Tourism Management 81 (2020) 104160

Contents lists available at ScienceDirect

Tourism Management

journal homepage: http://www.elsevier.com/locate/tourman

Monitoring tourists' specialisation and implementing adaptive governance is necessary to avoid failure of the wildlife tourism commons

Francesca Mancini [a,1,*], Ben Leyshon [b], Fiona Manson [c], George M. Coghill [d], David Lusseau [a]

[a] School of Biological Sciences, University of Aberdeen, Aberdeen, UK
[b] Scottish Natural Heritage, Fodderty Way, Dingwall Business Park, Dingwall, UK
[c] Scottish Natural Heritage, Battleby, Redgorton, Perth, UK
[d] School of Natural and Computing Sciences, University of Aberdeen, Aberdeen, UK

ARTICLE INFO

Keywords:
Common-pool resources
Simulation
Sustainability
Socioecological systems
Individual-based model
Wildlife watching
Adaptive governance
Tourist specialisation

ABSTRACT

Wildlife tourism (WT) is an important economic sector globally, which can sustain national and local economies. These activities have been reconceptualised as consumptive because of their impacts on the wildlife, and the problem of managing WT as a common-pool resource issue. We use an individual-based model to simulate the dynamics of a WT destination in different development phases. We then ask if any of the governance structures commonly proposed to solve common pool resource issues are appropriate to sustainably manage a WT destination during its development. The level of specialisation of tourists visiting a destination can influence both the exploitation of the wildlife and the socio-economic success of the industry, and no single governance structure leads to sustainability in every stage of a WT destination lifecycle. Given the dynamics of WT destinations, an adaptive governance framework is crucial to avoid wildlife depletion and economic failure of the industry.

1. Introduction

Nature recreation is becoming increasingly popular globally (Balmford et al., 2009, 2015). Wildlife recreation is a type of nature recreation that involves interactions with wildlife. Wildlife watching activities were initially welcomed by conservation and environmental organisations as good sustainable alternative use of wildlife compared to other recreational activities such as fishing or hunting (Tisdell & Wilson, 2002). However, in the last two decades, many studies have shown that wildlife watching can have behavioural and physiological impacts on the animals (Amo, López, & Martín, 2006; Beale & Monaghan, 2004; Christiansen, Rasmussen, & Lusseau, 2013; Frid & Dill, 2002; Lusseau, 2003; McClung, Seddon, Massaro, & Setiawan, 2004; Velando & Munilla, 2011), which can affect the individuals' survival and reproductive rates and result in population-level consequences (Bejder et al., 2006; Christiansen & Lusseau, 2015; McClung et al., 2004; Pirotta, New, Harwood, & Lusseau, 2014; Watson, Bolton, & Monaghan, 2014). The subject literature reports cases of both successful and unsuccessful governance of nature tourism systems. When managed successfully, sustainable nature tourism can alleviate poverty (Ferraro & Hanauer, 2014), stimulate development of infrastructure (Liu et al., 2012), create employment opportunities (Li, Jin, & Shi, 2018) and benefit wildlife conservation (R. C. Buckley, Castley, de Pegas, Mossaz, & Steven, 2012; Lindsey et al., 2014; Wilson, Hayward, & Wilson, 2017). However, when nature tourism systems fail it can lead to declines in wildlife population abundance (Lusseau & Bejder, 2007), reduced effectiveness of protected areas (Reed & Merenlender, 2008) and land-use conflicts with consequences for local populations (Sirima & Backman, 2013; Xi, Zhao, Ge, & Kong, 2014). This has led to a reconceptualization of wildlife tourism as a consumptive activity (James E.S. Higham, Bejder, Allen, Corkeron, & Lusseau, 2016; Meletis & Campbell, 2007) and the problem of how to manage it sustainably as a common pool resource issue (Briassoulis, 2002).

Hardin's (1968) paper introduced the concept of the "tragedy of the commons" to indicate the situation in which users of a common pool resource are trapped in a system of incentives that will encourage them to overexploit the resource and eventually collapse the socioecological system, unless the resource is managed by a central authority or under private property rights regimes. Since then, common pool resource research has documented cases of commons where users have been successful in self-organising and producing sustainable outcomes (Ostrom, 1990; Ostrom, Burger, Field, Norgaard, & Policansky, 1999;

* Corresponding author.
E-mail address: framan@ceh.ac.uk (F. Mancini).
[1] Present address: UK Centre for Ecology & Hydrology, Maclean Building, Crowmarsh Gifford, Wallingford, OX10 8BB, UK.

https://doi.org/10.1016/j.tourman.2020.104160
Received 27 April 2019; Received in revised form 22 March 2020; Accepted 18 May 2020
Available online 26 May 2020
0261-5177/© 2020 Elsevier Ltd. All rights reserved.

Ostrom, Janssen, & Anderies, 2007). We also have numerous examples of attempts to sustainably govern the commons that have failed (Acheson, 2006). Different governance structures have been proposed to manage common pool resources: private property (Lindsey et al., 2014; Muir-Leresche & Nelson, 2000; Wilson et al., 2017), government control (Lovejoy, 2006; Mayer et al., 2018), community-based management and co-management (Conley & Moote, 2003; Lamers, van der Duim, van Wijk, Nthiga, & Visseren-Hamakers, 2014; Sheppard, Moehrenschlager, Mcpherson, & Mason, 2010). Often, these governance solutions are advocated as panaceas, a single solution to every commons. But commons are complex systems and simple solutions are unlikely to be successful (Ostrom et al., 2007). Moving beyond panaceas requires us to navigate each single case study to find a sustainable solution.

Wildlife tourism and recreation systems can be described as made of four main components: the wildlife and the habitat, the tourists, the businesses that make up the tourism offer and the institutions and rules regulating the system. These subsystems are relatively separable, but interact in complex ways to produce outcomes at the system level, which then produce feedback that influences the individual subsystems (Ostrom, 2009). A number of variables have been identified as important to determine the outcomes of socioecological systems (Ostrom, 2009), such as the size and location of the resource system (the destination), the number and growth rate of the resource units (the wildlife) and the socioeconomic attributes of the users (tourists and tour operators). Measuring these variables in real systems provides insights into the social, economic and environmental outcomes of socioecological systems. Wildlife recreation destinations are dynamic, and, as the destination develops, it experiences substantial changes in these system properties that are important in determining if sustainability will be achieved (Ostrom, 2009). As a consequence, governance structures that used to be successful might eventually become inappropriate (Partelow & Nelson, 2018). Nearly 30 years ago, Duffus and Dearden published their conceptual model of non-consumptive wildlife-oriented recreation (Duffus & Dearden, 1990), a framework that brought together Butler's tourism area lifecycle (Butler, 1980), Bryan's tourist specialisation continuum (Bryan, 1977) and the concept of Limits of Acceptable Change – LAC - (Stankey, McCool, & Stokes, 1984). The temporal dynamics of a wildlife tourism destination (Fig. 1) can be described by following the change in the number of tourists visiting it through time. First, the destination goes through an exploration phase, where mostly specialist tourists start discovering the area. This phase is followed by a development phase, characterised by an exponential growth in the number of tourists, infrastructure development at the destination and a shift in tourist typology from specialists to a mixture of specialists and generalists. The last phase of the wildlife tourism destination lifecycle is the consolidation phase, when the number of tourists (mostly generalists) plateaus. Together with these changes in the social and economic dynamics of the wildlife tourism destination, effects on the environment also occur as the number of tourists increases and specialist wildlife watchers are displaced by more generalist tourists, who require more infrastructure and place greater pressure on the environment. After the consolidation phase there are three possible trajectories for the wildlife recreation destination: i) the industry can collapse because of a decline in attractiveness due to overcrowding and environmental degradation; ii) a stagnation phase, where numbers of visitors remain the same; iii) a period of rejuvenation, where the industry changes dramatically allowing a second period of growth (Catlin, Jones, & Jones, 2011; Duffus & Dearden, 1990). Since the publication of this conceptual framework, empirical research has attempted to identify the three stages of development in real tourism case studies and understand how to best manage the destination to minimise permanent effects on the environment and avoid the collapse of the tourism area (Catlin et al., 2011). However, management usually lags behind development and it is likely to intervene only after these effects have started to become obvious (Higham, 2007), and at that point some irreversible consequences might have already started to appear. An ultimate goal is to develop a mechanism to

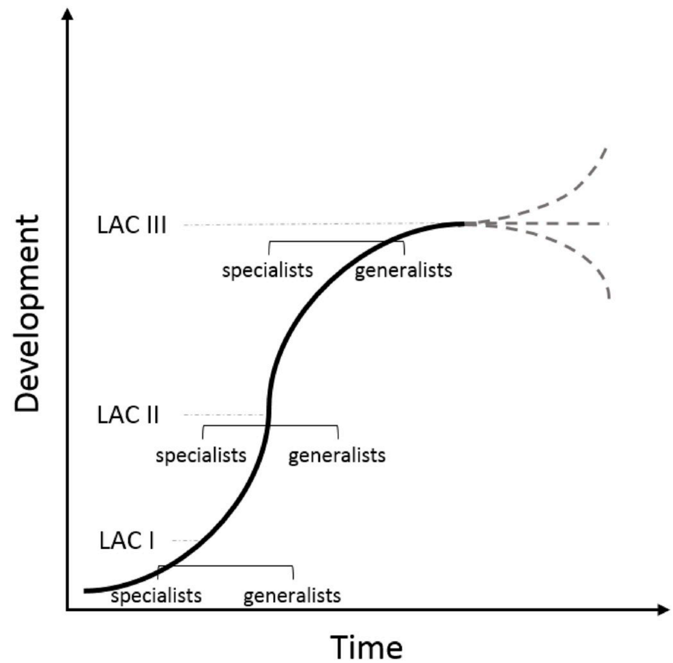

Fig. 1. The wildlife tourism destination lifecycle. During the development of the destination the level of specialisation of the tourists changes and the destination crosses three thresholds of Limits of Acceptable Change (LAC). Adapted from Duffus and Dearden (1990).

set up institutions and governance structures during the destination's initial phase that can ensure tourism remains sustainable by either avoiding collapse, or finding a stable state in which the destination can remain economically viable without damaging its social and environmental capitals.

Here we aim to investigate the institutions and governance structures that can result in socioeconomically and ecologically sustainable wildlife recreation operations at different stages of the tourism destination life cycle. We define a destination as an area where a number of wildlife watching operations exploit the same wildlife population (Center for Responsible Travel, 2014; Hughes, 2001; Semeniuk, Haider, Cooper, & Rothley, 2010). We build an individual based model (DeAngelis & Mooij, 2005) to simulate a generic wildlife watching destination (Fig. 2), with tourists, tour operators and wildlife agents (Pirotta & Lusseau, 2015) with the aim to determine how changes in the characteristics of tourists, their phenotype thereafter, can influence the sustainability of the destination. Individual-based models are a useful framework to study complex systems as they can show how system level properties emerge from the adaptive behaviour of individuals as well as how the system affects individuals (Railsback, 2001). As we saw, tourist phenotype changes drastically during the life cycle of a destination, yet the effect of these changes in customer phenotype on socioecological sustainability is largely unknown. We test how different tourists' characteristics and tourism volume trends influence the economic and ecological dynamics of a wildlife watching socioecological system and which governance structure is more likely to achieve a viable industry and the persistence of the targeted wildlife population.

2. Methods

2.1. Overview

The model has three main entities: the tourists, the tour operators and the wildlife (Fig. 2). We tested four different governance scenarios: voluntary code of conduct, licensing, user group governance and co-management. For each of these scenarios we simulated socioeconomic

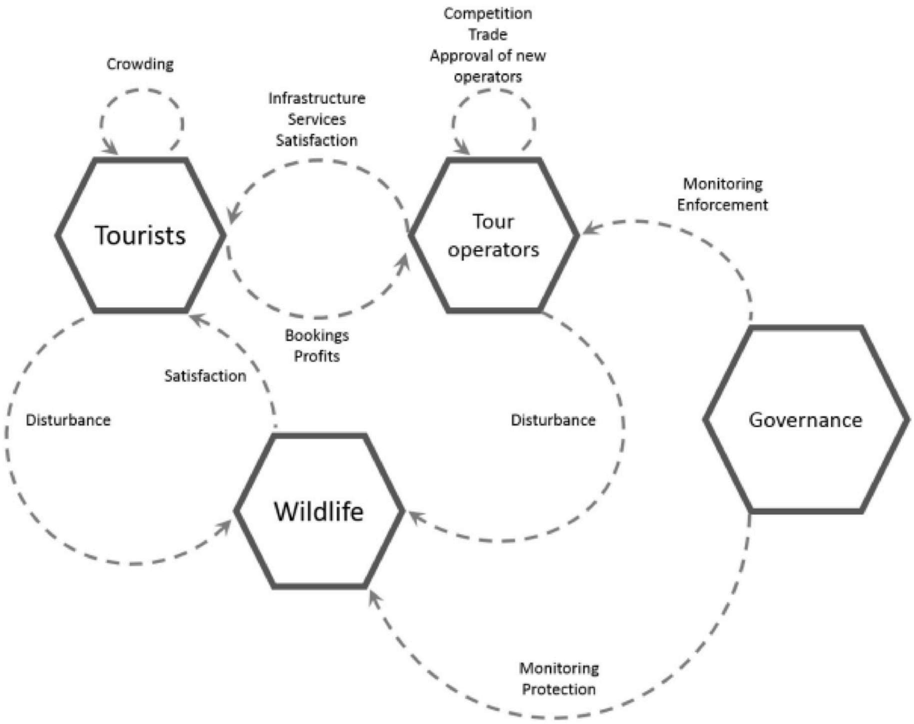

Fig. 2. Conceptual diagram of the individual based model. Hexagons represents the main entities in the model and dashed lines represent their relationships. The diagram is a representation of the structure of the model, which in turn is only a simplification of a tourism socioecological systems.

and ecological dynamics of the wildlife tourism destination, varying two parameters: type of tourists (3 values: mostly specialists, mostly generalists and mixed) and trend in demand (3 values: increasing, decreasing and stable number of tourists).

At the beginning of every year, the maximum amount of time spent with animals above which there starts to be a relevant effect on the population is calculated together with the probability of encountering the animals (Pirotta & Lusseau, 2015). From this maximum amount of time that the animals can sustainably spend with the operators, individual "time quotas" are calculated for each operator by dividing the total time equally between the active operators. Every year a population of tourists is initialised, from which every day a number of tourists is sampled. These tourists will try to book a tour with a tour operator on that day. The operators run one tour per day and they only run the tour if they received enough bookings to cover all the costs of running the tour. For the operators that will run a tour on that day, the model stochastically determines how many encounters with the animals they will have and how long they will last. If the cumulative encounter time during a tour is higher than the maximum time allowed per tour, the operator makes the decision whether to cooperate and only spend the time they are allowed with the animals, or to defect and spend all the time that is available with the animals. At the end of the tour the model calculates how satisfied the tourists are with their tour. The tour operators will then update their rating using the mean of their daily tourist satisfaction. In some scenarios tour operators can trade their wildlife time allowance (tradeable wildlife allowance – TWA). When a tour operator at the end of the day has spent all the time they were allowed with the animals, they will try to buy some extra TWA from tour operators that did not spend much of their allowed time. This is repeated for 365 days every year.

At the end of the year, in some scenarios, tour operators defecting from the rules set by the governance structure face the possibility of being fined. If a defecting tour operator is detected, 1000 money units are subtracted from their profits. At the end of the year the tour operators also decide on investments for the next year. The operators can make two types of investments: infrastructure or services. Tour operators also update their tour ticket price as a function of the demand: supply ratio. If a tour operator's profits have been 0 for the past 3 years the operator retires. New tour operators can start every year or every 6 years, depending on the governance scenario, and the tour operators or other governance institutions are responsible to decide whether new operators can start or not. At the end of every year, the cumulative effect of the tourism activities on the wildlife population is calculated.

We ran 5 simulations per combination of parameters (9×5 per governance scenario). The time horizon of the model is 50 years and the time step is one day. At the end of every year we collected simulation results on the effect of the tourism activity on the wildlife, the income of the tour operators as well as their decisions (both investments and defection). We also retained the volume of tourists carried by the destination (the number of bookings made), the tour ticket prices and the rating scores of the tour operators.

In the following sections we describe the different components of the model in more detail. We describe the model following the ODD protocol (Grimm et al., 2010) in Supplementary text.

2.2. Governance scenarios

Property is mainly considered as owned and affected by private individuals, communities or governments (Acheson, 2006; Hoffmann, 2013). Using this basic principle we design the model to test four different property rights regimes (Mancini, Coghill, & Lusseau, 2017) and their ability to sustainably manage a wildlife watching tourism system in every stage of Duffus and Dearden's conceptual framework (Duffus & Dearden, 1990). The first scenario describes a situation where property rights are not assigned, an open access regime where only a voluntary code of conduct is implemented. The second scenario simulates a top-down management of the wildlife tourism industry, where the government has property rights over the wildlife (or natural area) and releases licences to a number of tour operators. In the third scenario, property rights are shared between the tour operators in a community management scenario, where the users take responsibility to manage the resource. In the last scenario we simulate a co-management governance structure, where the tour operators still retain property rights but delegate the monitoring and enforcement of regulations to an external party, which could be either the government or an external private agency.

Independently of the governance scenario, part of the management process is the estimation of the maximum amount of time that tour operators can spend with the wildlife without having an impact of the population. This is estimated based on the effect that tourism had on the wildlife in the previous year:

$$max_y = max_{y-1} * \left(1.01 - effecty_{y-1}\right) \tag{1}$$

where max_{y-1} is the maximum time that was allowed with the wildlife the previous year, $effect_{y-1}$ is the effect tourism had on the growth rate of the population the previous year and 1.01 is the growth rate of the wildlife population (1% per annum, a typical growth rate for slow reproducing, long-lived species which are typically the focus of wildlife tourism).

In the next sections we introduce each scenario separately, highlighting the differences and similarities between them.

Code of conduct – This is the base scenario. In this governance structure tourism activities are not regulated and the only management tool used is a voluntary code of conduct, which suggests that the operators should spend a maximum of 10 min per encounter with the animals. We assume that all tour operators respect this code of conduct and we use this assumption in all the scenarios. Every year a new tour operator can start in the destination, with a probability given by the ratio between the demand (number of total bookings) and the supply (the combined capacity of all tour operators).

Licensing – In this scenario, tour operators need to obtain a license from the government to be able to use the wildlife. The government is in charge of calculating how much time the operators should be allowed to spend with the animals, and a fine is implemented for tour operators who defect. The fine is equal to 1000 money units and is enforced with a probability of detecting defectors of 0.5. This low probability reflects a high uncertainty in the monitoring process carried out by a central authority (Acheson, 2006). Every 6 years (EEC Council Directive, 1992), the government reviews the number of licenses released and decides if one new company can start to operate, according to how much time was spent with the wildlife during the last year. If the time that the existing operators have spent with the animals in the past year is less than the maximum time allowed by an amount that is equal or greater than the average time spent with the animals by an operator in that year then a new operator is allowed to start.

User group – In this scenario tour operators calculate the maximum amount of time they can sustainably spend with the wildlife and the effect their operations are having on the population. They own this quota of time with the animals (TWA) and they can trade it with the other operators. At the end of every day if a tour operator has used most of their TWA they will try to buy more time from those tour operators that have spent less than a third of their allowance, who, in turn, sell all the TWA they did not spend with the animals. The TWA sold by the tour operators is then distributed among the tour operators that want to buy extra time. The next day the tour operators who bought some TWA will have extra time to spend with the animals, while those who sold their TWA will have less time. The cost per minute with animals is 2 money units. Every 6 years (EEC Council Directive, 1992), the current tour operators decide if a new operator can start. Every operator gets a vote, which is a binomial draw with a probability given by an operator's own demand:supply ratio. Only if all tour operators agree a new operator is allowed to start.

Co-management – In this governance scenario, powers and responsibilities are shared between the operators and another entity, which can be either the government, an organisation or an environmental agency. The management authority, funded by the tour operators (1.8 money units per minute of tour), monitors the wildlife by calculating the maximum time allowed with the wildlife every year and the effect of the tourism activities on the population. This entity also enforces fines for defecting tour operators in the same way as in the licensing scenario. However, their probability of detecting defectors is higher (0.7; Pirotta & Lusseau, 2015). When a new operator wants to start, every 6 years, all the actors in the governance institutions need to agree. Access to the resource is granted to a new operator only when the population is not overexploited (same criterion as in the licensing) and all tour operators agree (same as in previous scenario). As in the user group scenario, the operators can trade the TWA.

2.3. Tourists

Every year a new annual population of 1 million tourists is initialised. We set up different populations of tourists that represent the different stages of the wildlife tourism destination life cycle (Fig. 1; Duffus & Dearden, 1990): a population of mainly specialist tourists, one of mainly generalists and one made up of both. Specialist tourists tend to be more knowledgeable about the wildlife species and the ecological system they are visiting, less reliant on infrastructure and generally willing to pay more money to access a high quality product (Bryan, 1977; Pabel & Coghlan, 2011). Specialist tourists usually have previous wildlife watching experience, they can be highly skilled and carry specialised equipment (Bryan, 1977; Lemelin, Fennell, & Smale, 2008). Generalist tourists, on the other hand, are less focused on the wildlife and more on the overall experience of the destination (Bryan, 1977; Catlin et al., 2011; Duffus & Dearden, 1990). They consider other aspects important in their destination or tour choices such as quality of service and presence of amenities. They are generally less knowledgeable or experienced and show a lower per capita expenditure (Catlin, Jones, Norman, & Wood, 2010; Jones, Wood, Catlin, & Norman, 2009).

In our simulations, every tourist has a maximum budget, which is taken from three normal distributions representing three socioeconomic statuses. High income tourists (10% of the total population) are assigned a maximum price taken from a normal distribution with a mean of 60 and a SD of 1.30% of the tourists belong to the middle income class and are assigned a maximum price from a normal distribution with a mean of 45 and a SD of 3.5, while the normal distribution describing the maximum price of low income tourists has a mean of 30 and a SD of 1.5. This was based on an average price of around US$ 30 per hour (Buckley, 2007; Mayer et al., 2018; Sekercioglu, 2002), assuming that the budget of high-income tourists would be around twice the average price of a ticket and the budget of middle and low income tourists be slightly higher than and just equal the average price respectively. Every tourist is also assigned a minimum rating (on a scale from 0 to 5) for a wildlife tour that they are willing to pay for, and this represents their level of specialisation. The minimum rating for each tourist is one of the parameters tested in the different simulations. It is a number between 0 and 1 taken from 3 beta distributions and multiplied by 5 so that a number between 0 and 5 is returned:

$$\beta(0.8,\ 3)\ \beta(3,\ 0.8)\ \beta(1.9,\ 1.9)$$

These three distributions simulate generalist tourists (minimum tour rating generalist tourists are willing to pay for between 0 and 1), specialist tourists (minimum rating between 4 and 5) and a mix between the two (minimum rating between 2 and 3 - Fig. S1).

Every day a number of tourists is sampled from the yearly population of tourists through independent binomial draws. Initially every tourist has the same probability of being sampled (0.5); if a tourist has tried to book a tour more than once they are more likely to be sampled (1), while once they go on a tour they are less likely to be sampled the following days (0.1). The daily number of tourists that are sampled from the year population is taken from a normal distribution with a SD of 10 and a mean that changes with season and year (Fig. S2):

$$\mu_{tourists} = \mu_0 + year*day - season*\cos\left(2*\frac{\pi}{365}*day_{ofyear}\right) \quad (2)$$

where $\mu_0 = 200$, *year* (the annual trend and one of the parameters tested) is either −0.005, 0 or 0.005, *day* is the day of the simulation (from 1 to years*365), and day_{ofyear} is the day of the year (from 1 to 365). *season* is the seasonal effect, which is taken from a normal distribution with a mean (μ_{season}) of 50 and a SD (σ_{season}) of 10. The seasonal effect simulates the seasonality in the number of tourists visiting a tourism destination (here parameterised on a dolphin watching destination in Scotland Davis, Pita, Lusseau, & Hunter, 2010) and the year effect simulates different trends in tourism demand: increasing ($year = 0.005$; exploration and development phase), stable ($year = 0$; consolidation and stagnation phase) and decreasing ($year = -0.005$; industry collapse).

Every day tourists try to book a wildlife watching tour with their preferred tour operator, according to their maximum price and minimum rating. Every tourist selects a set of tour operators that are within their budget and rank them according to the rating, then they will try to book with their preferred operator (the tour with the highest rating within their budget; Gavilan, Avello, & Martinez-Navarro, 2018; Ye, Law, & Gu, 2009). If the tour is fully booked, they will try to book with the second preferred and so on until they either book a tour or run out of tours above their minimum rating.

After the tour, each tourist estimates their satisfaction with the tour. Different drivers of tourists' satisfaction with wildlife watching tours have been identified in empirical studies, for example interactions with wildlife, satisfaction with prices charged, interpretation services and facilities provided (Mutanga, Vengesayi, Chikuta, Muboko, & Gandiwa, 2017; Okello & Yerian, 2009). Therefore, we implement 5 components to tourist satisfaction: time spent with the animals, price of the tour, how long they had to wait before booking (an indication of crowding), operator's investment in services (for example guides, interpretation material, equipment etc.) and an unassigned satisfaction point which reflects individual differences in tourists. The first 4 components of satisfaction are all sigmoid functions:

$$Satisfaction_{time} = 1 + \frac{(0.01 - 1)}{\left(1 + e^{\left(l_{time} * \left(\left(\frac{time_{withanimals}}{tour_{time}}\right) - i_{time}\right)\right)}\right)} \quad (3)$$

$$Satisfaction_{price} = 1 + \frac{(0.5 - 1)}{\left(1 + e^{\left(-l_{price} * \left(\frac{(price/max\_price)}{(rating/max\_rating)} - i_{price}\right)\right)}\right)} \quad (4)$$

$$Satisfaction_{waiting} = 1 + \frac{(0.01 - 1)}{\left(1 + e^{\left(-l_{waiting} * \left(\left(\frac{waiting}{365}\right) - i_{waiting}\right)\right)}\right)} \quad (5)$$

$$Satisfaction_{service} = 1 + \frac{(0.01 - 1)}{\left(1 + e^{\left(-l_{service} * \left(\left(\frac{investment}{max\_investment}\right) - i_{service}\right)\right)}\right)} \quad (6)$$

The functions were parameterised by visually inspecting the output. $Satisfaction_{time}$ was parameterised to be 1 when 45% of the tour was spent with the animals, with a slope (l_{time}) of 15 and an inflection point (i_{time}) of 0.3. $Satisfaction_{price}$ was calculated so that tourists were more satisfied when the price-quality ratio was lower in comparison to other operators, with a slope (l_{price}) of 15 and an inflection point (i_{price}) of 0.7. $Satisfaction_{waiting}$ was a function of the proportion of the year the tourist had to wait to successfully book a tour ($l_{waiting} = 60$ and $i_{waiting} = 0.1$). The satisfaction for services offered by the operator ($Satisfaction_{service}$) was in comparison to those offered by the other operators with a slope ($l_{service}$) of 10 and an inflection point ($i_{service}$) of 0.3. The random satisfaction component was a binomial draw with a probability equal to the other satisfaction components multiplied.

2.4. Tour operators

The model is initialised with 10 tour operators. The price of the tours offered by each operator is taken from a normal distribution with a mean of 30 and a SD of 1. They all start with an average rating of 3 and a capacity (number of tourists per tour) taken from a uniform distribution between 10 and 30. The tour operators are also assigned a behavioural phenotype that will determine their behavioural strategy when confronted with the choice of cooperating vs defecting. When modelling the behaviour of natural resource users, it is commonly assumed that they will make rational decisions driven by cost-benefit considerations. However, when faced with dyadic games such as prisoner dilemma, people's decisions follow four consistent behavioural phenotypes (Poncela-Casasnovas et al., 2016): optimist, pessimist, trustful and envious, while a small percentage shows undefined behaviour. Optimist individuals try to maximise the maximum pay-off by cooperating only in situations where the benefit of defecting when the others cooperate is less than the benefit of cooperating when the others cooperate. Pessimist individuals cooperate only when the payoff of cooperating when others defect is less than the payoff of defecting when the others defect, thus ensuring a best worst-case scenario. Envious individuals try to prevent the other users from receiving more payoffs than themselves, by only cooperating when the payoff from cooperating when the others defect is greater than or equals the payoff from defecting when the others cooperate. Trustful agents always cooperate, while undefined agents cooperate randomly with a probability of 0.5. We incorporate this principle and the behavioural phenotypes when modelling tour operators' choices of following or breaking regulations on the maximum time allowed with the animals. The behavioural phenotype are randomly assigned by sampling from a population of 17% trustful, 20% optimist, 21% pessimist, 30% envious and 12% undefined individuals, as calculated in the original study (Poncela-Casasnovas et al., 2016).

All the tour operators that receive enough bookings to cover all the costs of the tour (1.5 money unit per minute of tour) will run a tour on that day. Every tour lasts 90 min. We assume that all tour operators respect a voluntary code of conduct, which recommends a maximum of 10 min per encounter, and we assume a maximum of 5 encounters per tour. The time that they can potentially spend with the animals is then calculated as the sum of 5 binomial draws with probability equal to the probability of encounter multiplied by 5 random values taken from a truncated exponential distribution between 1 and 10.

If the time calculated is higher than the maximum time allowed, in the licensing and co-management scenarios, the tour operator needs to choose whether to cooperate and only spend the time they are allowed with the animals, or to defect and spend all the time that is available with the animals. The choice is made using a matrix of payoffs and the operator's phenotype. There are four possible situations and payoffs that make up the payoff matrix: the tour operator cooperates while the others also cooperate (R), the tour operator defects while the others cooperate (T), the tour operator cooperates while the others defect (S) and the tour operator defects while the others also defect (P).

	C	***D***
C	*R*	*S*
D	*T*	*P*

The payoffs are calculated based on the hypothetical income the tour operator would make by attracting more tourists in case of defection (because satisfaction increases with time spent with animals), the competitive advantage or disadvantage given by defecting while the others cooperate and by cooperating while the others defect and, in the licensing and co-management scenarios, the possibility of a fine.

$$R = ticket * sum\left(binom\left(capacity,\ tourist\ satisfaction_{coop}\right)\right) \tag{7}$$

$$T = ticket*sum\left(binom\left(capacity,\ tourist\ satisfaction_{defect}\right)\right) + \left(ticket*\ sum\left(binom\left(capacity,\ tourist\ satisfaction_{defect}\right)\right) - ticket*sum\left(binom\left(capacity,\ tourist\ satisfaction_{coop}\right)\right)\right) - fine*binom(1,\ detection\ prob) \tag{8}$$

$$S = ticket*sum\left(binom\left(capacity,\ tourist\ satisfaction_{coop}\right)\right) + \left(ticket*\ sum\left(binom\left(capacity,\ tourist\ satisfaction_{defect}\right)\right) - ticket*sum\left(binom\left(capacity,\ tourist\ satisfaction_{coop}\right)\right)\right) \tag{9}$$

$$P = ticket*sum\left(binom\left(capacity,\ tourist\ satisfaction_{defect}\right)\right) - fine*binom(1,\ detection\ prob) \tag{10}$$

where *ticket* is the price of the tour; *capacity* is the tour operator's maximum capacity; $tourist\ satisfaction_{defect}$ is calculated from equation (3) where $time_{withanimals}$ is the time spent with the wildlife in case of defection; $tourist\ satisfaction_{coop}$ is also calculated from equation (3) but where $time_{withanimals}$ is the time spent with the wildlife in case of cooperation; *fine* is the penalty for defecting and *detection prob* is the probability that a defecting tour operator will be caught and fined.

The decision to cooperate or defect will depend on the payoff matrix and the agent's behavioural phenotype:

$$if\ phenotype = trustful \Rightarrow behaviour = cooperate$$

$$if\ phenotype = optimist\ AND\ S < R \Rightarrow behaviour = cooperate$$

$$if\ phenotype = pessimist\ AND\ T > P \Rightarrow behaviour = cooperate$$

$$if\ phenotype = envious\ AND\ S \geq T \Rightarrow behaviour\ = cooperate$$

$$if\ phenotype = undefined \Rightarrow behaviour = sample((defect,\ cooperate),\ prob = (0.5,\ 0.5))$$

In any other instance the tour operator will defect.

At the end of each year, tour operators make investment decisions. Tour operators can make two types of investments: infrastructure and services. The infrastructure investment will allow them to have more tourists on their tours, so increasing profits keeping costs the same; the service investment will increase tourist satisfaction, thus attracting more tourists. The probability of investing in services is a logistic function of the ratio between the operator's rating in relation to the maximum rating of the other operators.

$$p_{investment\ service} = \frac{1}{1 + e^{\left(-l_{service}*(rating\ /\ \max\_rating) - i_{service}\right)}} \tag{11}$$

where $l_{service} = 20$ and $i_{service} = 0.75$. It is parameterised so that when the rating is up to 0.7 of the maximum (e.g. on a scale from 1 to 5, a rating of 3.5) the probability of investing is high, and it gets quickly smaller after reaching 3.75. The decision to invest is then a binomial draw based on this probability. The amount invested is a random proportion between 1 and 100% of the available money, which is the profit for that year.

The probability of investing in infrastructure is a logistic function of the proportion of the operator's profit in relation to their own maximum profits.

$$p_{investment\ infrastructure} = \frac{1}{1 + e^{\left(-l_{infrastructure}*\left(profit\ /\ \max\_profit\ /\ 2\right) - i_{infrastructure}\right)}} \tag{12}$$

where $l_{infrastructure} = 30$ and $i_{infrastructure} = 0.7$. It is parameterised so that when the profit (*profit*) is up to 0.7 of the maximum (*max_profit*; a profit of 7000 over a maximum of 10000) the probability of investing is low and it gets quickly higher after reaching 7500. The decision to invest is then a binomial draw based on this probability. The amount invested is given by the amount of money available and the maximum number of extra tourists that the operator can afford. The price per extra tourist is given by the equivalent of two weeks of work at full capacity.

Tour operators also update the price of their tour every year according to the demand/supply ratio:

$$ticket_{y+1} = ticket_y * \frac{bookings_y}{capacity_y} \tag{13}$$

where $ticket_y$ is the price of the tour in the previous year, $bookings_y$ is the number of bookings made in the previous year and $capacity_y$ is the tour operator's maximum capacity.

2.5. Wildlife

We model wildlife use as in Pirotta and Lusseau (2015). The wildlife has an annual growth rate of 1%, which is affected by tourism exposure. This effect is a sigmoid function of the time spent with the wildlife by all tour operators during the past year and the number of tourists on the tours. The time component is parameterised so that when the cumulative time with the animals (*withanimals*) is less than the maximum time allowed (*max*) then the effect is 0, when the time is $max + 1/5\ max$, then the effect results in a 0.05 change in the probability of encounter. Beyond that, the effect increases towards the maximum, which is 0.1. The tourist volume component is parameterised so that when the number of tourists is less than 50% more than the initial number of tourists, the effect is 0, when the operators expand by more than 50% of the initial capacity the effect starts to increase towards the maximum, which is 0.1.

$$effect_y = 0.1 + \frac{-0.1}{\left(1 + e^{\left(slope_{time} * \left(withanimals_y - \left(\max_y + max_y / 5\right)\right)\right)}\right)} + 0.1 + \frac{-0.1}{\left(1 + e^{\left(-slope_{capacity} * \left(\frac{capacity_0}{capacity_y} * 100 - 50\right)\right)}\right)} \tag{14}$$

The slopes of both effects are dependent on the population size since bigger populations show a slower response to disturbance because of decreased exposure per capita as abundance increases (Pirotta & Lusseau, 2015):

$$slope_{time} = \frac{0.00025}{max_y / 100000} \tag{15}$$

$$slope_{capacity} = \frac{0.2}{max_y / 100000} \tag{16}$$

The probability of encountering animals in the area is a measure of the density of animals and population size. It changed every year according to the tourism exposure in the previous year:

$$encounter\ prob_y = encounter\ prob_{y-1} * \left(1.01 * effect_{y-1}\right) \tag{17}$$

and it is initialised at 0.7.

2.6. Software

The model was implemented in R version 3.5.0 (R Core Team, 2018) using packages dplyr version 0.7.4 (Wickham, Francois, Henry, & Müller, 2017), RGeode version 0.1.0 (Rimella, 2017). Simulation runs were distributed for parallel computing using doParallel version 1.0.11 (Microsoft Corporation & Weston, 2017a) and foreach version 1.4.4 (Microsoft Corporation & Weston, 2017b) on a cluster with Bio-Linux OS (Kernal version (uname -r) = "3.13.0–128-generic" VERSION = "14.04.5 LTS, Trusty Tahr").

3. Results

We found that tourist phenotype influenced the socioeconomic and ecological dynamics of the simulated wildlife tourism destination (Figs. 3 and 4). Under any governance scenario and with any trend in demand, a destination visited mainly by generalist tourists will have the highest number of tour operators still in business after 40 years (between 6 and 17 – Fig. S3), while a destination dominated by specialist tourists will only have between 1 and 5 tour operators still in business after 40 years (Fig. S3). This effect is reflected into the average time that tour operators stay in business (Fig. 3). Destinations with generalist tourists show an average of 10–50 years of business for tour operators, while destinations with specialist tourists have an average length of business of around 5 years (Fig. 3). In the code of conduct scenario we can still see the same pattern, but the difference in how long a company stays in operation between the different tourist phenotypes is smaller in this scenario. We also see a higher turnover rate of tour operators in the code of conduct scenario and licencing compared to the other governance systems (Fig. S4), with more new operators entering the market. The average length of business in the code of conduct scenario is lowered by this higher turnover rate of operators increasing competition.

The volume of tourists per operator differs slightly between the different governance scenarios, but, once again, difference among tourist phenotypes was pronounced. Simulations with generalists and mixed tourists had more bookings compared to the specialists (Fig. 4). Simulations with the governance structures of co-management, user group and, to a lesser extent, licensing, had more bookings per tour operator on average (Fig. 4). This is a consequence of fewer tour operators in these governance scenarios compared to the code of conduct (Fig. S3).

The difference in tour ticket price between the co-management scenario and the other three (Fig. S5) is only due to the higher costs that tour operators have. In this governance structure tour operators directly fund the monitoring and management of the destination, therefore they keep prices higher to balance this extra cost. More interestingly, we find a difference in tour ticket price between simulations with specialist and generalist tourists, with prices slightly higher for specialist destinations than for generalist ones. It appears that, just as in real life, wildlife destinations with specialist tourists create a higher-quality product that tourists are more satisfied with (Fig. 5) and therefore willing to pay more for.

The code of conduct scenario has the most impact on the wildlife, with the growth rate of the wildlife population decreased by 10% and the probability of encountering the animals collapsed to between 0.4 and 0 (Fig. 6) by the end of the 50 years in a scenario where the number of generalist tourists increases. The licencing scenario can also result in overexploitation of the wildlife if the number of generalist visitors to the destination increases, but to a lesser degree compared to the code of conduct scenario, with a probability of encountering the wildlife still

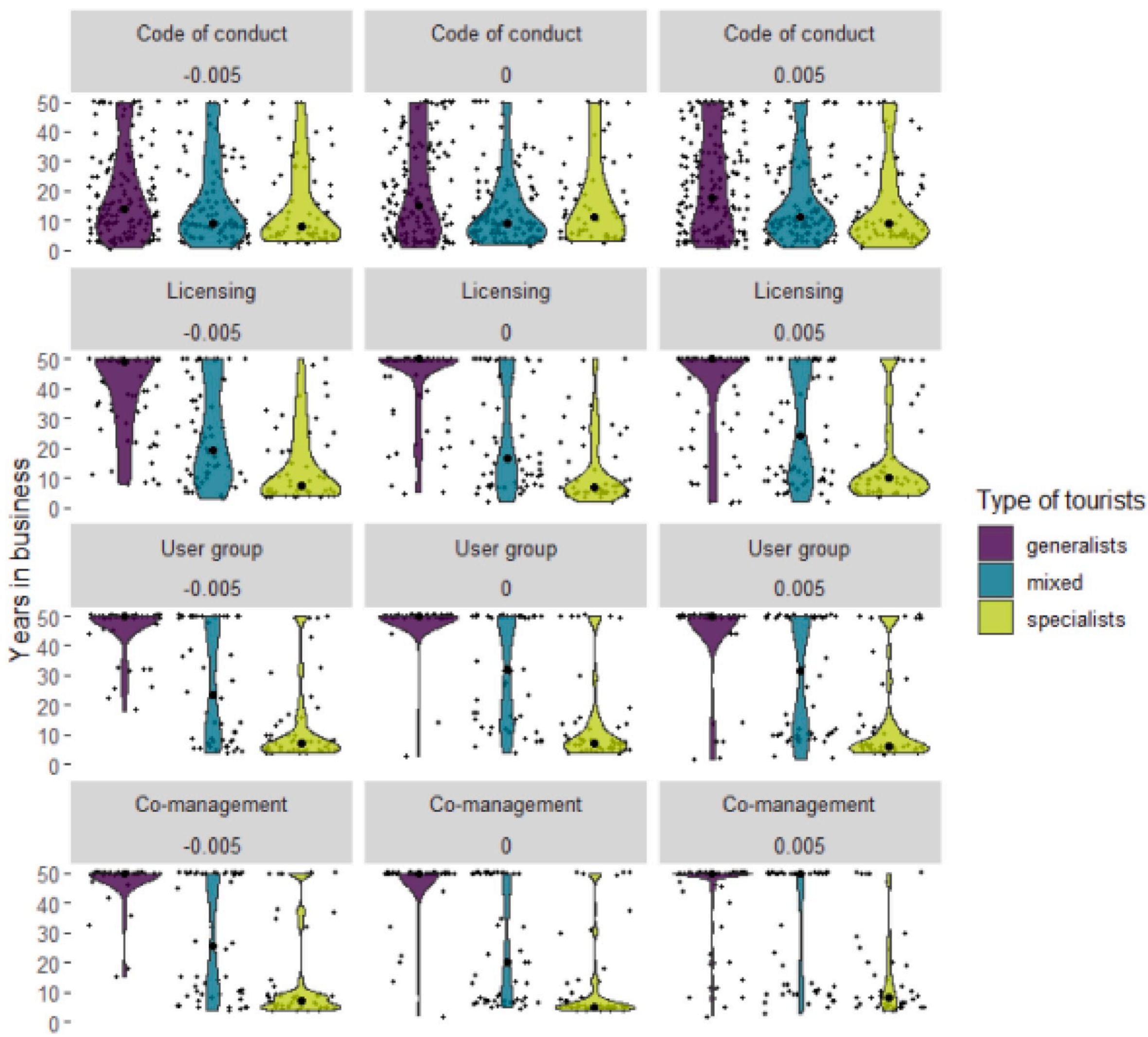

Fig. 3. Years tour operators remain in business. The points are the length of time (in years) that every single tour operator in the simulation was active. Violin plots show the distribution of the data, with the central dot representing the median. Rows are different governance scenarios and columns are different trends in tourism demand (−0.005: decreasing; 0: stable; 0.005: increasing).

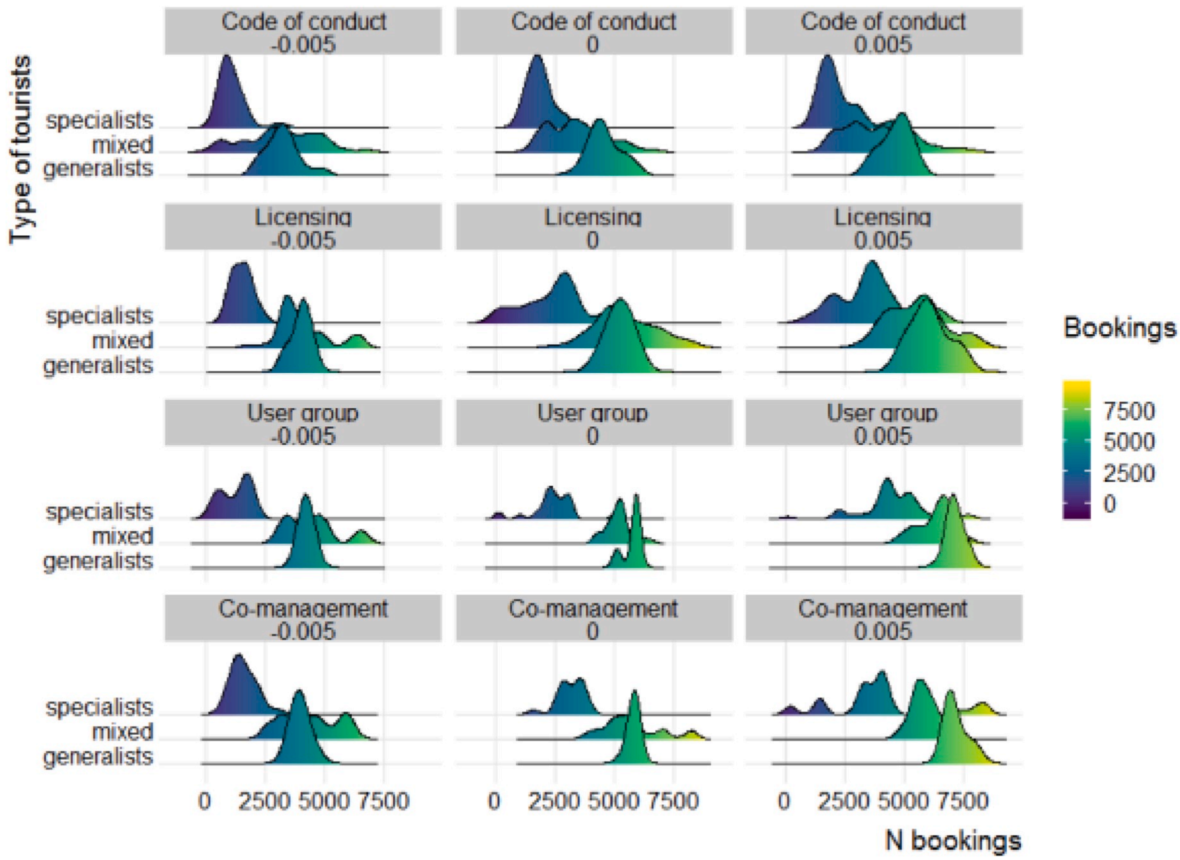

Fig. 4. Volume of tourists per tour operator. The plots show the distributions of the number of bookings an operator receives in the last 10 years of the simulation across all replicates. Rows are different governance scenarios and columns are different trends in tourism demand (−0.005: decreasing; 0: stable; 0.005: increasing).

between 0.7 and 0.5 by the end of the simulations (Fig. 6). The other two governance structures, on the other hand, never lead to overexploitation (Fig. 6).

4. Discussion

Looking at the three dimensions of sustainability (social, economic and ecological), we can see that there is no clear winner among the governance strategies tested in these simulations (Fig. 7). Both code of conduct and licensing governance structures can lead to overexploitation when the number of generalist tourists is increasing and the other scenarios can lead to low profits and a very small number of tour operators remaining in the industry. The main driver of sustainability was the type of visitors to the destination. Mixed and specialist tourists never produce overexploitation because they demand higher quality experiences (Figs. 4 and 5), while generalist tourists lead to a different industry, characterised by high volume and low prices (Fig. 4 and Fig. S5). However, even in a generalist destination, overexploitation of the wildlife can be avoided with user-group and co-management governance structures, which will reduce the number of tour operators active in the destination to an acceptable level (around 10 – Fig. S3) and keep disturbance down to a sustainable degree (Figs. 6 and 7). On the other hand, when the destination is mainly visited by specialists or by mixed tourists, these governance structures usually lead to monopolisation of the industry by two or three operators (Fig. 7 and Fig. S3).

In the initial phase of development of a wildlife tourism destination, when most visitors are specialists who do not require much infrastructure development, are low in numbers and are willing to pay more for higher quality experiences, codes of conduct can be a successful strategy. Up to 5 tour operators could successfully run their business without overexploiting the wildlife (Fig. 7). Even establishing a code of conduct that regulates the interactions between humans and wildlife is a process that requires collaboration and partnerships. Therefore, this could be a good tool to establish those institutions that will need to be in place for a successful change in governance structure later in the destination's development. When the destination reaches the stage where specialist tourists have been replaced by generalists, a change in governance is needed. If the destination is still experiencing an increase in demand, more operators will enter the business creating a strong competition that will lead to unsustainable socioeconomic dynamics (e.g. very high turnover of tour operators due to competition driving many of them out of business; Fig. 3 and Fig. S4) and the overexploitation of the wildlife (Figs. 6 and 7). This stage of development requires a stricter definition of resource system boundaries and rules on who can use the resource (Ostrom, 2009). The licensing scheme introduces such rules, by only releasing new licences if the wildlife is not overexploited. However, because of the high uncertainty that is typical of monitoring by central authorities and the number of defectors (Fig. S6), this governance system still does not prevent overexploitation of the wildlife in this stage of the destination development. The user group and co-management strategies introduce stricter rules on the access to the resource. In both the user group and the co-management governance scenario, new operators looking to start their business in the destination need the consent of all existing tour operators. In addition, in the co-management governance system the level of exploitation of the wildlife is also considered in the decision of allowing a new operator to start. This protects the wildlife when the number of tourists is high (Fig. 7).

Often, these governance solutions are advocated as panaceas, a single solution to every common pool resource issue (Armstrong & Sumaila, 2001; Hardin, 1968; Lovejoy, 2006). More recent work on socioecological systems and common pool resources have amply criticised this idea that there can be a blue print for successful governance of

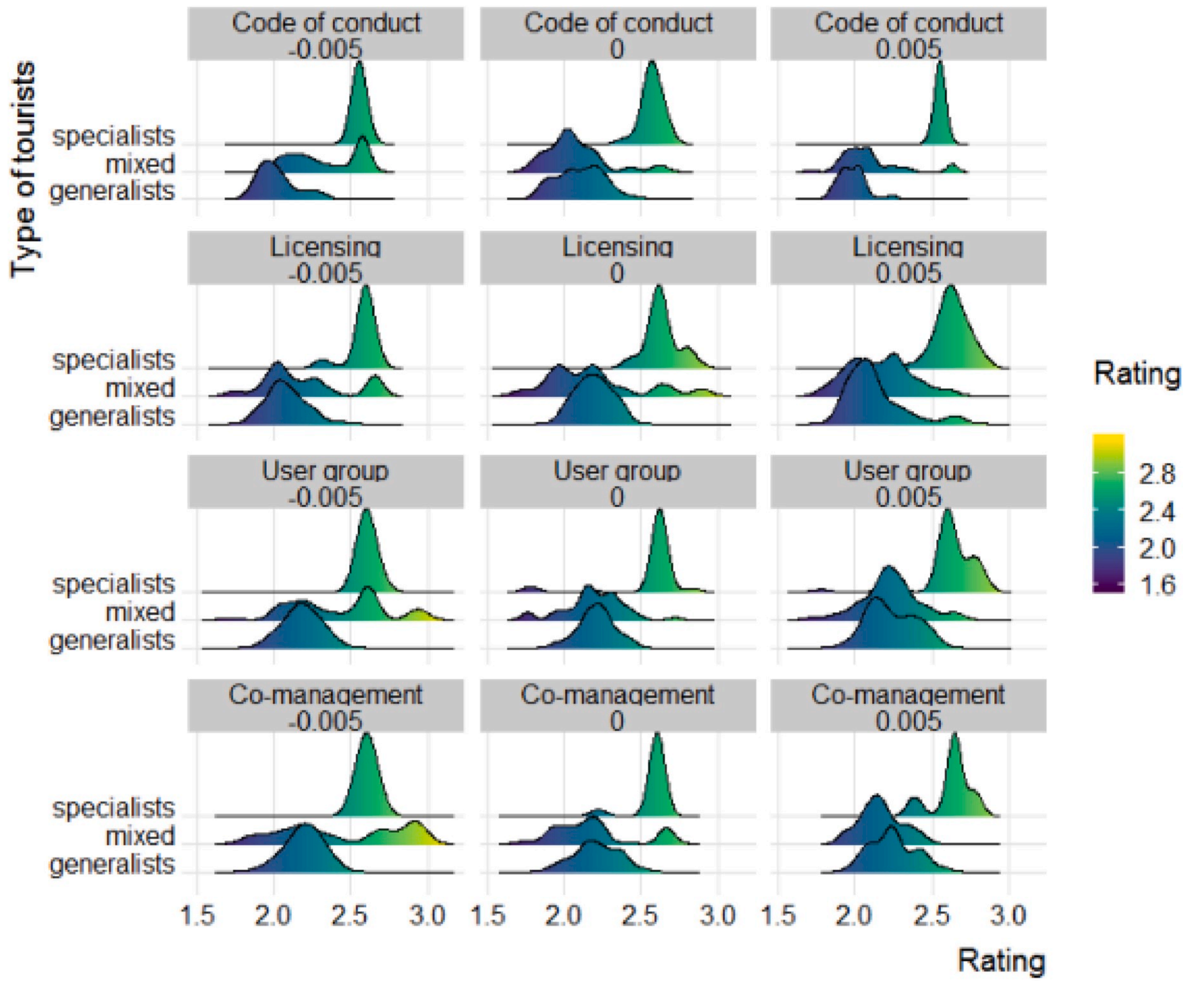

Fig. 5. Tour operators rating. The plots show the distributions of the average rating received by the tour operators in the last 10 years of the simulations across all replicates. Rows are different governance scenarios and columns are different trends in tourism demand (−0.005: decreasing; 0: stable; 0.005: increasing).

these systems (Anderies, Rodriguez, Janssen, & Cifdaloz, 2007; Meinzen-Dick, 2007; Ostrom, 2007; Ostrom et al., 2007). Panaceas fail because every commons is different: the ecology of the resource exploited, the socioeconomic and behavioural characteristics of the users, the size and dynamics of the resource system, all matter to determine the outcomes of a socioecological system (Ostrom, 2009). Applying one solution to a multitude of problems will inevitably fail multiple times (Acheson, 2006). Given the dynamics of a wildlife tourism destination as it matures (Duffus & Dearden, 1990) and the inability of any single governance structure to maintain sustainability in every phase of this tourism life cycle (Fig. 7), the best option is to adopt an adaptive governance approach. Importantly, it also means that this adaptive governance must include the monitoring of tourist phenotype to qualify the destination phase.

Robust governance of natural resources needs to be prepared for change (Dietz, Ostrom, & Stern, 2003), because the current state of knowledge is likely incomplete and the biophysical and social systems that constitute the commons can change very quickly. Dietz and colleagues introduced the concept of adaptive governance in 2003 (Dietz et al., 2003), and, since then, its use has steadily increased (Hasselman, 2017). It refers to the establishing of flexible and learning-based collaborations and decision-making processes involving different stakeholders at multiple levels, with the goal to adaptively negotiate and coordinate the management of socioecological systems (Schultz, Folke, Österblom, & Olsson, 2015). Adaptive governance provides the context and the coordination for choosing the best management tool from a toolbox (quotas, regulations etc.), and to adapt them as the socioecological system evolves. Adaptive governance can evolve from the beginning, thanks to strong social networks of users (Partelow & Nelson, 2018). In most cases, adaptive governance started from a moment of crisis and involved a mental shift, which reframed human-nature relationships by building trust and shared knowledge, and by connecting networks of stakeholders (Olsson, Folke, & Hahn, 2004; Olsson, Folke, & Hughes, 2008); in addition, using the knowledge and expertise from the different actors involved, it extended the classical toolbox of management solutions, including a series of informal governance tools (Schultz et al., 2015). We know what the trajectory of a wildlife tourism destination will be (Duffus & Dearden, 1990), what we do not know is when the phase shifts are going to happen and what the magnitude of the changes will be. As with any complex adaptive system, these state shifts are dependent on both internalities and externalities. However, by putting in place the mechanisms that allow for the emergence of adaptive governance from the early stages of development, we could avoid crises and allow for the development of institutions that can adapt to change and keep the destination sustainable. This requires social relationships between the operators, the creation of organisations that include all the key stakeholders and that will create trust and shared knowledge on the system across all the actors, a collaboration across sectors and institutional levels, the creation of flexible, adaptable rules and management tools (e.g. ecosystem-based management; Grumbine, 1994) and the willingness to embrace institutional change (Schultz et al., 2015). Monitoring of the system to detect early signs of change is crucial to anticipate crises and prepare for change. While tourism in some national parks and protected areas is well monitored and there are

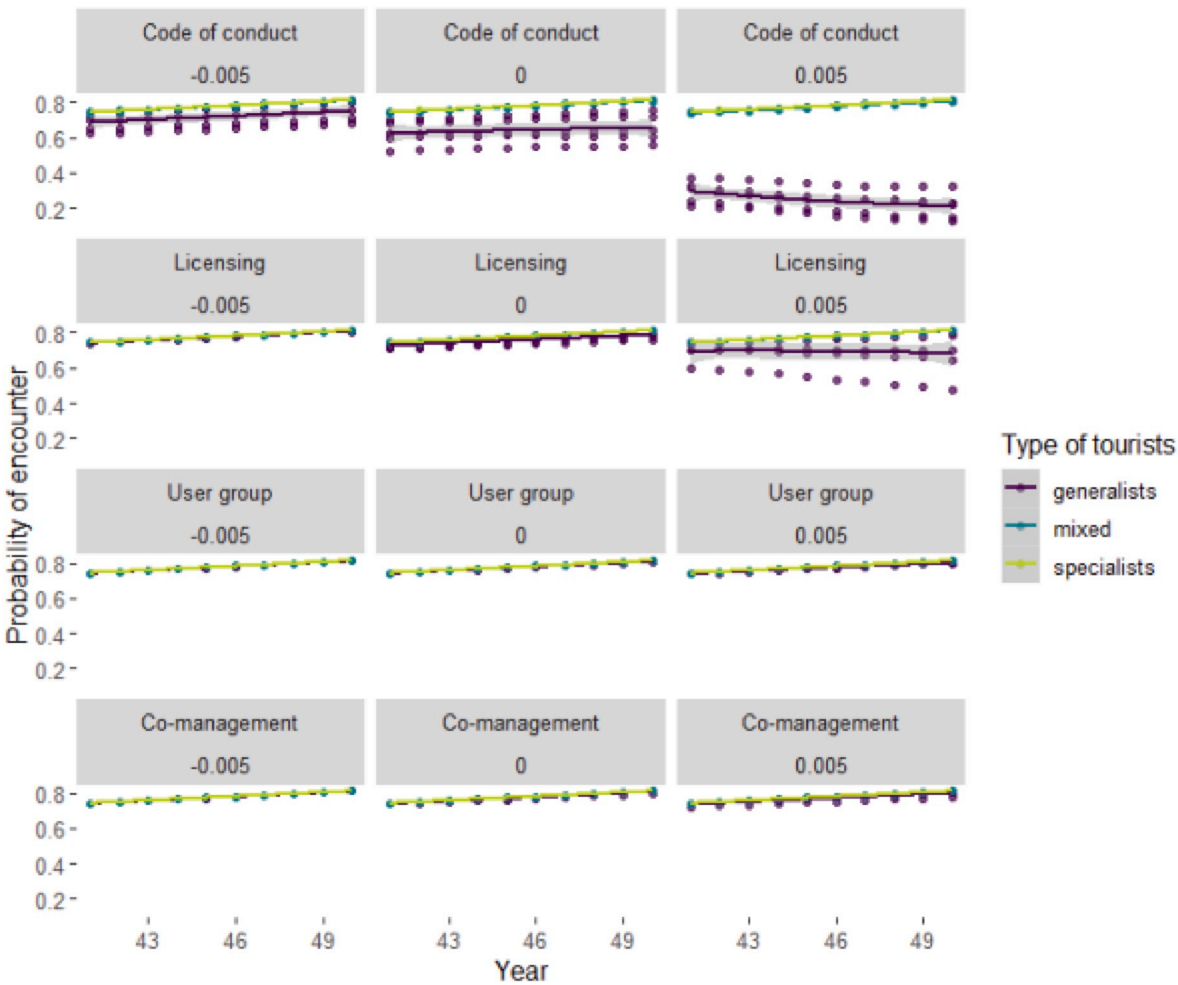

Fig. 6. Probability of encountering the wildlife. The plots show the probability of encountering the wildlife during a tour for the last 10 years of the simulations. Every data point is the value of the probability of encounter in a year for one replicate. Rows are different governance scenarios and columns are different trends in tourism demand (−0.005: decreasing; 0: stable; 0.005: increasing).

already governance structures in place, much wildlife tourism happens outside these areas, and unless there is a national or international strategy for wildlife tourism management, these areas will not be monitored and problems will only be detected when impacts have already occurred (Higham, 2007). Moreover, in many cases there will be no governance structures or partnerships to build on, and the process of developing the capacity for adaptive governance will have to start from scratch. Monitoring tourist phenotype around the world, using social media sampling (Mancini, Coghill, & Lusseau, 2018), may be a useful and efficient way to detect early warning signals that the state of destinations may be changing.

5. Limitations

As any model, the simulations presented in this study represent a simplified version of a real wildlife tourism system. From this simplification a number of limitations arise that need to be acknowledged and discussed.

Many components of real wildlife tourism systems have not been addressed in this study. We did not consider the impact of differences in some of the variables identified by Ostrom (2009) that can affect sustainability of socioecological systems, for example the focal species. We consider a slow-reproducing, large-bodied focal species that represents some of the charismatic wildlife groups targeted by tourism (e.g. large mammals) and our model will not be representative of tourism targeting different groups of species. For example snorkelling and diving destinations targeting coral reefs present a set of different challenges associated with recreational activities, including pollution, trampling and purchase or collection of reef species (Morais, Medeiros, & Santos, 2018). The focal species targeted by the tourism industry can also influence the trajectory of the destination's development and therefore the results of the simulations; for example, charismatic species will attract more generalist tourists from early stages, while less charismatic biodiversity might slow down the progression towards the saturation phase by attracting mainly specialist tourists (Hausmann, Slotow, Fraser, & Di Minin, 2017).

An important system component missing from our model is the local population, which can have important consequences on and be seriously affected by the development of the tourism destination. The values, attitudes and behaviours of local communities towards the focal species play an important role in the sustainability of the industry. For example if the animals are hunted for food or because they are perceived negatively by the local community (Hemson, Maclennan, Mills, Johnson, & Macdonald, 2009; Muboko, Gandiwa, Muposhi, & Tarakini, 2016), this can create conflicts and can damage the tourism destination either through a decline in the abundance of the target species or through decreased attractiveness to tourists because of these activities (Higham & Lusseau, 2007). The effect of tourism development on the lives of local communities is also an important component of the assessment of the sustainability of the tourism destination. Local communities can receive substantial benefits from the sustainable management of a tourism destination, such as poverty alleviation (Ferraro & Hanauer, 2014),

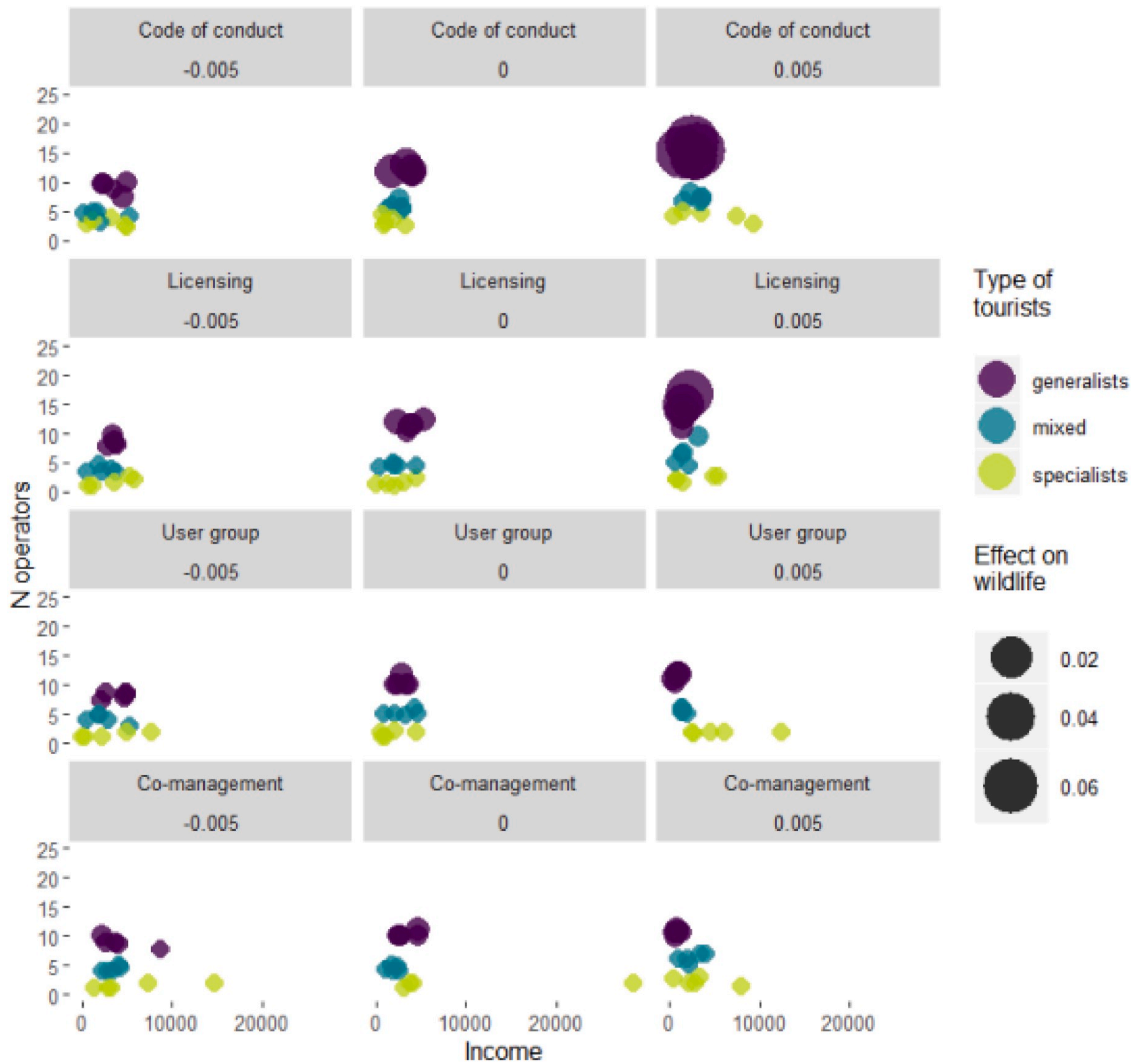

Fig. 7. Sustainability of a wildlife tourism destination. Every plot shows the mean profits earned by a tour operator on the x axis and the number of tour operators left in business on the y axis. The size of the bubbles indicates the mean effect of the tourism industry on the wildlife. The means are calculated for the last 10 years of the simulations and across the replicates. Rows are different governance scenarios and columns are different trends in tourism demand (−0.005: decreasing; 0: stable; 0.005: increasing).

improved quality of life through development of infrastructure and creation of employment opportunities (Li et al., 2018). On the other hand, the development of tourism destinations has been known to cause displacement of local communities from the destination (Sirima & Backman, 2013), damage to rural traditional culture and land-use conflicts (Xi et al., 2014). These variables and processes are difficult to quantify in empirical systems and therefore challenging to parameterise sensibly in a simulation study. Although not explicitly including local communities as agents, our individual-based model partially represents some of the processes in which local communities influence or are affected by the sustainability of a wildlife tourism destination in real systems. For example, the tour providers can be part of the local community and their success, or failure, can be extended to other members of the community who are involved in other tourism-related businesses. Additionally, local communities could be part of the monitoring and enforcement operations in the co-management strategy, as well as being involved in the assessment of the wildlife population status.

Our model also simplifies the behavioural responses of the human agents and does not consider the effect of their identity and values on their choices. For example, when the tour operators are part of the local community, they have a history of using the resource system and pre-existent social relationships, including recognised leaders in the community (Moore & Rodger, 2010). They also tend to have a shared knowledge of the resource system and their livelihoods is more dependent on it compared to any outsider (Moore & Rodger, 2010). These characteristics tend to lower the probability of defection and the perceived costs of self-organisation, leading to positive outcomes for the socioecological system (Dimmock, Hawkins, & Tiyce, 2014; Ostrom, 2009). The existence of shared knowledge and trust among the different interest groups is important because it leads to information, for example signs of environmental degradation, being recognised by all parties involved in the socioecological system as credible, salient and legitimate, which are the essential elements required for translating knowledge into action (Cash et al., 2003).

Finally, the simulations presented here do not consider the effect of externalities on the sustainability of the wildlife tourism system. Major drivers of global environmental change, such as agricultural intensification, habitat loss and climate change, have the potential to affect the sustainability of a wildlife tourism destination. Changes in the landscape and in species distributions can cause substantial economic losses to regional tourism systems, but the impacts of, for example, the loss of a target species on a tourism destination will vary among different tourist markets (Scott, Gössling, & Hall, 2012).

6. Conclusions

The expectations and preferences of tourists have a strong influence on the sustainability of a wildlife tourism destination. The dominating tourist phenotype in a destination can influence both the exploitation of the environment and the socio-economic success of the industry. We did not find a strong effect of governance type on the outcomes of the destination, with no governance structure appearing more successful than the others in every situation. What we find instead is that the best governance structure will depend on the state of the destination. When the destination is mainly visited by specialists tourists who demand higher standards of wildlife tours, then a simple code of conduct can lead to sustainable outcomes, but, as the population of tourists moves towards more generalist wildlife watchers, we find that this governance structure can lead to overexploitation of the wildlife. On the other hand, a governance structure where the central authority owns the property rights over the resource and issues licences to a restricted number of users can be successful in sustainably managing a destination dominated by generalist visitors, as long as numbers stay stable and do not increase. A destination that is undergoing a phase of growth in the number of visitors, can only be managed sustainably under a co-management or a user group governance regime. However, in both scenarios, the industry will not grow very large, with around 10 tour operators left in business after 50 years and a modest per capita income. Since a tourism destination will go through these different stages in its development (Duffus & Dearden, 1990), we can conclude that adaptive governance is necessary to avoid unsustainable outcomes.

The modelling approach presented in this study can be useful to those involved in the sustainable management of wildlife tourism destinations. The model can be applied to predict the trajectory of specific destinations starting from the development phase in which they are currently found and inform governance change. Because it is flexible, it can easily be parameterised to be adapted to specific real-world systems. It is also modular, so each component can be modified according to the specific needs and characteristics of the destination. Despite its potential to be specific, the model can still be general and therefore able to draw conclusions valid across specific contexts. The results from our simulations of a generic wildlife tourism destination point to the importance of monitoring of socioecological dynamics as the first take home message for all the parties involved in the management of wildlife tourism. Because the specific management tools more appropriate to lead to sustainable outcomes depend on the stage of development of the destination, it is important to determine how the destination is progressing through its natural life-cycle. Setting up a system of co-production of knowledge, where the tour operators collect information about their customers' preferences and about the wildlife would lead to evidence of change quickly being translated into action to adapt to this change (Cash et al., 2003). Shared knowledge is only the first step for building capacity for adaptive responses to change. Coordination, negotiation and collaboration also need to be enabled across sectors and institutional levels, which can be facilitated by building trust and establishing nested governance institution (Schultz et al., 2015). Sustainable management of the wildlife tourism commons is possible if its governance is flexible and prepared to respond timely to the changes brought in by the dynamic nature of these socioecological systems.

Author contribution

All authors contributed to conceiving the idea and designing the model. FM and DL planned and carried out the simulations. All authors contributed to the interpretation of the results. FM led the writing of the manuscript. All authors provided critical feedback and helped shape the research, analysis and manuscript.

Data statement

All the scripts necessary to reproduce the results presented in this paper are available at Mancini, F. (2018) WildlifeWatching IBM. https://doi.org/10.5281/zenodo.1443307. https://zenodo.org/account/settings/github/repository/FrancescaMancini/WildlifeWatchingIBM.

Declaration of competing interest

None.

Acknowledgments

This work was funded by the University of Aberdeen, through MASTS (the Marine Alliance for Science and Technology for Scotland), and Scottish Natural Heritage (SNH), through a Dominic Counsell studentship, and their support is gratefully acknowledged. The authors would also like to thank Alex Douglas for allowing us to run the simulations on the Catling computer cluster.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.tourman.2020.104160.

References

Acheson, J. M. (2006). Institutional failure in resource management. *Annual Review of Anthropology, 35*(1), 117–134. https://doi.org/10.1146/annurev.anthro.35.081705.123238.

Amo, L., López, P., & Martín, J. (2006). Nature-based tourism as a form of predation risk affects body condition and health state of Podarcis muralis lizards. *Biological Conservation, 131*(3), 402–409.

Anderies, J. M., Rodriguez, A. a, Janssen, M. a, & Cifdaloz, O. (2007). Panaceas, uncertainty, and the robust control framework in sustainability science. *Proceedings of the National Academy of Sciences of the United States of America, 104*(39), 15194–15199. https://doi.org/10.1073/pnas.0702655104.

Armstrong, C. W., & Sumaila, U. R. (2001). Optimal allocation of TAC and the implications of implementing an ITQ management system for the north-East Arctic Cod. *Land Economics, 77*(3), 350–359. https://doi.org/10.2307/3147129.

Balmford, A., Beresford, J., Green, J., Naidoo, R., Walpole, M., & Manica, A. (2009). A global perspective on trends in nature-based tourism. *PLoS Biology, 7*(6), e1000144. https://doi.org/10.1371/journal.pbio.1000144.

Balmford, A., Green, J. M. H., Anderson, M., Beresford, J., Huang, C., Naidoo, R., et al. (2015). Walk on the wild side: Estimating the global magnitude of visits to protected areas. *PLoS Biology, 13*(2), 1–6. https://doi.org/10.1371/journal.pbio.1002074.

Beale, C. M., & Monaghan, P. (2004). Human disturbance: People as predation-free predators? *Journal of Applied Ecology, 41*(2), 335–343. https://doi.org/10.1111/j.0021-8901.2004.00900.x.

Bejder, L., Samuels, A., Whitehead, H., Gales, N., Mann, J., Connor, R., et al. (2006). Decline in relative abundance of bottlenose dolphins exposed to long-term disturbance. *Conservation Biology, 20*, 1791–1798. https://doi.org/10.1111/j.1523-1739.2006.00540.x.

Briassoulis, H. (2002). Sustainable tourism and the question of the commons. *Annals of Tourism Research, 29*(4), 1065–1085.

Bryan, H. (1977). Leisure value systems and recreational specialization: The case of trout fishermen. *Leisure Research, 9*, 174–187.

Buckley, R. (2007). Adventure tourism products: Price, duration, size, skill, remoteness. *Tourism Management, 28*(6), 1428–1433. https://doi.org/10.1016/J.TOURMAN.2006.12.003.

Buckley, R. C., Castley, J. G., de Pegas, F. V., Mossaz, A. C., & Steven, R. (2012). A population accounting approach to assess tourism contributions to conservation of IUCN-redlisted mammal species. *PloS One, 7*(9), e44134. https://doi.org/10.1371/journal.pone.0044134.

Butler, R. W. (1980). The concept of a tourist area cycle of evolution: Implications for management of resources. *Canadian Geographer, 24*(1), 5–12. https://doi.org/10.1111/j.1541-0064.1980.tb00970.x.

Cash, D. W., Clark, W. C., Alcock, F., Dickson, N. M., Eckley, N., Guston, D. H., et al. (2003). Knowledge systems for sustainable development. *Proceedings of the National Academy of Sciences of the United States of America, 100*(14), 8086–8091. https://doi.org/10.1073/pnas.1231332100.

Catlin, J., Jones, R., & Jones, T. (2011). Revisiting Duffus and Dearden's wildlife tourism framework. *Biological Conservation, 144*(5), 1537–1544. https://doi.org/10.1016/j.biocon.2011.01.021.

Catlin, J., Jones, T., Norman, B., & Wood, D. (2010). Consolidation in a wildlife tourism industry: The changing impact of whale shark tourist expenditure in the Ningaloo coast region. *International Journal of Tourism Research, 12*(2), 134–148. https://doi.org/10.1002/jtr.742.

Center for Responsible Travel. (2014). *Economic impact of bear viewing and bear hunting in the great bear rainforest of British Columbia*. Retrieved from https://pacificwild.org/news-and-resources/research-and-reports/economic-impact-of-bear-viewing-and-bear-hunting-in-the.

Christiansen, F., & Lusseau, D. (2015). Linking behavior to vital rates to measure the effects of non-lethal disturbance on wildlife. *Conservation Letters, 8*(6), 424–431. https://doi.org/10.1111/conl.12166.

Christiansen, F., Rasmussen, M., & Lusseau, D. (2013). Whale watching disrupts feeding activities of minke whales on a feeding ground. *Marine Ecology Progress Series, 478*, 239–251. https://doi.org/10.3354/meps10163.

Conley, A., & Moote, M. A. (2003). Evaluating collaborative natural resource management. *Society & Natural Resources, 16*(5), 371–386. https://doi.org/10.1080/08941920309181.

Davis, B., Pita, C., Lusseau, D., & Hunter, C. (2010). *The value of tourism expenditure related to the East of Scotland Bottlenose dolphin population.* Aberdeen.

DeAngelis, D. L., & Mooij, W. M. (2005). Individual-based modeling of ecological and evolutionary processes. *Annual Review of Ecology, Evolution and Systematics*, (36), 147–168. https://doi.org/10.1146/annurev.ecolsys.36.102003.152644.

Dietz, T., Ostrom, E., & Stern, P. C. (2003). The struggle to govern the commons. *Science, 302*, 1907–1912. https://doi.org/10.1126/science.1091015.

Dimmock, K., Hawkins, E. R., & Tiyce, M. (2014). Stakeholders, industry knowledge and adaptive management in the Australian whale-watching industry. *Journal of Sustainable Tourism, 22*(7), 1108–1121. https://doi.org/10.1080/09669582.2013.879311.

Duffus, D., & Dearden, P. (1990). Non-consumptive wildlife-oriented recreation: A conceptual framework. *Biological Conservation, 53*, 213–231.

EEC Council Directive. (1992). On the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Communities, L 269(May 1992*, 1–15. Retrieved from https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:31992L0043:EN:HTML.

Ferraro, P. J., & Hanauer, M. M. (2014). Quantifying causal mechanisms to determine how protected areas affect poverty through changes in ecosystem services and infrastructure. *Proceedings of the National Academy of Sciences of the United States of America, 111*(11), 4332–4337. https://doi.org/10.1073/pnas.1307712111.

Frid, A., & Dill, L. (2002). Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology, 6*(1), 11. Retrieved from http://www.ecologyandsociety.org/vol6/iss1/art11/.

Gavilan, D., Avello, M., & Martinez-Navarro, G. (2018). The influence of online ratings and reviews on hotel booking consideration. *Tourism Management, 66*, 53–61. https://doi.org/10.1016/j.tourman.2017.10.018.

Grimm, V., Berger, U., DeAngelis, D. L., Polhill, J. G., Giske, J., & Railsback, S. F. (2010). The ODD protocol: A review and first update. *Ecological Modelling*, (221), 2760–2768. https://doi.org/10.1016/j.ecolmodel.2010.08.019.

Grumbine, R. E. (1994). Society for conservation biology what is ecosystem management? *Conservation Biology, 8*(1), 27–38. Retrieved from http://www.jstor.org/stable/2386718%5Cnhttp://www.jstor.org/page/.

Hardin, G. (1968). The tragedy of the commons. *Science, 162*, 1243–1248. Retrieved from http://www.sciencemag.org/content/162/3859/1243.full.pdf&embedded=true.

Hasselman, L. (2017). Adaptive management; adaptive co-management; adaptive governance: what's the difference? *Australasian Journal of Environmental Management, 24*(1), 31–46. https://doi.org/10.1080/14486563.2016.1251857.

Hausmann, A., Slotow, R., Fraser, I., & Di Minin, E. (2017). Ecotourism marketing alternative to charismatic megafauna can also support biodiversity conservation. *Animal Conservation, 20*, 91–100. https://doi.org/10.1111/acv.12292.

Hemson, G., Maclennan, S., Mills, G., Johnson, P., & Macdonald, D. (2009). Community, lions, livestock and money: A spatial and social analysis of attitudes to wildlife and the conservation value of tourism in a human–carnivore conflict in Botswana. *Biological Conservation, 142*(11), 2718–2725. https://doi.org/10.1016/J.BIOCON.2009.06.024.

Higham, J. E. S. (2007). *Critical issues in ecotourism: Understanding a complex tourism phenomenon*. Elsevier. Retrieved from https://www.sciencedirect.com/science/book/9780750668781.

Higham, J. E. S., Bejder, L., Allen, S. J., Corkeron, P. J., & Lusseau, D. (2016). Managing whale-watching as a non-lethal consumptive activity. *Journal of Sustainable Tourism, 24*(1), 73–90. https://doi.org/10.1080/09669582.2015.1062020.

Higham, J. E. S., & Lusseau, D. (2007). Urgent need for empirical research into whaling and whale watching. *Conservation Biology, 21*(2), 554–558. https://doi.org/10.1111/j.1523-1739.2006.00580.x.

Hoffmann, S. (2013). Property, possession and natural resource management: Towards a conceptual clarification. *Journal of Institutional Economics, 9*(1), 39–60. https://doi.org/10.1017/S1744137412000264.

Hughes, P. (2001). Animals, values and tourism — structural shifts in UK dolphin tourism provision. *Tourism Management, 22*(4), 321–329. https://doi.org/10.1016/S0261-5177(00)00070-4.

Jones, T., Wood, D., Catlin, J., & Norman, B. (2009). Expenditure and ecotourism: Predictors of expenditure for whale shark tour participants. *Journal of Ecotourism, 8*(1), 32–50. https://doi.org/10.1080/14724040802517922.

Lamers, M., van der Duim, R., van Wijk, J., Nthiga, R., & Visseren-Hamakers, I. J. (2014). Governing conservation tourism partnerships in Kenya. *Annals of Tourism Research, 48*, 250–265. https://doi.org/10.1016/J.ANNALS.2014.07.004.

Lemelin, R. H., Fennell, D., & Smale, B. (2008). Polar bear viewers as deep ecotourists: How specialised are they? *Journal of Sustainable Tourism, 16*(1), 42–62. https://doi.org/10.2167/jost702.0.

Li, K. X., Jin, M., & Shi, W. (2018). Tourism as an important impetus to promoting economic growth: A critical review. *Tourism Management Perspectives, 26*, 135–142. https://doi.org/10.1016/j.tmp.2017.10.002.

Lindsey, P. A., Nyirenda, V. R., Barnes, J. I., Becker, M. S., McRobb, R., Tambling, C. J., et al. (2014). Underperformance of African protected area networks and the case for new conservation models: Insights from Zambia. *PloS One, 9*(5), e94109. https://doi.org/10.1371/journal.pone.0094109.

Liu, W., Vogt, C. A., Luo, J., He, G., Frank, K. A., & Liu, J. (2012). Drivers and socioeconomic impacts of tourism participation in protected areas. *PloS One, 7*(4), e35420. https://doi.org/10.1371/journal.pone.0035420.

Lovejoy, T. E. (2006). Protected areas: A prism for a changing world. *Trends in Ecology & Evolution, 21*(6), 329–333. https://doi.org/10.1016/J.TREE.2006.04.005.

Lusseau, D. (2003). Effects of tour boats on the behavior of bottlenose dolphins: Using Markov chains to model anthropogenic impacts. *Conservation Biology, 17*(6), 1785–1793.

Lusseau, D., & Bejder, L. (2007). The long-term consequences of short-term responses to disturbance experiences from whalewatching impact assessment. *International Journal of Comparative Psychology, 20*, 228–236. Retrieved from http://escholarship.org/uc/item/42m224qc.pdf.

Mancini, F., Coghill, G. M., & Lusseau, D. (2017). Using qualitative models to define sustainable management for the commons in data poor conditions. *Environmental Science & Policy, 67*, 52–60. https://doi.org/10.1016/j.envsci.2016.11.002.

Mancini, F., Coghill, G. M., & Lusseau, D. (2018). Using social media to quantify spatial and temporal dynamics of wildlife tourism activities. *PloS One, 13*(7), e0200565. https://doi.org/10.1371/journal.pone.0200565.

Mayer, M., Brenner, L., Schauss, B., Stadler, C., Arnegger, J., & Job, H. (2018). The nexus between governance and the economic impact of whale-watching. The case of the coastal lagoons in the El Vizcaíno Biosphere Reserve, Baja California, Mexico. *Ocean & Coastal Management, 162*, 46–59. https://doi.org/10.1016/j.ocecoaman.2018.04.016.

McClung, M. R., Seddon, P. J., Massaro, M., & Setiawan, A. N. (2004). Nature-based tourism impacts on yellow-eyed penguins megadyptes antipodes: Does unregulated visitor access affect fledging weight and juvenile survival? *Biological Conservation, 119*(2), 279–285. https://doi.org/10.1016/J.BIOCON.2003.11.012.

Meinzen-Dick, R. (2007). Beyond panaceas in water institutions. *Proceedings of the National Academy of Sciences of the United States of America, 104*(39), 15200–15205. https://doi.org/10.1073/pnas.0702296104.

Meletis, Z.a., & Campbell, L. M. (2007). Call it consumption! Re-conceptualizing ecotourism as consumption and consumptive. *Geography Compass, 1*(4), 850–870. https://doi.org/10.1111/j.1749-8198.2007.00048.x.

Microsoft Corporation, & Weston, S. (2017). doParallel: Foreach parallel adaptor for the "parallel" package. Retrieved from https://cran.r-project.org/package=doParallel.

Microsoft Corporation, & Weston, S. (2017). foreach: Provides foreach looping construct for R. Retrieved from https://cran.r-project.org/package=foreach.

Moore, S. A., & Rodger, K. (2010). Wildlife tourism as a common pool resource issue: Enabling conditions for sustainability governance. *Journal of Sustainable Tourism, 18*(7), 831–844. https://doi.org/10.1080/09669582.2010.483281.

Morais, J., Medeiros, A. P. M., & Santos, B. A. (2018). Research gaps of coral ecology in a changing world. *Marine Environmental Research, 140*, 243–250. https://doi.org/10.1016/j.marenvres.2018.06.021.

Muboko, N., Gandiwa, E., Muposhi, V., & Tarakini, T. (2016). Illegal hunting and protected areas: Tourist perceptions on wild animal poisoning in Hwange National Park, Zimbabwe. *Tourism Management, 52*, 170–172. https://doi.org/10.1016/J.TOURMAN.2015.06.023.

Muir-Leresche, K., & Nelson, R. H. (2000). *Private property rights to wildlife: The southern African experiment*. https://doi.org/www.cei.org\rkay@padda.icon.co.za\rrn.29@umail.umd.edu.

Mutanga, C. N., Vengesayi, S., Chikuta, O., Muboko, N., & Gandiwa, E. (2017). Travel motivation and tourist satisfaction with wildlife tourism experiences in Gonarezhou and Matusadona National Parks, Zimbabwe. *Journal of Outdoor Recreation and Tourism, 20*, 1–18. https://doi.org/10.1016/J.JORT.2017.08.001.

Okello, M. M., & Yerian, S. (2009). Tourist satisfaction in relation to attractions and implications for conservation in the protected areas of the Northern Circuit, Tanzania. *Journal of Sustainable Tourism, 17*(5), 605–625. https://doi.org/10.1080/09669580902928450.

Olsson, P., Folke, C., & Hahn, T. (2004). Social-ecological transformation for ecosystem management: The development of adaptive co-management of a wetland landscape in southern Sweden. *Ecology and Society, 9*(4), 2. https://doi.org/10.5751/ES-00683-090402.

Olsson, P., Folke, C., & Hughes, T. P. (2008). Navigating the transition to ecosystem-based management of the great barrier reef, Australia. *Proceedings of the National Academy of Sciences, 105*(28), 9489–9494. https://doi.org/10.1073/pnas.0706905105.

Ostrom, E. (1990). *Governing the commons. The evolution of institutions for collective action.* Cambridge university press.

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proceedings of the National Academy of Sciences of the United States of America, 104*(39), 15181–15187. https://doi.org/10.1073/pnas.0702288104.

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science, 325*, 419–422. https://doi.org/10.1126/science.1172133.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., & Policansky, D. (1999). Revisiting the commons: Local lessons, global challenges. *Science, 284*(5412), 278–282. https://doi.org/10.1126/science.284.5412.278.

Ostrom, E., Janssen, M. A., & Anderies, J. M. (2007). Going beyond panaceas. *Proceedings of the National Academy of Sciences of the United States of America, 104*(39), 15176–15178. https://doi.org/10.1073/pnas.0701886104.

Pabel, A., & Coghlan, A. (2011). Dive market segments and destination competitiveness: A case study of the great barrier reef in view of changing reef ecosystem health. *Tourism in Marine Environments, 7*(2), 55–66. https://doi.org/10.3727/154427311X13038402065785.

Partelow, S., & Nelson, K. (2018). Social networks, collective action and the evolution of governance for sustainable tourism on the Gili Islands, Indonesia. *Marine Policy*. https://doi.org/10.1016/j.marpol.2018.08.004.

Pirotta, E., & Lusseau, D. (2015). Managing the wildlife tourism commons. *Ecological Applications, 25*(3), 729–741. https://doi.org/10.1890/14-0986.1.

Pirotta, E., New, L., Harwood, J., & Lusseau, D. (2014). Activities, motivations and disturbance: An agent-based model of bottlenose dolphin behavioral dynamics and interactions with tourism in Doubtful Sound, New Zealand. *Ecological Modelling, 282*, 44–58. https://doi.org/10.1016/J.ECOLMODEL.2014.03.009.

Poncela-Casasnovas, J., Gutierrez-Roig, M., Gracia-Lazaro, C., Vicens, J., Gomez-Gardenes, J., Perello, J., et al. (2016). Humans display a reduced set of consistent behavioral phenotypes in dyadic games. *Science Advances, 2*(8). https://doi.org/10.1126/sciadv.1600451. e1600451–e1600451.

R Core Team. (2018). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.r-project.org/.

Railsback, S. F. (2001). Concepts from complex adaptive systems as a framework for individual-based modelling. *Ecological Modelling, 139*(1), 47–62. https://doi.org/10.1016/S0304-3800(01)00228-9.

Reed, S. E., & Merenlender, A. M. (2008). Quiet, nonconsumptive recreation reduces protected area effectiveness. *Conserv. Lett., 1*(3), 146–154. https://doi.org/10.1111/j.1755-263X.2008.00019.x.

Rimella, L. (2017). *RGeode: Geometric density estimation*. Retrieved from https://cran.r-project.org/package=RGeode.

Schultz, L., Folke, C., Österblom, H., & Olsson, P. (2015). Adaptive governance, ecosystem management, and natural capital. *Proceedings of the National Academy of Sciences, 112*(24), 7369–7374. https://doi.org/10.1073/pnas.1406493112.

Scott, D., Gössling, S., & Hall, C. M. (2012). International tourism and climate change. *Wiley Interdisciplinary Reviews: Climate Change, 3*(3), 213–232. https://doi.org/10.1002/wcc.165.

Sekercioglu, C. H. (2002). Impacts of birdwatching on human and avian communities. *Environmental Conservation, 29*(3), 282–289. https://doi.org/10.1017/S0376892902000206.

Semeniuk, C. A. D., Haider, W., Cooper, A., & Rothley, K. D. (2010). A linked model of animal ecology and human behavior for the management of wildlife tourism. *Ecological Modelling, 221*, 2699–2713. https://doi.org/10.1016/j.ecolmodel.2010.07.018.

Sheppard, D. J., Moehrenschlager, A., Mcpherson, J. M., & Mason, J. J. (2010). Ten years of adaptive community-governed conservation: Evaluating biodiversity protection and poverty alleviation in a West African hippopotamus reserve. *Environmental Conservation, 37*(3), 270–282. https://doi.org/10.1017/S037689291000041X.

Sirima, A., & Backman, K. F. (2013). Communities' displacement from national park and tourism development in the Usangu Plains, Tanzania. *Current Issues in Tourism, 16*(7–8), 719–735. https://doi.org/10.1080/13683500.2013.785484.

Stankey, G., McCool, S., & Stokes, G. (1984). Limits of acceptable change: A new framework for managing the bob marshall wilderness complex. *Western Wildlands, 3*(10), 33–37. Retrieved from http://prdp2fs.ess.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5346576.pdf.

Tisdell, C., & Wilson, C. (2002). Ecotourism for the survival of sea turtles and other wildlife. *Biodiversity & Conservation, 11*(9), 1521–1538. https://doi.org/10.1023/A:1016833300425.

Velando, A., & Munilla, I. (2011). Disturbance to a foraging seabird by sea-based tourism: Implications for reserve management in marine protected areas. *Biological Conservation, 144*(3), 1167–1174. https://doi.org/10.1016/j.biocon.2011.01.004.

Watson, H., Bolton, M., & Monaghan, P. (2014). Out of sight but not out of harm's way: Human disturbance reduces reproductive success of a cavity-nesting seabird. *Biological Conservation, 174*, 127–133. https://doi.org/10.1016/j.biocon.2014.03.020.

Wickham, H., Francois, R., Henry, L., & Müller, K. (2017). *Dplyr: A grammar of data manipulation*. Retrieved from https://cran.r-project.org/package=dplyr.

Wilson, G. R., Hayward, M. W., & Wilson, C. (2017). Market-based incentives and private ownership of wildlife to remedy shortfalls in government funding for conservation. *Conserv. Lett., 10*(4), 485–492. https://doi.org/10.1111/conl.12313.

Xi, J., Zhao, M., Ge, Q., & Kong, Q. (2014). Changes in land use of a village driven by over 25 years of tourism: The case of Gougezhuang village, China. *Land Use Policy, 40*, 119–130. https://doi.org/10.1016/J.LANDUSEPOL.2013.11.014.

Ye, Q., Law, R., & Gu, B. (2009). The impact of online user reviews on hotel room sales. *International Journal of Hospitality Management, 28*(1), 180–182. https://doi.org/10.1016/j.ijhm.2008.06.011.

Francesca Mancini: Francesca is an early-career researcher and an ecological modeller at the Centre for Ecology & Hydrology. Francesca completed her bachelor's degree in Biological Sciences at La Sapienza University of Rome. She obtained an MRes degree in Applied Marine and Fisheries Ecology at the University of Aberdeen, where she also completed a PhD on sustainable management of wildlife tourism.

Ben Leyshon: Ben has a BSc in Marine and Freshwater Biology, University of London and an MSc in Rural Resource Management University College of North Wales. For 27 years Ben has worked for Scottish Natural Heritage and he is Operations Manager for the Scottish Highlands. Ben has a lead on marine issues in the north of Scotland and in particular the Moray Firth. This has involved working with port and harbour authorities, the energy sector, fisheries, recreation and tourism groups. Ben is actively involved in marine planning and policy and has represented SNH on multiple marine partnerships and management groups.

Fiona Manson: Dr Fiona Manson is a marine adviser at Scottish Natural Heritage (SNH). Fiona provides advice on the conservation of marine wildlife in Scotland, covering a range of sectors including marine tourism. She leads on the development and promotion of the Scottish Marine Wildlife Watching Code. Prior to working at SNH, Fiona has worked in fisheries research and aquaculture in Scotland, Australia and Iceland.

George M. Coghill: George is SICSA Chair in System Modelling and Professor of Computing Science at the University of Aberdeen. His main research interests are in Model-based Systems & Qualitative Reasoning, Bio-inspired Computing, and Philosophy of Information & Modelling. His research is very interdisciplinary and he has applied it to areas across a spectrum from biology and medicine, to music and sociology. He is a Fellow of the Institution of Engineering and Technology.

David Lusseau: David is Professor of Behavioural Biology at the University of Aberdeen. He works at the intersection of life, formal, and social sciences to understand how individuals make decisions when uncertain and what the consequences of those decisions are for their lives and their contributions to others. He obtained his PhD at the University of Otago in 2003. He was elected Fellow of the Royal Statistical Society in 2009, the Royal Society of Biology in 2016, and member of the Young Academy of Scotland in 2011. He currently serves on IUCN Species Survival Commission and IUCN Sustainable Use and Livelihoods Specialist Group.