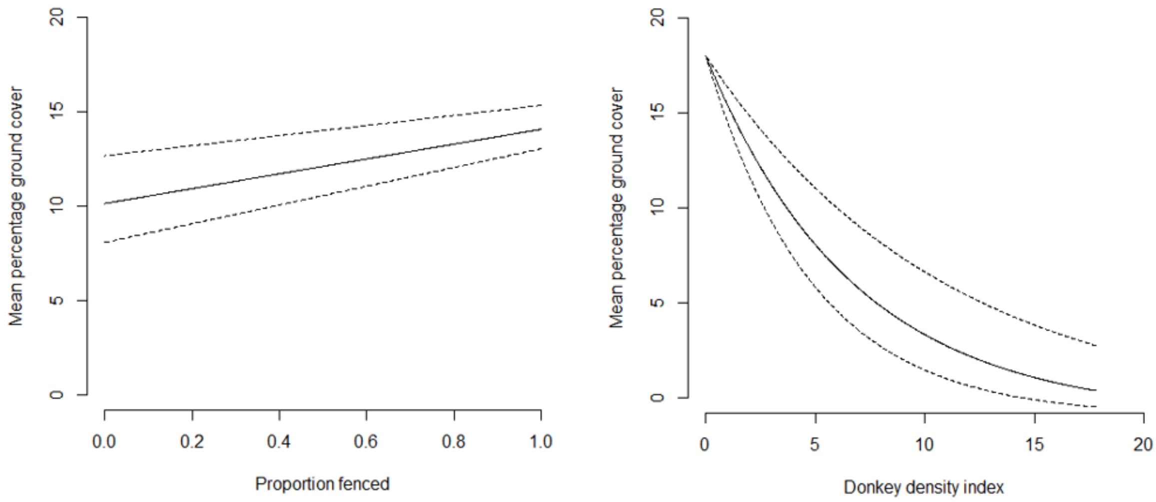

Fig 3 Ground cover change with alternative grazer control measures. Left: Fencing of nature areas; Right: Removal of donkeys. Dashed lines show estimates using lower and upper bounds of donkey densities. Median donkey density = 3.6, max donkey density 17. Current proportion fenced <0.01.

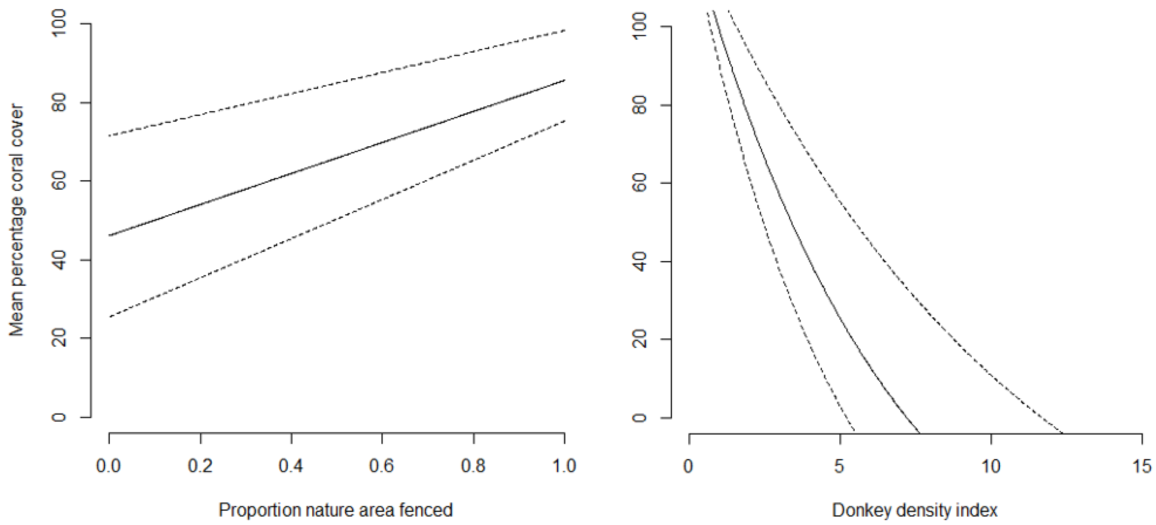

Fig 4 Changes in coral cover with alternative grazer control strategies. Left: Fencing, Right: Donkey control. Dashed lines show estimates using upper and lower estimates of ground cover**.** Median donkey density = 3.6, max donkey density = 17. Current proportion fenced <0.01.

To estimate eradication costs, six papers detailing the costs of eleven eradications were identified (Holmes et al., 2015; Martins et al., 2006; McCann and Garcelon, 2008; Melstrom, 2014; Ramsey et al., 2009). We considered only ungulate eradications, within wooded areas. Ten of the eradications were on true islands (Cruz et al., 2009; Holmes et al., 2015; Martins et al., 2006; Massei et al., 2011; Melstrom, 2014), and one within a fenced area (McCann and Garcelon, 2008). Eradication costs ranged from $10/ha USD2015 (Cruz et al., 2009) to $1,353/ha USD2015 (Holmes et al., 2015) (Appendix D) For this analysis, cost estimates calculated from the median value ($118/ha), and lower and upper quantile ($30.50/ha and $174/ha) were used. Total eradication of donkeys (goat and pig eradications were not costed) was estimated to cost $3.5 million (lower estimate: $0.8m; upper estimate: $5.1m). For Bonaire-specific estimates, costs (NPV, 2% discount rate over 10 years) ranged from $8.1 million for eradication including two months helicopter use and six months of monitoring, to $11.8 million for ground hunting only and 24 months of monitoring (Appendix D). Given the highly context specific nature of such cost estimates (de Brooke et al., 2007; Donlan and Wilcox, 2007; Martins et al., 2006), and the preference for overestimating, rather than underestimating costs, we have only use the expert estimated costs for further analysis. Although costs calculated in such a way do not allow for uncertainties to be quantified, in each case the median cost estimates as well as the lower and upper estimates have been included, to enable comparison across the range of likely costs.

From the latent class modelling results for the choice experiment undertaken with divers, mean maximum willingness to pay for class one (latent class share: 0.66, Appendix C for reef recovery arising from fencing (85% coral cover), when compared to predicted cover with median donkey density (46% coral cover), was estimated at $107.76/individual/year (lower bound: $82.11/individual/year; upper bound: $128.29/individual/year). Mean maximum willingness to pay for donkey removal (for a predicted improvement to 100% coral cover), was estimated at $149.21/individual/year (lower bound: $120.79/individual/year; upper bound: $177.00/individual/year). These estimates presume a linear relationship between willingness to pay and coral cover, following visual assessment of the results. Estimates have not been extrapolated beyond the levels presented within the survey. It is estimated 89,460 dive tags were sold in 2015, when this is multiplied by individual willingness to pay for improvements seen with fencing, funds raised (NPV, 2% discount rate over 10 years) are estimated at $8,832,588 ($6,730,176 - $10,515,337), exceeding estimated costs of fencing. Funds raised for donkey control across divers was estimated at $12,230,053 ($9,900,597 - $14,507,870), exceeding the costs of full eradication. To account for uncertainties within these estimates we also include the lower and upper bounds, with the estimated willingness to pay from the lower bound exceeding the cost of fencing, but being lower by ~$2 million for the highest estimated cost of eradication (Fig 5).

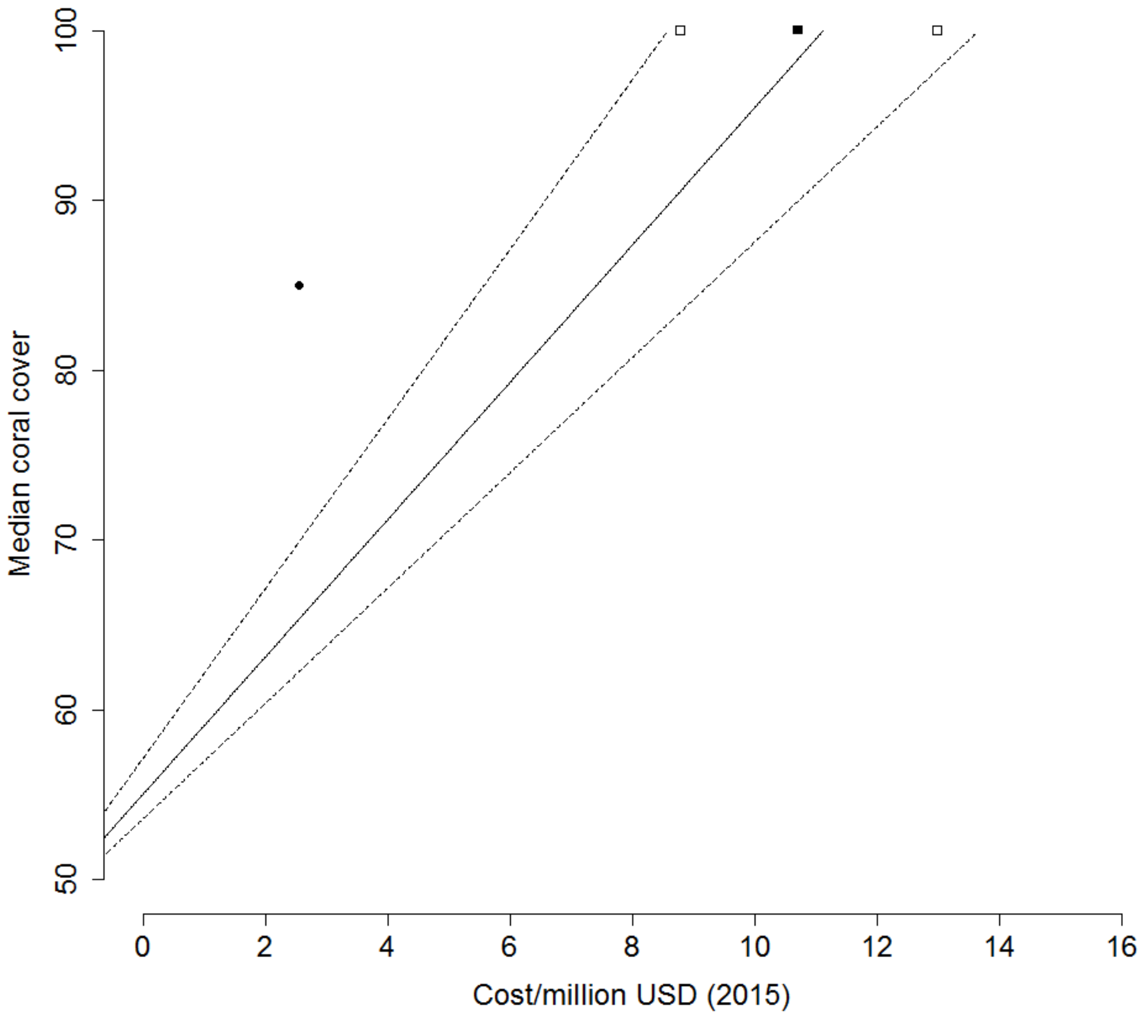

Fig 5 Estimated income from a user fee on divers for increasing levels of coral cover (line), related to costs of alternative conservation measures, and their predicted impacts on coral cover (points). Solid line shows mean willingness to pay, whilst dotted lines show higher and lower confidence intervals of the coral coefficient, as estimated from the choice experiment. Circle = fencing, square = donkey eradication. Filled symbols represent mean cost estimates, with empty symbols representing upper (ground hunting and 24 months' monitoring) and lower cost (helicopter and ground hunting, and six months' monitoring) estimates. Note the points and lines represent different data, and are not dependent on one another.

Fencing of nature areas had a mean social acceptability score of 0.52 (SE= 0.12, 0= fully acceptable, 2=unacceptable), while donkey control had a score of 0.95 (SE= 0.14). Taking no action had a mean score of

0.72 (SE=0.15). All options, including no action, received a score of 2 for at least one stakeholder from at least one expert.

4. Discussion

Using the island of Bonaire as a case study, we demonstrate the incorporation of ecological, economic and social domains for prioritising conservation actions for donkey control. Though eradication provides the largest ecological benefits, initial assessments suggest that lethal control is unlikely to be successful due to resistance by local stakeholders. Incorporation of economic costs shows that, in the short term, control of donkeys through exclusion areas created through fencing is most cost effective and is covered by the lowest estimate of diver willingness to pay. However, within 30 to 50 years, eradication would be more cost-effective, when considering only impacts from donkey control, though these costs exceed the lowest estimates of funds from a diver fee.

Including these four strands (conservation effectiveness; economic costs; willingness to pay of beneficiaries, and social acceptance) into decision making we can make the recommendation for fencing of nature areas as a short-term program for donkey control on Bonaire. Long term donkey control will require undertaking a full social program, including a full survey to understand social barriers, and working to improve social acceptability of lethal control. Considered from only an ecological standpoint this action would appear to have lower ecological impacts while from an economic standpoint it is also less cost effective than eradication over the long term. However though we were able to only broadly assess social acceptability of actions, the results from our expert survey indicate that eradication would have a low chance of success, and therefore in reality likely result in less ecological improvement. The incorporation of a user fee illustrates that a funding mechanism for such a program exists, which improves the potential for planning to move into action, and for the program to be sustained over the long term (Whitelaw et al., 2014).

When considering this recommendation for fencing it is important to note that our calculations consider only those impacts from donkey control, the additional benefits of excluding goats and pigs which would arise from fencing are not estimated. This is due to a limitation in the models used to estimate grazer impacts on vegetation, which relies on natural spatial variation to estimate impacts on vegetation. Though our models do estimate donkey impacts in the presence of goats and pigs, suggesting therefore that some additive impact is present (areas with no donkeys have higher ground vegetation cover despite the presence of goats and pigs), we are not able to consider the interactions of the three grazing species. With this in mind our estimates of the impacts of eradication may be overestimated, as we cannot account for increased grazing by goats or pigs. Fencing would therefore also present the opportunity to further refine our understanding of the impacts of grazing species on Bonaire, to inform future control actions. Additionally fencing will provide the opportunity to identify any unexpected ecosystem responses from removal of grazers, such as increases in invasive plant species, and enable plans to be put into place to address such issues prior to further eradication or control.

Further limitations of our models are also apparent when considering the estimated improvements from donkey control, which are estimated to exceed 100%. This illustrates the importance of considering such models as guidelines only, and the challenges of estimating models in situ, with multiple interacting factors. Though we are confident larger improvements would be observed with donkey control than fencing, continued monitoring would be needed to refine estimates of true improvements to coral cover.

Though it is suggested that inclusion of even rough cost estimates greatly improves prioritisation of conservation actions (Boyd et al., 2015), prioritisation remains highly problematic due to the scarcity of reporting of eradication costs. We identified only six studies, reporting the costs of eleven ungulate eradications (Cruz et al., 2009; Holmes et al., 2015; Martins et al., 2006; Massei et al., 2011; McCann and Garcelon, 2008; Melstrom, 2014), with further challenges presented due to lack of reporting on time scales; habitat types; or number of individuals removed. Martins et al. (2006) identify island area and taxon group as significant in determining eradication costs. Median island area in studies considered was 5,683ha (500ha – 464,000ha), compared to Bonaire size 2,940ha. Larger islands are predicted to have lower per ha costs (Martins et al., 2006), therefore costs reported here may underestimate eradication costs for Bonaire. This is supported by expert estimation of costs, which estimated costs between $8,773,831 to $12,968,945 for full donkey eradication on Bonaire, more than twice that estimated from the literature. Such differences indicate the importance of estimating costs in the local context, as well supported within the literature (de Brooke et al., 2007; Donlan and Wilcox, 2007; Martins et al., 2006). While these costs are valuable for initial prioritisation they refer to broad costs for hypothetical projects, that is they do not take account of variations in spatial and temporal design of control actions, which are known to impact cost-effectiveness of invasive species control (Baker and Bode, 2016). Further refinement of these costs would therefore be valuable to design any final control program.

Willingness to pay for grazer control actions to improve reef health was positive for the majority of divers responding to our choice experiment study, and exceeded the estimated costs of fencing and donkey eradication. However a minority of divers were not willing to pay an increased fee for reef health improvements achieved through terrestrial conservation, and therefore the risk of pushing these divers to alternative locations (and thus losing their expenditures on the island) must be considered when increasing fees on all divers. One response to this diversity in willingness to pay for conservation policy is to differentiate user fees according to variations in preferences. Despite the shore accessibility of Bonaire's dive site preventing the setting of site-specific fees, lower fee options could be offered for a restricted numbers of dives, or for family groups. Though it is useful to account for preference variations, analysis also indicates that those divers with a the highest positive willingness to pay are those most likely to return within the next five years. In calculating total funds raised no account has been made of increases in visitors arising from improved coral cover. Divers lost through increased fees may therefore have little impact on overall diver numbers, and thus on local incomes. Our survey also only considered willingness to pay for coral reef improvements arising from terrestrial grazing control. Willingness to pay for improvements arising from other actions, such as reducing diver numbers or putting restrictions on cruise ships, may therefore vary. Such actions would also be expected to have a more direct impact on the coral reef, and therefore preferences between actions should be considered where coral reef improvements are the sole project aim.

Our study considered only broad understanding of the social acceptability of donkey control, as the sensitive nature of control meant that a full social survey would have been detrimental to future conservation work. However, even at this broad level, considering only expert opinion, it is apparent that lethal control would be precluded by social opposition at this time. The higher social acceptability and lower costs of fencing, despite consequent lower levels of ecological improvement, indicate that fencing of nature areas presents the best option for coral reef restoration through donkey control on Bonaire in the immediate future. However, it is important to note that fencing is expected to have a life of only ten years, compared to indefinite length of control for donkey eradication. Within 30 to 50 years, therefore, eradication becomes the most cost-effective option. Long term donkey control on Bonaire would therefore benefit from increased understanding of the social barriers present for lethal control, and targeted campaigns to improve acceptability for such programs. Further gains would be seen with additional studies to understand the impacts of goats and pigs. Finally the models presented here and in Roberts et al 2017, 2017a and 2017b are based on the current ecological state of the system, and contain inherent uncertainty surrounding the ecological, economic, and social data. Throughout data analysis and modelling upper and lower bounds of estimates have been incorporated, and for the recommended action of fencing highest costs and lowest ecological outcomes still fall within the lowest willingness to pay of divers, suggesting that even under the least favourable outcome, fencing remains a viable option for control donkey populations on Bonaire. However given the dynamic nature of ecosystem restoration, particularly when working across ecosystem boundaries, as well as the impact this has on consumer preferences, the management recommendations are suitable only for near-term decision making. For effective management of grazing species on Bonaire management, plans should be updated with changing situations as control actions are rolled out over time.

5. Conclusions

Prioritisation of conservation actions is vital in achieving conservation goals. Previous studies have highlighted that ecological outcomes of conservation can be improved through considering impacts across ecosystem boundaries (Klein et al., 2014; Maina et al., 2013; Mateos-Molina et al., 2015), accounting for economic costs (Boyd et al., 2015), considering social concerns (Guerrero et al., 2010; McLeod et al., 2015), and become self-financing (Whitelaw et al., 2014). Here we have illustrated an integrated application for considering all of these issues, in the context of donkey control on an island. While ecological outcomes are central to environmental conservation, the option with the highest potential for ecological success is only optimum as long as it is cost effective, socially acceptable, and connected to funding. Achieving significant gains in biodiversity conservation requires that decision makers are able to incorporate all of these considerations into prioritisation of alternative actions.

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Appendix A

Table 1 Results from General Linear Model (log transformed data) investigating effects of grazing on ground cover. The full model (ground cover ~ goat density + dry season donkey density + wet season donkey density + pig presence + land use + landscape type + soil + goat density: dry season donkey density + goat density: wet season donkey density + wet season donkey density: dry season donkey density, n=86) is presented alongside the representative model (ground cover ~ goat density + dry season donkey density + landscape type + soil class, n=86). Full model deviance = 110.8, df=68, representative model deviance = 128.8, df=78. Full model intercept set to landscape type: higher terrace; soil type: sand and land use: agriculture. Best model intercept set to landscape type: higher terrace; soil type: sand. Values log transformed.

Ground cover								
Full model AIC = 303.8					Representative model AIC = 296.8			
	Est.	SE	t	P	Est.	SE	t	P
(Intercept)	1.79	1.03	1.73	0.09	**3.00**	**0.67**	**4.48**	**<0.01**
Goat density	-501.99	316.39	-1.59	0.12				
Dry season donkey density	-0.12	0.10	-1.18	0.24	**-0.15**	**0.06**	**-2.61**	**0.01**
Wet season donkey density	0.06	0.12	0.51	0.61				
Pig presence	-0.40	0.48	-0.83	0.41				
Nature area	**1.10**	**0.51**	**2.14**	**0.04**				
National Park	0.85	0.74	1.16	0.25				
Open use area	0.97	0.58	1.67	0.10				
Urban use area	-0.67	1.33	-0.50	0.62				
Lower terrace	-1.28	0.77	-1.66	0.10	**-1.28**	**0.65**	**-1.96**	**0.05**
Middle terrace	0.00	0.64	0.00	1.00	-0.46	0.57	-0.81	0.42
Undulating landscape	-0.30	0.64	-0.48	0.63	**-0.95**	**0.49**	**-1.95**	**0.05**

Loam soil	-0.35	0.58	-0.60	0.55	-0.47	0.53	-0.89	0.38
Rocky soil	0.27	0.62	0.44	0.66	0.36	0.56	0.64	0.52
Terraced soil	0.87	0.62	1.40	0.17	**1.25**	**0.58**	**2.14**	**0.04**
Goat density : Dry season donkey density	164.62	138.61	1.19	0.24				
Goat density : Wet season donkey density	-45.19	82.13	-0.55	0.58				
Dry season donkey density: Wet season donkey density	0.00	0.02	-0.10	0.92				

Appendix B

Table 2 Results from General Linear Model investigating effects of watershed vegetation on mean coral cover deeper than 5m. n=49. Full model deviance = 17.39, df=37, representative model deviance = 19.08, df=41. Intercept for full model set to soil type: loam; shore access: no; salina: no' land use: nature. Representative model: shore access: no; land use: nature. Significant terms in bold. Table from (Roberts et al., 2017b)

	Full Model AIC: 114.3				Representative Model AIC: 110.85			
	Est.	SE	t	P	Est.	SE	t	P
Intercept	**4.85**	**1.25**	**3.88**	**<0.01**	**3.09**	**0.44**	**6.99**	**<0.01**
Tree biomass index	**-1.43**	**0.41**	**-3.53**	**<0.01**	**-0.77**	**0.15**	**-5.21**	**<0.01**
Percentage ground cover	-0.02	0.02	-1.33	0.19	0.00	0.01	-0.27	0.79
Shore accessible	**-0.73**	**0.32**	**-2.27**	**0.03**	**-0.71**	**0.30**	**-2.35**	**0.02**
Distance from town	**0.63×10^{-4}**	**0.26×10^{-4}**	**2.47**	**0.02**	**0.66×10^{-4}**	**0.23×10^{-4}**	**2.84**	**0.01**
Rocky soil	-1.67	0.91	-1.83	0.07				
Terrace soil	-1.73	1.14	-1.51	0.14				
Terrace/rocky soils	-2.00	1.41	-1.42	0.17				
Salina present	1.50	0.83	1.81	0.08	0.78	0.46	1.70	0.10
Slope	2.14	7.19	0.30	0.77				
Urban use	-1.88	1.68	-1.12	0.27	**-1.06**	**0.53**	**-2.00**	**0.05**
Tree biomass index : percentage ground cover	**0.11**	**0.03**	**3.51**	**<0.01**	**0.06**	**0.01**	**5.21**	**<0.01**

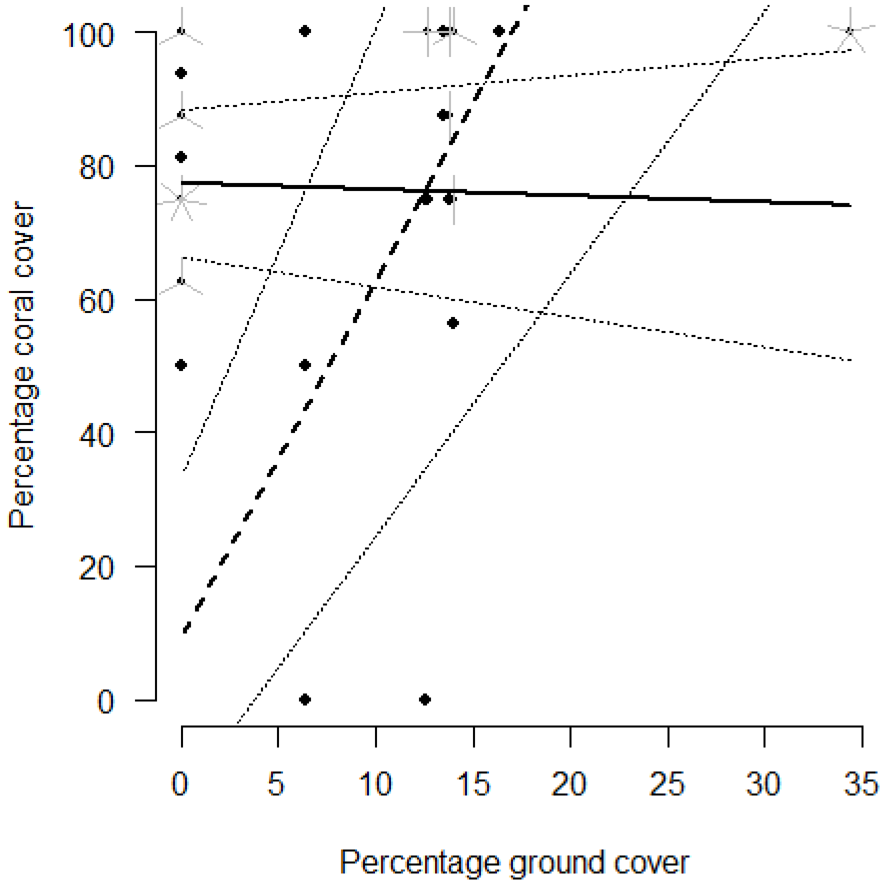

Fig 6. Change in deep coral cover with ground cover showing how this relationship was dependent on tree biomass. Dashed – Median tree biomass; Solid – Min tree biomass. Estimates with maximum tree biomass are not presented as these are not representative of the majority of locations on Bonaire. Dotted lines indicate upper and lower confidence intervals of ground cover impact. Originally presented in (Roberts et al., 2017b)

Appendix C

$$WTP = \left(\left(\frac{\beta_{Vis}}{\beta_{Cost}}\right) \times \Delta Vis \right) + \left(\left(\frac{\beta_{Coral}}{\beta_{Cost}}\right) \times \Delta Coral \right) + \left(\left(\frac{\beta_{Fish}}{\beta_{Cost}}\right) \times \Delta Fish \right)$$

β_{Vis} = Visibility preference coefficient (Table)

β_{Coral} = Coral preference coefficient (Table)

β_{Fish} = Fish preference coefficient (Table)

β_{Cost} = Cost preference coefficient (Table)

ΔVis = Change in visibility/m

ΔCoral = Percentage change in coral cover

ΔFish = Percentage change in fish abundance

Table 3 Results from latent class logit model on choice experiment data for SCUBA divers valuing coral reef attributes. Significant results in bold. This table has been summarised from data originally reported in Roberts et al. 2017a

	Class 1		Class 2		Class 3	
	Coef.	SE	Coef.	SE	Coef.	SE
Visibility	**0.023**	0.003	**0.021**	0.005	0.032	0.034
Coral cover	**0.021**	0.002	**0.018**	0.004	0.040	0.028
Reduced fish decline	**0.027**	0.005	0.002	0.009	-0.063	0.056
Cost	**-0.007**	0.003	**-0.058**	0.005	-0.141	0.081
Status quo	**-3.04**	0.5	**-2.31**	0.30	**2.91**	0.81
Return within 5 years	1.5		1.7		-	
Class share	0.65		0.20		0.16	

Table 4 Results from latent class logit model on choice experiment data for SCUBA divers valuing coral reef attributes, with coral cover dummy coded. Significant results in bold.

	Class 1		Class 2		Class 3	
	Coef.	SE	Coef.	SE	Coef.	SE
Visibility	**0.02**	0.003	**0.02**	0.005	0.03	0.04
Coral cover - Mid	-0.28	0.59	**1.03**	0.48	0.69	1.53
Coral cover - High	0.67	0.61	**1.62**	0.56	2.00	1.53
Coral cover – Very High	**1.36**	0.17	**1.49**	0.33	2.90	2.64
Reduced fish decline	**0.03**	0.005	0.001	0.009	-0.06	0.07
Cost	**-0.005**	0.17	**-0.06**	0.006	-0.14	0.09
Status quo	**-3.46**	0.18	**-1.92**	0.38	**2.92**	0.93
Class share	0.65		0.20		0.16	

Fig 7. Information cards presented to participants of the choice experiment to explain the connection between terrestrial grazing, sediment run-off and coral reef decline.

Bonaire is internationally renowned as a high quality SCUBA dive destination (SCUBA Diving Magazine, 2015). However, like coral reefs worldwide, the health of Bonaire's reef is declining over the long-term.

Studies carried out on Bonaire's reef by the University of Maine (Steneck and colleagues 2003-2013) have shown the number of young corals is falling, and the diversity of fish species is changing. This will reduce the quality of the coral reef for diving.

Soil run-off from land is one cause of reef health decline. On Bonaire this is increased due to grazing by introduced goats, donkeys and pigs.

Goats, donkeys and pigs were introduced to Bonaire by Spanish settlers, they are not native to the island. Grazing by these animals reduces plant numbers, meaning that there are fewer roots to hold the soil, and it is washed onto the reef.

Increased soil on the reef reduces the number of young corals. In time this will lead to reduced coral cover and fish diversity. Increased soil in the water also reduces visibility for divers.

One way to maintain the health of Bonaire's coral reef is therefore to reduce grazing. This could be done by:

- Restricting movements of grazing animals;
- Reducing the number of grazing animals on Bonaire;
- Restricting where goat farmers can graze their goats.

To maintain the reef requires funding. You already pay an annual nature (dive tag) fee of $25 to STINAPA, which is used for the running of the Bonaire National Marine Park. This study is to find out if you would be willing to pay a higher fee in the future, to be used to reduce grazing. This fee would be collected at the same time as the current nature (dive tag) fee, but would be administered by a new non-governmental organisation. The fee would be guaranteed to be used for this purpose.

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The following questions will present you with a choice of three dive sites under different management conditions:

- The first two dive sites show diving conditions where grazing has been reduced
- The final dive site shows diving conditions where grazing has been allowed to continue

In each round you will be asked to choose which of the three dive sites you would like to visit. You should assume that the sites are identical except in the ways presented on the card.

Each site has a different annual fee associated with it. Remember to pay close attention to the fee, and take into account the cost of your holiday, and other economic constraints before making a decision. If the prices of the dive sites with management are too high, choose the option with no management.

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Appendix D

Table 5 Cost of eradication of goats and pigs from islands

Species	Methods	Island size/ha	Human population	Individuals removed	Cost/ha (USD2015)	Study
Goat	Helicopter Dogs Judas goat Corrals Ground hunting	58,465	No	79,000	$129	(Cruz et al., 2009)
Goat	Helicopter Dogs Judas goat Corrals Ground hunting	464,000	Yes	59,000	$10	(Cruz et al., 2009)
Goat	Ground hunting Corrals	520	No	Unknown	$1354	(Holmes et al., 2015)
Goat	Ground hunting Corrals	500	No	Unknown	$91	(Holmes et al., 2015)
Pig	Trapping Ground hunting Dogs Judas pigs	5,700	No	200	$120	(McCann and Garcelon, 2008)
Pig	Helicopter	25,000	No	5,036	$219	(Melstrom, 2014)

Pig	Ground hunting Trapping Judas pigs	5,666	No	Unknown	$118	(Massei et al., 2011)
Cattle	Ground hunting (primary, others unknown)	710	No	Unknown	$19	(Martins et al., 2006)
Goat	Unknown	3,230	No	Unknown	$13	(Martins et al., 2006)
Goat	Ground hunting (primary, others unknown)	14,600	Yes	Unknown	$42	(Martins et al., 2006)
Goat	Ground hunting (primary, others unknown)	2,938	No	Unknown	$242	(Martins et al., 2006)

Table 6 Estimated costs of donkey eradication on Bonaire for ground and aerial hunting, plus 6, 12, or 24 month monitoring period following eradication. Costs are shown per unit, as defined in row heading (e.g. day, month, or per equipment piece), and multiplied by number required for each option. Time taken for ground hunting without monitoring is 24 months, and aerial hunting without monitoring 14 months. This initial time is added to costs of 6, 12, or 24 month monitoring in each column. Costs in USD2015

	Cost per unit	Ground hunting			Helicopter		
		6 months	12 months	24 months	6 months	12 months	24 months
Professional hunter /day	320	4454400	4915200	5836800	3686400	4147200	5068800
Local hunter /day	160	1113600	1228800	1459200	921600	1036800	1267200
Housing /hunter /month	800	950400	1056000	1267200	598400	704000	915200
Ammunition	1500	1500	1500	1500	1500	1500	1500
GPS collar	3000	90000	90000	90000	90000	90000	90000
Fitting GPS collar	1000	30000	30000	30000	30000	30000	30000
Corral	2250	2250	2250	2250	2250	2250	2250
Firearms /unit	2000	72000	72000	72000	72000	72000	72000
Permit /firearm	2000	72000	72000	72000	72000	72000	72000
Dog and handler /day	400	1856000	2048000	2432000	1536000	1728000	2112000
Management /day	480	307200	364800	480000	259200	316800	432000

Transport /km	0.3	5760	6840	9000	4860	5940	8100
Vehicle	1500	3000	3000	3000	3000	3000	3000
Camera Traps	700	35000	35000	35000	35000	35000	35000
Helicopter /hour	2000	0	0	0	640000	640000	640000
Pilot /day	600	0	0	0	24000	24000	24000
Admin		899311	992539	1178995	797621	890849	1077305
TOTAL		9892421	10917929	12968945	8773831	9799339	11850355

Table 7 Breakdown of costs for removal phase of eradication by ground control only. 24 month long project, not including monitoring of success.

Ground hunting - Removal phase	
24 Professional hunters, 24 months full time, $40/hour	$3,993,600.00
12 Local hunters, 24 months full time, $20/hour	$998,400.00
Accommodation, 36 hunters, 8 dog handlers, 24 months	$844,800.00
Ammunition, 3000 bullets (3 times estimated donkey population)	$1,500.00
30 GPS collars, including VHF transmitters, for Judas donkeys	$90,000.00
Fitting GPS collar, including tranquiliser and trained personnel	$30,000.00
Corral, fence materials for single semi-permanent corral	$2,250.00
Firearms, 36 rifles of high power	$72,000.00
36 firearm permits over two years (approximate fee)	$72,000.00
8 dogs and handlers, 24 months full time, $50/hour	$1,664,000.00
Project manager, 24 months full time, $60/hour	$249,600.00
Transport, estimated 30km/day, $0.3/km	$4,680.00
Vehicle, used pickup, price for acquiring on island	$3,000.00
Admin, 10% of project cost	$802,583.00
Total	$8,828,413.00

Table 8 Breakdown of costs for removal costs of eradication including 2 months aerial hunting and 14 months ground hunting, not including monitoring of success.

Ground hunting and helicopter - Removal phase	
24 Professional hunters, 14 months full time, $40/hour	$3,225,600.00
12 Local hunters, 14 months full time, $20/hour	$806,400.00
Accommodation, 36 hunters, 8 dog handlers, 14 months	$492,800.00
Ammunition, 3000 bullets (3 times estimated donkey population)	$1,500.00
30 GPS collars, including VHF transmitters, for Judas donkeys	$90,000.00
Fitting GPS collar, including tranquiliser and trained personnel	$30,000.00
Corral, fence materials for single semi-permanent corral	$2,250.00
Firearms, 36 rifles of high power	$72,000.00
36 firearm permits over 14 months (approximate fee)	$72,000.00
8 dogs and handlers, 14 months full time, $50/hour	$1,344,000.00
Project manager, 14 months full time, $60/hour	$201,600.00
Transport, estimated 30km/day, $0.3/km	$3,780.00
Vehicle, used pickup, price for acquiring on island	$3,000.00
Helicopter, full day for 2 months	$640,000.00
Pilot, full time, 2 months	$24,000.00
Admin, 10% of project cost	$700,893.00
Total	$7,709,823.00

Table 9 Breakdown of costs for 6 months monitoring post-eradication

6 months monitoring	
12 Professional hunters, 6 months half time, $40/hour	$460,800.00
6 Local hunters, 6 months half time, $20/hour	$115,200.00
Accommodation, 18 hunters, 4 dog handlers, 6 months	$105,600.00
4 dogs and handlers, 6 months half time, $50/hour	$192,000.00
Project manager, 6 months half time, $60/hour	$57,600.00
Transport, estimated 30km/day, $0.3/km	$1,080.00
50 Camera traps, Infrared, no glow, including batteries and memory cards	$35,000.00
Admin, 10% of project cost	$93,228.00
Total	$1,060,508.00

Table 10 Breakdown of costs for 12 months monitoring post-eradication

12 months monitoring	
12 Professional hunters, 12 months half time, $40/hour	$921,600.00
6 Local hunters, 12 months half time, $20/hour	$230,400.00
Accommodation, 18 hunters, 4 dog handlers, 12 months	$211,200.00
4 dogs and handlers, 12 months half time, $50/hour	$384,000.00
Project manager, 12 months half time, $60/hour	$115,200.00
Transport, estimated 30km/day, $0.3/km	$2,160.00
50 Camera traps, Infrared, no glow, including batteries and memory cards	$35,000.00
Admin, 10% of project cost	$186,456.00
Total	$2,086,016.00

Table 11 Breakdown of costs for 24 months of monitoring post-eradication

24 months monitoring	
12 Professional hunters, 24 months half time, \$40/hour	\$1,843,200.00
6 Local hunters, 24 months half time, \$20/hour	\$460,800.00
Accommodation, 18 hunters, 4 dog handlers, 24 months	\$422,400.00
4 dogs and handlers, 24 months half time, \$50/hour	\$768,000.00
Project manager, 24 months half time, \$60/hour	\$230,400.00
Transport, estimated 30km/day, \$0.3/km	\$4,320.00
50 Camera traps, Infrared, no glow, including batteries and memory cards	\$35,000.00
Admin, 10% of project cost	\$372,912.00
Total	\$4,137,032.00