How moral values influence conservation: a framework to capture different management perspectives

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Abstract

Perspectives in conservation can be based on a variety of value systems and normative postulates. Perspectives also vary between cultures. Such differences in what and how people value nature, underlie many disagreements and conflicts during the formulation and implementation of environmental management policies. Specifically, whether an action intended to promote conservation (e.g. killing cats to save birds threatened with extinction) is viewed as moral can vary among people who hold different value systems. Here, we present a conceptual framework that mathematically formalises the interplay of value systems. We argue that this framework provides a heuristic tool to clarify normative postulates in conservation approaches, and highlights how different value systems might rank various management options differently. We illustrate this by applying the framework to specific cases involving invasive alien species, rewilding, and trophy hunting; and comparing how management decisions would likely be viewed under different idealised value systems (ecocentric conservation, new conservation, and compassionate conservation). By making value systems and their consequences in practice explicit, the framework can facilitate debates on contested conservation issues, and, we hope, will ultimately provide insights into how conflicts in conservation can be reduced.

Keywords: environmental management; impact; invasive alien species; moral values; rewilding; speciesism; trophy hunting

INTRODUCTION

The consideration of the moral relationship between humans and nature and the consequent ethical obligations for conservation is relatively recent in Western culture. Environmental ethics only emerged as an academic discipline in the 1970s (Brennan & Lo, 2016) and the concepts of values and duty are now increasingly appreciated in applied ecology and conservation (e.g. Díaz et al. 2018). Respecting standards of animal ethics is now mandatory in scientific research, and most ecological journals now include an ethics statement on animal welfare. The development and application of clear guidelines to ensure animal welfare in conservation practice is similarly becoming more standard, although there is still room for improvement (Dubois et al., 2017). Despite these advances, the formulation and implementation of environmental management policies, usually aimed at conserving biological diversity and the services it provides, is, however, often associated with conflicts between different groups of stakeholders and between people with different values and interests (Crowley, Hinchliffe, & McDonald, 2017; Redpath et al., 2013). This is because any environmental management decision is shaped by the moral value system of the assessor, and different systems may advocate different management actions for the same issue, depending on the elements of the systems that are affected. An examination of how value systems could be explicitly accounted for in decision making could offer opportunities for better identifying conflicts, potentially helping to resolve them, and overall improve environmental management.

Value systems vary, and include: anthropocentric views that value human beings above everything else; sentientist views that value all sentient beings; biocentric views that value all biota (including bacteria, fungi and plants); and ecocentric views that value ecosystems (Rolston III, 2003). Each value system rests upon a set of explicit or implicit normative postulates (Table 1). Because these postulates are considered as truths, they are, as it were, non-negotiable. Therefore, if the truths of different value systems come into conflict, it is hard for a resolution to be found, and so conservationists who value biodiversity *per se* [as defined initially by Soulé (1985), called hereafter ‘traditional conservation’ (Table 1)] can be at odds with those who value biodiversity based on human welfare and economic aspects [‘new conservation’ (Kareiva & Marvier, 2012)] (Doak, Bakker, Goldstein, & Hale, 2015; Kareiva, 2014; M. Soulé, 2014).

Similarly, a recent suite of articles illustrates different perspectives on the concept of 'compassionate conservation' for animal welfare (Driscoll & Watson, 2019; Hayward et al., 2019; Wallach, Bekoff, Batavia, Nelson, & Ramp, 2018), and conflicts have arisen between different groups of stakeholders whose members share common moral values, involved in deciding how to manage invasive alien species (Crowley et al., 2017).

In the following, our aim is to conceptualize and decompose different value systems in an explicitly fashion and to explore their repercussions for the perception of conservation management actions. First, we recapitulate some archetypal value systems in environmental affairs and relate them to different conservation philosophies, including traditional conservation (M. E. Soulé, 1985), new conservation (Kareiva, 2014; Kareiva & Marvier, 2012), compassionate conservation (Wallach et al., 2018) and the four views of nature and conservation described in Mace (2014) (Table 1). Since identifying commonalities in the perspectives of different parties is key in conflict management (Redpath et al., 2013), we then introduce a formal framework to conceptualise these value systems, and examine how this framework can be applied to clarify different perspectives on specific cases. Finally, we discuss opportunities for identifying commonalities between different value systems that may enable identifying widely acceptable solutions to otherwise polarising issues.

Many different value systems of nature are held by people around the world (Díaz et al., 2018). Here, we will focus on a Western perspective of value systems that have been internationally considered for policies and the management of nature (e.g. Mace 2014). The archetypes of value systems and of conservation approaches were chosen for their importance in the past and present literature and their clear differences, to illustrate our framework. We acknowledge this is a small part of the global diversity of value systems. It would be interesting to see if the proposed framework could be usefully applied to other contexts, and identify limitations of the approach outlined here.

ENVIRONMENTAL ETHICS AND CONSERVATION

From the valuation of humans to that of ecosystems: a complex spectrum of perspectives.

The Western perspective of moral valuation encompasses a diverse set of value systems with respect to the components of nature. Traditionally, one can distinguish at least four archetypal value systems: anthropocentrism, sentientism, biocentrism, and ecocentrism (Palmer, McShane, & Sandler, 2014; Rolston III, 2003) (Table 1; Figure 1).

Anthropocentrism values nature by the benefits it brings to people through ecosystem services and more inclusively, nature's contribution to people, which encompasses biological, economic and cultural benefits humans can derive from nature (Díaz et al., 2018). One justification for anthropocentrism is that humans are (arguably) the only self-reflective moral beings, and people are therefore both the subject and object of ethics (Rolston III, 2003), therefore constituting the moral community (Table 1). In an anthropocentric system, individuals from non-human species only have value based on their benefits for humans (instrumental or non-instrumental).

Sentientism, in contrast, considers that not only sentient humans, but also other sentient animals value their life, and experience pleasure, pain, and suffering. In this view, it is the sentience (e.g. measured through cognitive ability, Singer 2009), rather than species themselves, that has intrinsic value. Sentient individuals should therefore also be part of the moral community (i.e. have an intrinsic value).

Biocentrism considers that life has intrinsic value. Although different perspectives on why life has value exist (see e.g. Taylor 2011), living organisms are valued for being alive, and not differently based on other specific characteristics.

Some ecocentric, or holistic, value systems consider that ecological collectives, such as species or ecosystems, have intrinsic value, independently from the individuals that comprise them. Species can have different values, i.e. speciesism (Table 1), and these values can be influenced by a multitude of factors, discussed in more details below.

In practice, the separation between these different normative approaches of environmental ethics is blurry, and values given to different species may vary under the same general approach. For example, biocentrism can range from complete egalitarianism between organisms, i.e. universalism (Table 1), to a graduation in value resembling sentientism. In addition, humans rarely follow a specific system objectively. Their attribution of values to individuals from different species can be deeply embedded in individual psychologies (Palmer et al., 2014; Waytz, Iyer, Young, Haidt, & Graham, 2019). Further, values and personal interests interact in making and expressing environmental moral judgements (Essl et al., 2017). Thus, the archetypes of value systems presented above rarely occur in a clear and obvious fashion in individual humans.

Moral valuation and the management of nature.

Conservation practices can historically be divided into three main categories. At one extreme, a 'nature for itself' (Table 1) view mostly excludes humans from the assessment of the efficacy of conservation management actions. This ecocentric perspective is the foundation of traditional conservation as defined by Soulé (1985), and relies on the four following normative postulates: "diversity of organisms is good," "ecological complexity is good," "evolution is good," and "biotic diversity has intrinsic value" (Soulé 1985). It historically underlies widely-used conservation tools, like the IUCN Red List of Threatened Species (IUCN, 2019). Ecocentrism is often not limited to the valuation of species, but can encompass wider collectives, i.e. assemblages of species and functions, or ecosystems. This perspective is captured, for example, by the IUCN Red List of Ecosystems (IUCN-CEM, 2016), and it is strongly reflected in international conservation-legislation such as the Convention on Biological Diversity (UNEP CBD, 2010). In the following we refer to traditional conservation as an ecocentric value system where species are intrinsically valuable (nature for itself; Figure 1) and humans are mostly excluded from management. We acknowledge that this is an archetypal view of traditional conservation, and is used here simply for illustrative purposes.

By contrast the 'nature for people' perspective (Mace, 2014) values species and ecosystems only to the extent that they contribute to the well-being of humans. These values encompass ecosystem services that help sustain human life (Bolund & Hunhammar, 1999) or economic assets (Fisher et al., 2008), and can rely on the assessment of species and ecosystem services in terms of their economic value (Costanza et al., 1997). The anthropocentric 'nature for people' perspective is exemplified by 'new conservation', also termed 'social conservation' (Kareiva, 2014; Miller, Minteer, & Malan, 2011) (Table 1). It has been argued that such an anthropocentric perspective will, by extension, help and even be necessary to conserve the aspects of nature that contribute to wellbeing. The exact set of normative postulates proposed by the proponents of new conservation is nonetheless not always clearly defined, and is likely to be interpreted differently by different people, as shown by an exchange of criticisms and responses published in the recent years (Doak et al., 2015; Kareiva, 2014; Kareiva & Marvier, 2012; M. Soulé, 2014). The need for further clarifications of the normative postulates of new conservation approaches has therefore been advocated (Miller et al., 2011).

More recently, these approaches have expanded to consider nature's contribution rather than services to people, by incorporating context-specific local and cultural knowledge into assessments and the design of conservation actions (Díaz et al., 2018). This perspective spans from a still largely anthropocentric perspective to a subtle and complex perspective on nature management, termed 'people and nature' (Mace, 2014). This view acknowledges the fact that anthropocentric aspects and traditional biodiversity conservation goals based on nature's intrinsic value are not independent but influence each other, and can go as far as considering that humans and non-human entities have reciprocal obligations (Díaz et al., 2018). The necessity to account for the interdependence between the health of nature and human wellbeing is also advocated in the United Nations Sustainable Development Goals (Weitz, Carlsen, Nilsson, & Skånberg, 2018). Similarly, "nature-based solutions" is an approach endorsed by the IUCN, which aims at protecting, sustainably managing, and restoring natural or modified ecosystems, to address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits (Cohen-Shacham, Walters, Janzen, & Maginnis, 2016). The difference compared to new conservation approaches therefore lies in the fact that it simultaneously fits within the anthropocentric and ecocentric systems, rather than considering that the latter will be addressed by focusing on the former (see Section "Nature despite/for/and people" below for details).

Finally, a recently developed approach, coined 'compassionate conservation' (Table 1; Ramp and Bekoff 2015, Wallach et al. 2018), advocates for environmental management actions that do not inflict suffering for sentient animals, as a manifestation of virtue by humans (Table 1 and section 5 below). Although they acknowledge the value of all wildlife individuals and collectives, compassionate conservationists reject notions of collectivism and nativism (Table 1).

FRAMING MORAL VALUES FOR CONSERVATION

Many of the debates in conservation are grounded in different world views in which elements of the four archetypes presented above may be mixed and influenced by cultural norms, economic incentives etc. in a way that is rarely clearly reflected (Essl et al., 2017). Here we propose that conceptualising different world views, using mathematical formulation, provides one method to clarify moral discourses in conservation by making the underlying value systems and their normative postulates transparent to all participants of the discourse. Such clarification should help identify and facilitate the discussion of shared values and incompatibilities between different environmental policies and management options (Miller et al., 2011). In a similar vein, Parker et al. (1999) proposed a mathematical framework for assessing the environmental impacts of alien species. This work was highly influential in the conceptualisation of biological invasions (being cited more than 1,900 times until May 2020 according to Google Scholar), and more recent developments on the conceptualisation of the impact of invasive alien species are illustrated by semi-quantitative risk assessment schemes that take into account – and make explicit – quantifiable impacts, uncertainty, and normative dimensions (e.g. Blackburn et al. 2014, Bacher et al. 2018).

Aside from the value systems described above, two main normative theories are relevant to decision making in environmental management: consequentialism and deontology. Consequentialism aims at choosing an action to optimise an objective function, which can be determined based on a given value system. In other words, consequentialism aims at maximising the 'greater good' (e.g. maximising the average wellbeing of people). In contrast, deontology considers that some actions are intrinsically morally wrong or right based on specific criteria, and that decisions should be based on the moral status of actions. These often come into conflict, e.g. is it acceptable to hurt a few to improve the wellbeing of many. In practice, both theories have their limitations, and a combination of both can offer a solution to complex situations (Alexander & Moore, 2016). A third normative theory is virtue ethics, which emphasises the role of virtue or moral character to make a decision. The distinction with the other two theories is nonetheless less clear, as virtue is also a common concern in consequentialist and deontological perspectives (Nussbaum, 1999; Varner, 2008).

Here, we propose a mathematical formalisation approach to conceptualise the appropriateness of environmental management actions, decomposed into the consequences and morality of an action, and argue that it can account for different value systems, including the anthropocentric, sentientist, biocentric and species-based ecocentric systems (see Appendix S1 for an extension to ecocentrism beyond species and considering wider collectives, i.e. ecosystems), while also accounting for cultural and personal perspectives. We believe that while formalising the different approaches of environmental ethics has specific limitations (discussed below), it is useful for practical and theoretical reasons, as it allows to clarify normative postulates, and to evaluate differences in the moral implications of alternative decisions. From a consequentialist perspective, a formalisation approach must combine both the impact of an action on the different species or individuals involved and the value given to said species and individuals under different value systems. Consequentialism can then be combined with considerations about the morality of actions.

The general Equation 1 provides the core principles for a mathematical formalisation of environmental ethics upon which more complex perspectives can be built, as we demonstrate below. The appropriateness of the management action (A) under a given value system, sought to be maximized, can be defined as:

$$A = \frac{M}{C} = \frac{\prod(M_1,\ldots,M_n)}{\sum_{species\ s} \bar{I}_s \times V_s \times {N_s}^a} \qquad \text{Eq.1}$$

where M is its morality (using $M_1,\ldots,M_n$ to assess morality based on multiple criteria, see below), and C represents its consequences. C is a function of the following parameters. $\bar{I}_s$ is a function (e.g. mean, maximum, etc.) of the impact (direct and indirect) resulting from the management action on all individuals of species *s*, V_s is the inherent value attributed to an individual of species *s* under a value system (objective or subjective), N_s is the abundance of species *s*, and *a* determines the importance given to a species based on its abundance or rarity (and enables to account for the importance of a species rather than an individual, see below). The unit of *A* depends on how other parameters are defined, which themselves depend on the value system considered.

The parameter *a* can take both positive and negative values. A value of 1 would mean that the same importance is given to all individuals in the moral community (Table 1) and be typical of individual-centred value systems, i.e. anthropocentrism, sentientism, and biocentrism. As a result, the more individuals in a population, the more the population would weigh on the outcome. As *a* decreases towards 0, the correlation between the value of a species and its abundance decreases. For $a = 0$, the consequence of a management action becomes abundance-independent. $a = 0$ therefore allows for an assessment of the appropriateness of management with respect to species, rather than to individuals, which corresponds to ecocentrism. For $a < 0$, rare species would be valued higher than common species (or the same impact would be considered to be higher for rare species), for example due to the higher risk of them going extinct. The lower the value of *a*, the more species rarity influences the outcome. Under complex perspectives, *a* may differ depending on the species (see section “Conceptualising traditional and new conservation” below for an example). $a > 1$ would give a disproportionate weight to abundant species, which are often important for providing ecosystem services (Gaston, 2010).

The impact *I* as defined in this framework (see Table 1) can vary depending on the local context (and the cultural context for humans). It can be limited to the death of individuals or the extinction of species (for $a = 0$), but also to animal welfare, biophysical states, etc. *I* encompasses both the direct impact of a management action, and its indirect impact resulting from biotic interactions. One would therefore need to define a baseline corresponding to: 1) the lowest possible measurable level of impact (e.g. being alive if death is the only measure of impact, or no sign of disease and starvation for biophysical states; this would obviously be more complicated for welfare), so that *I* would only be positive; and 2) the duration over which to measure such impact. The exact quantification of impact will also be influenced by different value systems and personal subjectivity, as some impacts may be considered incommensurable (Essl et al., 2017). Which impacts are incommensurable may depend on the value system (e.g. death of non-human sentient individuals vs. human health; Table 2). The range of values for I would therefore need to be established, for example using a grading system (see section 7 for some discussions on this issue). The average impact $\bar{I}_S$ could be used as a measure at the species-level, as different individuals may experience different impacts, if the management action targets only part of a given population, for example. Using the average impact is not without shortcomings though, and other measures such as the maximum impact experienced by individuals, or more complex functions accounting for the variability of impacts and values across individuals of a same species, and of different types of impact, may be used.

The inherent value V_s accounts for the fact that individuals of different species would have different values under different value systems, i.e. speciesism (Table 1). In mathematical terms, different value systems will be characterised by different distributions of V_s (Figure 1). The inherent value given to an individual from a particular species by a person or a group of persones will be influenced by its intrinsic value as defined by a value system, but will also likely be influenced by many other subjective factors. These factors include, for example, charisma (Courchamp et al., 2018; Jarić et al., 2020), anthropomorphism (Tam, Lee, & Chao, 2013), organismic complexity (Proença, Pereira, & Vicente, 2008), neoteny (Stokes, 2007), cultural importance (Garibaldi & Turner, 2004), religion (Bhagwat, Dudley, & Harrop, 2011), or parochialism (Waytz et al., 2019) (Table 1). Inherent values are therefore subjective and likely to vary in time and across locations, and depend not only on the characteristics of the species but also on those of the assessor. For example, some alien species that did not have any value prior to their introduction have been incorporated in local cultures, therefore providing them a novel and higher inherent value such as horses being linked to a strong local cultural identity in some parts of the USA (Rikoon, 2006). Note that we distinguish between the inherent value given to a species here, which is captured by V_s and is determined by value systems described above and the assessor's subjectivity, and the utilitarian value that a species has due to its impact, through exploitation or biotic interactions, on the species with intrinsic values. The utilitarian value is accounted for by $\bar{I}_S$ (Figure 1), because a management action can change the impact a species has on another, therefore representing the indirect impact of a management action, as explained above.

Many factors can therefore influence the assessment of the $\bar{I}_s$ and V_s variables (see Table 2 for a list of important factors, and section "Unresolved questions and limitations" below for some discussions on these aspects). For example, the indirect impact of a management action on a multitude of species resulting from complex biotic interactions is difficult to precisely understand and quantify. Concepts such as keystone species (Mills, Soulé, & Doak, 1993) can then offer a convenient way to overcome such complexity by modifying V_s rather than $\bar{I}_s$. Let us assume that a management action will have a direct impact on a keystone species, which will result in indirect impacts on multiple other species with inherent values. Increasing the value of the keynote species can result in the same assessment of C as to explicitly model the biotic interactions and compute the resulting indirect impacts $\bar{I}_s$.

Finally, the morality M of an action can vary between 0 and 1, where 0 would mean that an action is morally unacceptable, and 1 would indicate a fully acceptable (moral) action. M may take intermediate values between 0 and 1: for example, although killing an animal can be considered morally problematic (i.e. $M < 1$), it might be acceptable in some situations (i.e. $M > 0$), but killing an animal using a painful poison would likely be deemed less moral than killing an animal using a non-painful method. If multiple criteria are used or different actions are implemented, the product $\prod(M_1, \dots, M_n)$ for all criteria and actions can be used, as it would result in 0 if one criterion is not fulfilled or one action is immoral. M can also be defined to change depending on appropriateness thresholds (Alexander & Moore, 2016), i.e. account for the limits of basing a decision on the morality of an action if the consequences are too costly. In this situation, M can be increased if no management action produces an appropriateness A over an acceptable threshold (the value of the threshold depends on the specific case and chosen measure of impact I). The number of criteria to determine the morality of different actions will be extremely context- and user-dependent, and it would be difficult to provide general insights in the context of this manuscript. In the following, we will therefore focus mostly on a consequentialist perspective.

Note that Equation 1 does not produce absolute, but relative values. That is, the appropriateness A of a management action under different value systems cannot be directly compared with each other. This is because the unit and range of values of C and M can vary between value systems, as they may consider different measures of impact. Instead, Equation 1 can be used to rank a set of management actions (including non-action, which can be used as a baseline) for each value system based on their assessed appropriateness, to identify management actions representing consensus, compromises or conflicts amongst value systems. In the following, we show how, even despite the difficulty to quantify the variables described above, this framework can be used as a heuristic tool to capture the implications of considering different value systems for determining the appropriateness of a conservation action, and to better understand conservation disputes.

NATURE DESPITE/FOR/AND PEOPLE

Over the past decade there has been some debate between proponents of traditional conservation, and those of new conservation (Table 1), as each group assumes different relationships between nature and people. Here, we show how the formal conceptualisation of Equation 1 could help clarifying the position of the new conservation approach in response to its criticisms (Kareiva, 2014).

As an ecocentric value system, in traditional conservation, consequences C in the genereal Equation 1 can be expressed as follows:

$$C = \sum_{\text{species s (excluding humans)}} \bar{I}_s \times V_s \times {N_s}^{\text{a<0}} \qquad \text{Eq. 2}$$

The traditional conservation in Equation 2 emphasises that “diversity of organisms is good” and that “biotic diversity has intrinsic value” (M. E. Soulé, 1985). Here, we propose to assign a stronger weight to rare species, indicated by the parameter $a < 0$, to account for the fact that rare species are more likely to go extinct, which should result in high consequences under ecocentrism because it would decrease the “diversity of organisms”. Evolution is not explicitly accounted for in the framework, but emphasising the importance of rare species should lead to fewer extinctions and may increase the chance of species with high evolutionary potential to remain within the species pool. Ecological complexity is not explicitly considered either, but could be accounted for by attributing higher values to species with rare traits and specific functional roles.

In contrast, new conservation considers that maximising appropriateness under an anthropocentric value system, i.e. following a nature for people approach in which non-human species and collectives are only considered to have a utilitarian value, will also increase appropriateness under ecocentrism. New conservation considers that stakeholders tend to have an anthropocentric value system, and that conservation approaches that do not incorporate such a perspective will likely not succeed (Kareiva, 2014; Kareiva & Marvier, 2012). Therefore, although the aim of new conservation appears to be similar to traditional conservation approaches, its literal application follows anthropocentric principles.

Maximising appropriateness under an anthropocentric view implies that Equation 1 should be modified to only incorporate humans. Species are only conserved due to their utilitarian value, i.e. their effect on I for humans, rather than based on an inherent value V. Different groups of stakeholders are nonetheless likely to be impacted differently (e.g. monetary benefits / losses vs. changes in access to nature, accounting for cultural differences, etc.), and we propose the following extension of Equation 1 to account for this variability:

$$C = \sum_{\text{stakeholders } t} \bar{I}_t \times V_t \times N_t \qquad \text{Eq.3}$$

where $\bar{I}_t$ is the average impact of management on the group of stakeholders t, including indirect impacts through the effect of management of non-human species, V_t is the value of the group of stakeholders t, and N_t is its abundance. Note that including inherent values V_t in Equation 3 does not imply that we consider that different humans should be valued differently, but that is a view that some people have, and this needs to appear here to capture the full spectrum of perceived consequences of a management action.

The assumption of new conservation is that a management action that minimizes consequences C in the new conservation Equations 3 will also decrease C in the traditional conservation Equations 2. This assumption relies on the developments of functional ecology and its integration with community ecology over the last few decades (Loreau, 2010; Mace, 2014). Especially, the link between biodiversity and ecosystem services has been demonstrated, even if many unknowns remain (Cardinale et al., 2012; Chivian & Bernstein, 2008), implying that high biodiversity can support the provision of ecosystem services to humans. Nonetheless, such an approach will necessarily distinguish between "useful" species and others, and impacts will be perceived differently by different groups of stakeholders. This assumption is therefore likely to be context dependent.

In contrast, by seeking simultaneous human well-being and biodiversity benefits, people and nature approaches combine all these aspects into a single equation as follows, capturing a more diverse set of value systems than Equations 2 and 3 alone:

$$C = \sum_{\text{stakeholders } t} \bar{I}_t \times V_t \times N_t + \sum_{\text{species } s \text{ (excluding humans)}} \bar{I}_s \times V_s \times {N_s}^{\mathrm{a<0}} \qquad \text{Eq. 4}$$

Different concepts that may be difficult to directly compare in a quantified fashion are combined in the people and nature Equation 4, such as economic benefits / losses and human well-being. This is especially true when local cultural values are considered in assessments of the impact of management actions on humans, as in the most recent developments of new conservation approaches (Díaz et al., 2018). Equation 4 is therefore intended to be taken conceptually. In practice, the different terms may be considered independently, and approaches such as multi-criteria decision analyses (Huang, Keisler, & Linkov, 2011) may be used instead.

Note that we considered two extreme interpretations of traditional and new conservation, as illustrated by Equations 2 and 3, excluding either humans or non-human species. The exclusion of humans from Equation 2 corresponds to an extreme perspective of traditional conservation, championed by 'fortress conservation' (Büscher, 2016; Siurua, 2006). Similarly, new conservation is defined as purely anthropocentric in Equation 3, which is an argument of its detractors (e.g. Soulé 2014). In response to these criticisms, Kareiva (2014) explains that the purpose of new conservation is not to replace traditional conservation by an anthropocentric perspective that would use economic success as a measure of achievement (i.e. instead of C in equation 1 and 3). Similarly, some may argue that traditional conservation is not restricted to fortress conservation. Equations 2 and 3 nontheless clearly show how failing to explicitly define normative postulates for conservation approaches can lead to extreme interpretations and conflicts and to overlook nuances between perspectives. In particular, it has been argued that the normative postulates of new conservation need to be more explicitly defined (Miller et al., 2011). Our framework could help doing so, by being more explicit about how new conservation would be defined relative to the traditional conservation and the people and nature perspective in Equations 3 and 4.

THE CASE OF ANIMAL WELFARE

The question of integrating animal welfare into conservation practice (and of how to integrate it), i.e. considering a sentientist value system, is subject to debate. For example, compassionate conservation, as defined by Wallach et al. (2018) (Table 1), emphasises animal welfare and is based on the "growing recognition of the intrinsic value of conscious and sentient animals". It stipulates that "we need a conservation ethic that incorporates the protection of other animals as individuals, not just as members of populations of species but valued in their own right" (Ramp & Bekoff, 2015). Compassionate conservation for example opposes the killing of sentient invasive alien species such as cats and camels in Australia; the killing of native species predating on endangered species, such as wolves on caribou in Canada; or the killing of specific individuals to fund broader conservation goals, i.e. trophy hunting (Wallach et al., 2018).

Despite the near-universal support of conservation practitioners and scientists for compassion towards wildlife and ensuring animal welfare (e.g. Russell et al. 2016, Oommen et al. 2019, Hayward et al. 2019), the concept of compassionate conservation as presented by Wallach et al. (2018) has sparked vigorous responses (Driscoll & Watson, 2019; Hayward et al., 2019; Oommen et al., 2019). Amongst the main criticisms of compassionate conservation is that the absence of action, for example to control animal populations, can result in unintended detrimental effects and increased suffering for individuals of other or the same species (including humans), as a result of altered biotic interactions across multiple trophic levels, i.e. "not doing anything" is an active choice that has consequences (Table 2). For example, not culling individuals may result in greater numbers dying of diseases or hunger caused by over-population (e.g. ICMO2 2010), and not controlling populations of predators may result in the extirpation of prey populations. The number of individuals dying over time may be larger in the absence of action, which would result in direr consequences under our framework. Another argument is the inconsistency in the (subjective) moral valuation of individuals from different species and taxonomic groups (i.e. inconsistencies in the distribution of inherent values V) (Hayward et al., 2019; Oommen et al., 2019). Finally, some authors have pointed out the lack of clarity of the approach when facing complex situations with multiple conflicting perspectives (Rohwer & Marris, 2019).

Compassionate conservation is defined through the lens of virtue ethics (Wallach et al., 2018). The lack of a clear boundary between virtue ethics and consequentialism or deontology may explain some lack of clarity in the normative foundations of compassionate conservation. To avoid such ambiguities, in the following, we propose to discuss animal welfare from the perspective of consequentialism and morality, as captured by the general Equation 1, and we show how this approach may be coherent with the perspective of compassionate conservation, or how it may highlight the need for clarifications.

A mathematical conceptualisation of animal welfare.

A consequentialist, sentientist perspective aims at maximizing happiness, or conversely minimising suffering, for all sentient beings, an approach also termed 'utilitarianism' (Singer, 1980; Varner, 2008). That is, suffering is considered as a measure of impact (or, in mathematical terms, impact is a function of suffering, which can be expressed as $I(S)$ in Equation 1).

It has become widely accepted that animals experience emotions (de Waal, 2011), but compassionate conservation is vague on the possibility of individuals from different species experiencing different levels of suffering. A recent article on compassionate conservation suggests that the presence of sentience in an animal should be sufficient to give it the status of person, but does not seem to consider sentience as a graded concept (Wallach et al., 2020). Quantifying the suffering (or negative emotions) experienced by an individual along a one-dimensional axis requires strong simplification of this complex concept (see Shriver 2006 and Bermond et al. 2008 for different conclusions about the capacity of animals to experience suffering). Nonetheless, emotions have been shown to be linked to cognitive processes (Boissy & Lee, 2014), which differ greatly among species (MacLean et al., 2012), and behavioural approaches have been used to evaluate emotional responses (e.g. Désiré et al. 2002). We therefore postulate that such quantification is conceptually feasible in the context of the heuristic tool presented here. In a utilitarian approach, the inherent value of a species would therefore be a function of its capacity to experience emotions and suffering, which can be expressed as $V(E)$ instead of V in Equation 1.

Assuming that quantifying both the emotional capacity of individuals belonging to different species and the suffering they experience under a given conservation action can be done, the consequences of this conservation action under the objective of minimising suffering can be computed as:

$$C = \sum_{\text{species } s} \overline{I(S_s)} \times V(E_s) \times {N_s}^{a=1} \qquad \text{Eq.5}$$

where $\overline{I(S_s)}$ is a function (e.g. mean or maximum) of the suffering experienced by individuals from species s, used to assess the impact. Note that the suffering of an individual may be assessed through a wide variety of proxies, including access to food and water, death, number of dead kin for social animals, physiological measurements of stress hormones, etc. E_s is the emotional capacity of individuals belonging to species s (assuming uniform intra-specific emotional capacity for simplification), that allows to assess its value $V(E_s)$. Although $V(E_s)$ should be measured in an objective fashion, many factors may influence the relationship between the inherent value and the emotional capacity of a species. For example, high empathy (Table 1) from the observer will tend to make the distribution uniform (therefore in line with the perspective defended by Wallach et al. 2020), whereas anthropomorphism and parochialism (Table 1) may lead to higher rating of the emotional capacities of species phylogenetically close to humans or with which humans are more often in contact, such as pets. Finally, N_s is the abundance of species *s* in the area affected by the management action. Here, we assumed that $a = 1$, therefore giving equal importance to any individual regardless of the abundance of its species.

Although compassionate conservation is not a consequentialist approach, we believe the sentientist Equation 5 aligns with its main tenets. The minimization of suffering and considering individuals equally irrespective of species abundance encompasses the "do not harm" and "individuals matter" tenets of compassionate conservation. The "inclusivity" tenet "acknowledges the intrinsic value of all wildlife individuals and collectives" (Wallach et al., 2018). However, in the same article, Wallach et al. (2018) reject the notion of collectivism. It is therefore difficult to clearly identify how inclusivity would be incorporated in Equation 5 due to the lack of clarity in the definition. Finally, the "peaceful coexistence" tenet, which corresponds to "critically examine and in many cases modify one's own practices, rather than pursuing acts of aggression against wildlife individuals", is addressed by the evaluation of Equation 5.

In practice, Equation 5 may be combined with a perspective considering the morality of culling sentient species ($M \leq 1$), by discussing if one may consider a threshold in total suffering (quantified by C) over which culling is acceptable or not (an issue related to the concept of "moral residue"; Batavia et al. 2020). This threshold is likely to vary with each person, and we will focus on the consequentialist perspective in the following.

Assessing suffering in the presence and absence of conservation management actions.

If we assume that the distribution for the emotional capacity E of individuals from different species can and has been quantified prior to analyses (Equation 5), the remaining challenge is to assess the suffering experienced by individuals. The short-term suffering resulting from pain and directly caused by lethal management actions, such as the use of poison to control invasive alien species (e.g. McIlroy 1981) or the use of firearms and other mechanical device to cull native species threatening other native species (e.g. wolves threatening caribous in Canada; Proulx et al. 2016) or humans (e.g. shark attacks; Gibbs and Warren 2015), is the most straightforward type of suffering that can be assessed, and is usually sought to be minimised in all conservation approaches. However, impacts can take various forms, and commensurability can be an issue

(Table 2). Distinguishing between lethal actions and non-lethal suffering is, in particular, morally complex. For example, non-lethal suffering can result from unfavourable environmental conditions (e.g. leading to food deprivation) and occur over long periods, while lethal actions could be carried out in a quick, non-painful fashion (see the example of the Oostvaardersplassen nature reserve below, or the use of culling to prevent epidemics, Shao et al. 2018), but may be deemed immoral ($M = 0$). One of the main criticisms of compassionate conservation is that the assessment of suffering is restricted to direct relationships, i.e. to the suffering directly caused by humans to animal individuals, while neglecting lethal and non-lethal suffering resulting from biotic interactions between non-human species, or indirect interactions through the abiotic environment.

We therefore advocate for a conceptual approach that takes into account indirect consequences of management actions within a certain timeframe; similarly, uncertainty should be considered (Table 2). Direct and indirect biotic interactions may be explicitly modelled to quantify the impact on animals and therefore their suffering. Simulation models can also make projections on how populations may change in time, therefore enabling to account for future suffering. For example, absence of management of feral camels in Australia would likely lead to ecosystem degradation in which the individuals of co-occurring resident species increasingly have difficulties in finding resources and therefore to increased suffering for these species (Brim-Box et al., 2010; Edwards, Zeng, Saalfeld, & Vaarzon-Morel, 2010).

Are traditional conservation and animal welfare compatible?

In their response to Wallach et al. (2018), Driscoll & Watson (2019) refer to Soulé's (1985) normative postulate that "diversity of organisms is good", which differs indeed from the postulates of compassionate conservation, but without exploring further why diversity of organisms is good. They then provide examples when traditional and compassionate conservation would advocate for opposite approaches, conveying the message that the two approaches are necessarily incompatible. It has nonetheless been argued that sentientism and ecocentrism are not fully incompatible (Varner, 2011). The relationship between biodiversity and animal suffering can be formalised more clearly using the traditional conservation and the sentientist Equations 2 and 5, to explore if the same management action can minimize the consequences evaluated using the two equations (see also Supplementary material S2 for the application of the framework to theoretical cases). The main difference with the traditional vs new conservation debate here is that Equations 2 and 5 share a number of species, whereas the new conservation Equation 3 only contains humans, which are excluded from Equation 2. Even though the variables of Equation 5 differ from those of Equation 2 (V and I are computed differently, and the value of a is different), there may be a higher chance that these equations will vary in similar way for different management actions due to their similar structure. Clarifying how compassionate conservationists would define the variables of Equation 5 (especially in terms of the direct and indirect impacts through suffering) would be necessary not only to better defend this value system, but would also clarify the criticisms of some of their detractors and could eventually help identify some common ground.

One issue that may be irreconcilable between traditional conservation and approaches based on sentientism (besides the moral aspects linked to the culling of sentient animals) is the fate of rare and endangered species with limited or no sentience. Under utilitarian sentientism, the conservation of non-sentient species ranks lower (if at all) than the conservation of sentient species, and consequently they are not included in Equation 5. For example, endangered plant species that are not a resource for the maintenance of sentient populations (and therefore do not influence Equation 5, contrary to plants that are resources for a sentient species s and therefore influence the value of impact $I(S_s)$) would receive no attention, as there would be few arguments for their conservation. On the contrary, traditional conservation would focus on their conservation, as they would have a disproportionate impact in Equation 2, due to low abundance leading to a high value for $N^{a<0}$.

Finally, it is important to note that the current body of knowledge shows that the link between biodiversity and animal welfare mentioned above especially applies to the increase of native biodiversity. The local increase of biodiversity due to the introduction of alien species (which may only be temporary due to extinction debt; Kuussaari et al. 2009) often results in reduced quantity and quality of ecosystem functioning (Cardinale et al., 2012). Therefore, it is important to distinguish between nativism (criticised by advocates of compassionate conservation), which considers that native species intrinsically have higher value than alien species (Table 1), and the proven detrimental effects of *invasive* alien species on biodiversity and ecosystem functioning and services (Bellard, Cassey, & Blackburn, 2016). Nativism would result in shifting native species to the left regarding the distribution of $V(E)$ (Figure 1), whereas in the second case, insights from science on the impact of invasive alien species would modify the distribution $I(S)$ rather than the distribution $V(E)$. This can also apply to native species whose impacts on other species, such as predation, are increased through environmental changes (Carey, Sanderson, Barnas, & Olden, 2012).

UNRESOLVED QUESTIONS AND LIMITATIONS

This framework is designed as a heuristic tool to clarify normative postulates, and to qualitatively evaluate differences in outcomes of alternative decisions. The approach shares similarities with mathematical approaches used in conservation triage (Bottrill et al., 2008), but has two crucial differences. First, conservation triage equations use an ecocentric perspective with relatively easily measurable variables. Bottrill et al. (2008) provided an example using phylogenetic diversity as a measure of value V, and a binomial value b to quantify biodiversity benefit that can be interpreted as the presence or absence of a species (i.e. $I = 1 / \mathrm{b}$). Because it is ecocentric, local species abundance is not considered, which corresponds to setting $a = 0$. In this example, consequences (C) in the general Equation 1 are therefore defined simply by V / b.

In contrast, our framework allows much more flexibility to encompass a range of value systems, as shown above. Given that the data needed for quantifying parameters of Equations 1 to 5 related to value, impact, emotional capacity and suffering are scarce and often very difficult to measure, this framework in its current form would nonetheless be difficult to use as a quantitative decision tool to evaluate alternative management actions, contrary to triage equations. Rather, our equations decompose the question underlying many controversies around management decisions in conservation: what or who is valued, how, and by how much?

There are nonetheless a number of approaches that may be used to develop quantification schemes for the different parameters of the framework. Grading systems may be developed to assess impact and suffering based on various indicators, including appearance, physiology, body function, and behaviour (Broom, 1988). For assessing the value of different species, questionnaires may be used to assess how different species are valued by people, and influenced by their social and cultural background, similar to what has been done to assess species charisma (e.g. Colléony et al. 2017, Albert et al. 2018). It will nonetheless be important to acknowledge the corresponding uncertainties in the assessment of impact and value, differences in perception among societal groups for different taxa and potential shifts in perception over time (Table 2).

The second difference from conservation triage is that the latter considers additional criteria that were not addressed here, including feasibility, cost, and efficiency (including related uncertainties). The combination of these different perspectives calls for appropriate methods to include them all in decision making, which can be done using multi-criteria decision analyses (Huang et al., 2011). Here, good communication and transparency of the decision process is key to achieve the highest possible acceptance across stakeholders, and to avoid biases in public perception (see case studies below for examples).

The issue of spatial and temporal scale also warrants consideration (Table 2). In the case of a species that may be detrimental to others in a given location but in decline globally, the spatial scale and the population considered for evaluating the terms of Equations 1 to 5 will be crucial to determine appropriate management actions. Similarly, management actions may also result in a temporary decrease in welfare conditions for animals, which may increase later on (Ohl & Van der Staay, 2012), or the impacts may be manifested with a temporal lag. In that case, determining the appropriate time period over which to evaluate the terms of Equations 1 to 5 will not be straightforward. Impacts may also not have the same importance depending on whether they occur in the short- or long-term, especially since long-term impacts are harder to predict and involve higher uncertainty. Discount rates (Table 2) may therefore be applied, in a similar way they are applied to the future effects of climate change and carbon emissions (Essl, Erb, Glatzel, & Pauchard, 2018), or to assess the impact of alien species (Essl et al., 2017).

Equations 1 to 5 assume that all individuals from a given species have the same value or emotional capacities (or use the average of the value across individuals). However, there may be intraspecific differences in value, and such variations may be important for conservation. For example, trophy hunters might prefer to hunt adult male deer with large antlers. Reproductively active individuals contributing to population growth/recovery may be given a higher value. Intraspecific value may also vary spatially, for example comparing individuals in nature reserves or in highly disturbed ecosystems. Equation 1 may therefore theoretically be adapted to use custom groups of individuals with specific values within species, similar to Equation 3 (although in Equation 3, impact varied between groups of stakeholders but values were assumed to be the same).

Finally, it is crucial to account for biotic interactions in our framework to comprehensively assess the indirect impacts of management actions on different species (Table 2). Some species with low values in a certain value system may have little weight V in Equation 1, but they may be crucial for assessing the impact I on other species, because of pollination, source of food, etc. These biotic interactions will therefore determine the time frame over which the framework should be applied, as impacts on one species at a given time may have important repercussions in the future. These biotic interactions can be complex, and multiple tools, such as simulation models and ecological network analyses can be used to address them.

CASE STUDIES ILLUSTRATING ETHICAL CONFLICTS IN CONSERVATION DECISIONS

In the following, we present three case studies where conservation actions have either failed, had adverse effects, or were controversial, and we explore how our framework can shed some light on these situations.

Invasive alien species management: the case of the alien grey squirrel in Italy

Invasive alien species management is a common source of conflict between different stakeholders using different value systems, such as conservationists and animal right activists (Perry & Perry, 2008). These conflicts can be decomposed into distinct successive phases characterised by the intensity of the conflict, starting at low intensity in the form of disagreement before escalating as a result of polarisation of opinions, and eventually either reaching a destructive phase, or de-escalating if conflict management is adequately implemented (Crowley et al., 2017). Communication and inclusive engagement are key to conflict de-escalation. This requires the capacity to acknowledge and conceptualise the value systems of the different parties.

The grey squirrel (*Sciurus carolinensis*) is native to North America and was introduced in various locations in Europe during the late nineteenth and the twentieth century (Bertolino, 2008). It threatens native European red squirrel (*Sciurus vulgaris*) populations through competitive exclusion, and is also a vector of transmission of squirrel poxvirus in Great Britain (Schuchert, Shuttleworth, McInnes, Everest, & Rushton, 2014). Furthermore, it has wider impacts on woodlands and plantations, reducing value of tree crops, and potentially affects bird populations through nest predation (Bertolino, 2008).

Based on the impacts of the grey squirrel, an eradication campaign was implemented in 1997 in Italy, with encouraging preliminary results (Genovesi & Bertolino, 2001). However, this eradication campaign was halted by public pressure from animal rights movements. The strategy of the animal rights activists consisted in (i) humanising the grey squirrel and using emotive messages (referring to grey squirrels as “Cip and Ciop”, the Italian names of the Walt Disney “Chip and Dale” characters) and (ii) minimising or denying the effect of grey squirrel on native taxa, especially the red squirrel (Genovesi & Bertolino, 2001). In addition, the activists did not mention, (iii) the difference in abundance between a small founding population of grey squirrels that could be eradicated by managers, and a large population of native red squirrels that would be extirpated or severely impacted by grey squirrels if control was not implemented.

Genovesi & Bertolino (2001) explain that the main reason for the failure of the species management is a different perspective on primary values: the eradication approach was underlain by species valuation, following traditional conservation, whereas the animal right activists and the public were more sensitive to animal welfare. However, the application of our framework reveals some inconsistencies in the animal right activists’ arguments that could have been used to advocate for the eradication approach. Translating this situation in our framework indicates that (i) the humanisation of the grey squirrel consists of increasing the perception of its emotional capacity $E_{gs} > E_{rs}$ (and therefore $V(E_{gs}) > V(E_{rs})$), (ii) minimising the impact of grey squirrel is equal to restricting the time scale to a short one and to likely minimising the amount of suffering S caused by grey squirrels on other species, i.e. $S_{gs} = S_{rs}$ (and therefore $I(S_{gs}) = I(S_{rs})$) without management and $S_{gs} > S_{rs}$ (and therefore $I(S_{gs}) > I(S_{rs})$) under management, and (iii) not mentioning differences in species abundance corresponds to setting $a = 0$. Following these three points, the consequences under management $C_m = I(S_{gs}) \times V(E_{gs}) + I(S_{rs}) \times V(E_{rs})$ are higher than without management, due to the increase in $V(E_{gs})$ and $I(S_{gs})$.

The framework can thus be used to provide recommendations for what the advocates for the eradication campaign would have needed to have done: i) increase the value E_{rs} of red squirrels in a similar way as what was done for grey squirrels, so that their relative values compared to grey squirrels would remain the same as before the communication campaign by the animal right activists; ii) better explain the differences in animal suffering caused by the long-term presence of the grey squirrel compared to the short-term, carefully designed euthanasia protocol, would avoid a subjective perception of the distribution of S; and iii) highlight the differences in the number of individuals affected. The consequences would then be computed as $C = V(E_{gs}) \times I(S_{gs}) \times N_{gs} + V(E_{rs}) \times I(S_{rs}) \times N_{rs}$. In that case, assuming the same suffering through euthanasia for grey squirrels vs. other causes caused by grey squirrels for red squirrels, for simplification, and the same value to individuals of each species (i.e. avoiding nativism), the mere differences $N_{rs} > N_{gs}$ in abundance would lead to a higher value of C without management. This would be even increased by extending the impacts of grey squirrels to other species, as mentioned above.

A more fundamental issue, however, is that in some value systems it would not be acceptable to actively kill individuals, even if that meant letting grey squirrels eliminate red squirrels over long periods of time. This is in essence the deontological viewpoint, and would correspond to setting M = 0 / 1 in Equation 1. The reluctance to support indirectly positive conservation programs is a common issue (Courchamp et al., 2017). Whether an acceptable threshold for M could be determined through discussion would depend, in part, on the willingness of the affected parties to compromise.

De-domestication: the case of Oostvaardersplassen nature reserve

De-domestication, the intentional reintroduction of domesticated species to the wild is a recent practice in conservation that raises new ethical questions related to the unique status of these species (Gamborg, Gremmen, Christiansen, & Sandoe, 2010). Oostvaardersplassen is a Dutch nature reserve where two domesticated species of large herbivores (Heck cattle, *Bos primigenius*, and konik horses, *Equus ferus caballus*) have been 'rewilded' in addition to the reintroduction of the red deer (*Cervus elaphus*) to act as landscape engineers by grazing (ICMO2, 2010). The populations increased rapidly, as natural predators are missing and population regulation was not conducted, as a result of a 'non-intervention-strategy'. The project was widely criticized when a considerable number of individuals died from starvation during a harsh winter, resulting in the introduction of population reduction by culling weak animals in order to prevent starvation (other approaches, such as the reintroduction of large predators were discarded due to lack of experience and too many uncertainties in efficiency, ICMO2 2010).

From a traditional conservation perspective, disregarding animal welfare and focusing on species diversity and ecological restoration, the project was a success. The introduction of the three herbivore species led to sustainable populations (despite high winter mortality events), and ensured stability of bird populations without the need for further interventions (ICMO2 2010), i.e. the conditions of many species were improved (the impact was lowered), leading to lower consequences C overall (Equation 2). However, as the general public tends to have a sententist perspective (Equation 5), the welfare of individuals from the three charismatic large herbivorous species became a point of conflict. Interestingly, it appears that the conflict was driven by a shift in attitude, from considering the herbivore species as a natural way to manage the grasslands to being part of the ecosystem changed the value V_s, or by the importance given to their emotional capacity E_s (Ohl & Van der Staay, 2012), therefore leading to increasing the consequences $C = V(E_s) \times I(S_s) \times {N_s}^1$ under sentientism, with S_s and N_s constant. Temporal changes in the distributions of the V and E variables should therefore be taken into account when implementing conservation management actions, and even monitored through time in a way similar to adaptive management approaches. Another possible explanation for this shift in attitude is that the notion of responsibility (Table 2) affected the morality value M in Equation 1. If culling animals can be considered acceptable in some cases ($M > 0$), it may not be the case if these individuals were purposefully introduced, leading to a decrease of M.

As a result, the reserve management has examined a number of sustainable measures to improve the welfare of individuals from the three species (therefore decreasing S_s to compensate the increase in V_s). Among those were recommendations to increase access to natural shelter in neighbouring areas of woodland or forestry, to create shelter ridges to increase survival in winter as an ethical and sustainable solution, and to use early culling to regulate populations and avoid suffering from starvation in winter (ICMO2 2010). This example shows how a combination of two complementary management actions (the rewilding of the OVP and the provision of shelter) led to minimised consequences under both the traditional conservation and the sentientist Equations 2 and 5, whereas only rewilding would increase consequences under Equation 5. Culling may still face opposition based on moral arguments though.

Trophy hunting

Trophy hunting, the use of charismatic species for hunting activities, has been argued to be good for conservation when revenues are reinvested properly into nature protection and redistributed across local communities, but faces criticisms for moral reasons (Di Minin, Leader-Williams, & Bradshaw, 2016; Lindsey, Frank, Alexander, Mathieson, & Romanach, 2007). The action of killing some individuals to save others might be incompatible with a deontological perspective, but, assuming a consequentialist perspective, the framework can be applied to formalise the assessment of different management options.

In traditional conservation, trophy hunting is desirable if it directly contributes to the maintenance of species diversity. The potential of trophy hunting to contribute to the maintenance of biodiversity is via creating economic revenues, i.e. an anthropocentric perspective, and it therefore falls under the umbrella of new conservation. In theory, trophy hunting should lead to the increase of both the traditional and new conservation (Equations 2 and 3), and therefore affect both segments of the 'people and nature' Equation 4, as they are in this case not independent from each other (Lindsey, Roulet, & Romanach, 2007). Many social and biological factors currently affect the efficacy of trophy hunting as a conservation tool. Corruption and privatisation of the benefits have sometimes prevented the revenues to be reinvested into conservation, but also to be redistributed across local communities, whereas doing so has been shown to increase their participation in conservation actions with proven benefits for local biodiversity (Di Minin et al., 2016). In other words, a decrease in the first anthropocentric term of equation 4 leads to a decrease in the second econcentric term too. In addition, trophy hunting can lead to unexpected evolutionary consequences (Coltman et al., 2003), overharvesting of young males (Lindsey, Frank, et al., 2007), and disproportionate pressure on threatened species (Palazy, Bonenfant, Gaillard, & Courchamp, 2011, 2013, 2012) and therefore to population declines and potential detrimental effects on biodiversity, i.e. in the second component of Equation 4. Despite these issues, it has been argued that banning trophy hunting may create replacement activities that would be more detrimental to biodiversity (Di Minin et al., 2016).

From an animal welfare perspective, trophy hunting appears to be in direct contradiction with a decrease in animal suffering, and has been criticised by proponents of compassionate conservation (Wallach et al., 2018). However, as for the culling of invasive alien species, we suspect the story is more complex. To our knowledge, there have not been many studies comparing the welfare of individual animals to quantify the elements of the sentientist Equation 5 (for example assessed through access to resources) in areas where trophy hunting is practiced and where it is not (we are not considering canned hunting here, the practice of farming animals for the specific purpose of being hunted). Given the links between biodiversity and animal welfare described above, it seems plausible that good practice in trophy hunting may benefit the welfare of individuals from other and from the same species. There may also be direct benefits if money from trophy hunting is reinvested in protection measures against poaching. However, more precise quantifications would be needed when incorporating the morality M of hunting in the equation, and especially the notion of threshold (i.e. how much improvement to all other individuals is necessary to consider trophy hunting acceptable).

CONCLUSIONS

A variety of value systems exist in conservation, which are based on different underlying normative postulates and can differ between stakeholders, resulting in differing preferences for conservation practices among people. Here, we have proposed a framework with a formal set of equations to conceptualize and decompose these different perspectives. In this framework, the different value systems supported by different conservation approaches follow the same structure, but can differ in the variables that are used, and in the values they are taking. While such formalisations by necessity do not capture the full range of complex and nuanced real-world situations in environmental decision-making, they provide a method to make their underlying value systems and the resulting conflicts explicit and transparent, which is essential for the planning and implementation of pro-active management. The search for consensus in conservation can be counter-productive and favour status-quo against pro-active management (Peterson, Peterson, & Peterson, 2005), however our framework may help identify hidden commonalities between seemingly antagonistic stances. We hope that, by doing so, this framework can foster debates on contested conservation issues, and will ultimately contribute to a broader appreciation of different viewpoints. In an increasingly complex world shaped by human activities, this is becoming ever more important.

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REFERENCES

Albert, C., Luque, G. M., & Courchamp, F. (2018). The twenty most charismatic species. *PloS One*, *13*(7), e0199149. doi: 10.1371/journal.pone.0199149

Alexander, L., & Moore, M. (2016). Deontological Ethics. In *The Stanford Encyclopedia of Philosophy*. Retrieved from https://plato.stanford.edu/archives/win2016/entries/ethics-deontological/

Bacher, S., Blackburn, T. M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J. M., … Kumschick, S. (2018). Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution*, *9*(1), 159–168. doi: 10.1111/2041-210X.12844

Batavia, C., Nelson, M. P., & Wallach, A. D. (2020). The moral residue of conservation. *Conservation Biology*. doi: 10.1111/cobi.13463

Bellard, C., Cassey, P., & Blackburn, T. M. (2016). Alien species as a driver of recent extinctions. *Biology Letters*, *12*(2), 20150623. doi: 10.1098/rsbl.2015.0623

Bermond, B., Armstrong, S. J., & Botzler, R. G. (2008). A neuropsychological and evolutionary approach to animal consciousness and animal suffering. *The Animal Ethics Reader*, *2*, 99–112.

Bertolino, S. (2008). Introduction of the American grey squirrel (*Sciurus carolinensis*) in Europe: a case study in biological invasion. *Current Science*, *95*(7), 903–906.

Bhagwat, S. A., Dudley, N., & Harrop, S. R. (2011). Religious following in biodiversity hotspots: challenges and opportunities for conservation and development. *Conservation Letters*, *4*(3), 234–240. doi: 10.1111/j.1755-263X.2011.00169.x

Blackburn, T. M., Essl, F., Evans, T., Hulme, P. E., Jeschke, J. M., Kühn, I., … Bacher, S. (2014). A unified classification of alien species based on the magnitude of their environmental impacts. *PLoS Biology*, *12*(5), e1001850. doi: 10.1371/journal.pbio.1001850

Boissy, A., & Lee, C. (2014). How assessing relationships between emotions and cognition can improve farm animal welfare. *Revue Scientifique et Technique (International Office of Epizootics)*, *33*, 103–110.

Bolund, P., & Hunhammar, S. (1999). Ecosystem services in urban areas. *Ecological Economics*, *29*(2), 293–301. doi: 10.1016/S0921-8009(99)00013-0

Bottrill, M. C., Joseph, L. N., Carwardine, J., Bode, M., Cook, C., Game, E. T., … McDonald-Madden, E. (2008). Is conservation triage just smart decision making? *Trends in Ecology &*

Evolution, *23*(12), 649–654. doi: 10.1016/j.tree.2008.07.007

Brennan, A., & Lo, Y.-S. (2016). Environmental Ethics. In *The Stanford Encyclopedia of Philosophy* (Winter 201). Retrieved from https://plato.stanford.edu/archives/win2016/entries/ethics-environmental/

Brim-Box, J., Guest, T., Barker, P., Jambrecina, M., Moran, S., & Kulitja, R. (2010). Camel usage and impacts at a permanent spring in central Australia: a case study. *The Rangeland Journal*, *32*(1), 55–62. doi: 10.1071/RJ09042

Broom, D. M. (1988). The scientific assessment of animal welfare. *Applied Animal Behaviour Science*, *20*(1–2), 5–19. doi: 10.1016/0168-1591(88)90122-0

Büscher, B. (2016). Reassessing fortress conservation? New media and the politics of distinction in Kruger National Park. *Annals of the American Association of Geographers*, *106*(1), 114–129. doi: 10.1080/00045608.2015.1095061

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., … Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature*, *486*(7401), 59. doi: 10.1038/nature11148

Carey, M. P., Sanderson, B. L., Barnas, K. A., & Olden, J. D. (2012). Native invaders–challenges for science, management, policy, and society. *Frontiers in Ecology and the Environment*, *10*(7), 373–381. doi: 10.1890/110060

Chivian, E., & Bernstein, A. (2008). *Sustaining life: how human health depends on biodiversity*. Oxford University Press.

Cohen-Shacham, E., Walters, G., Janzen, C., & Maginnis, S. (2016). Nature-based solutions to address global societal challenges. *IUCN, Gland, Switzerland*, *97*.

Colléony, A., Clayton, S., Couvet, D., Saint Jalme, M., & Prévot, A.-C. (2017). Human preferences for species conservation: Animal charisma trumps endangered status. *Biological Conservation*, *206*, 263–269. doi: 10.1016/j.biocon.2016.11.035

Coltman, D. W., O'Donoghue, P., Jorgenson, J. T., Hogg, J. T., Strobeck, C., & Festa-Bianchet, M. (2003). Undesirable evolutionary consequences of trophy hunting. *Nature*, *426*(6967), 655. doi: 10.1038/nature02177

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., … van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, *387*(6630), 253–260. doi: 10.1016/S0921-8009(98)00020-2

Courchamp, F., Fournier, A., Bellard, C., Bertelsmeier, C., Bonnaud, E., Jeschke, J. M., & Russell, J. C. (2017). Invasion biology: specific problems and possible solutions. *Trends in Ecology & Evolution*, *32*(1), 13–22. doi: 10.1016/j.tree.2016.11.001

Courchamp, F., Jaric, I., Albert, C., Meinard, Y., Ripple, W. J., & Chapron, G. (2018). The paradoxical extinction of the most charismatic animals. *PLoS Biology*, *16*(4), e2003997. doi: 10.1371/journal.pbio.2003997

Crowley, S. L., Hinchliffe, S., & McDonald, R. A. (2017). Conflict in invasive species management. *Frontiers in Ecology and the Environment*, *15*(3), 133–141. doi: 10.1002/fee.1471

de Waal, F. B. M. (2011). What is an animal emotion? *Annals of the New York Academy of Sciences*, *1224*(1), 191–206. doi: 10.1111/j.1749-6632.2010.05912.x

Désiré, L., Boissy, A., & Veissier, I. (2002). Emotions in farm animals:: a new approach to animal welfare in applied ethology. *Behavioural Processes*, *60*(2), 165–180. doi: 10.1016/S0376-6357(02)00081-5

Di Minin, E., Leader-Williams, N., & Bradshaw, C. J. A. (2016). Banning trophy hunting will exacerbate biodiversity loss. *Trends in Ecology & Evolution*, *31*(2), 99–102. doi: 10.1016/j.tree.2015.12.006

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., … Brauman, K. A. (2018). Assessing nature's contributions to people. *Science*, *359*(6373), 270–272. doi: 10.1126/science.aap8826

Doak, D. F., Bakker, V. J., Goldstein, B. E., & Hale, B. (2015). What is the future of conservation? In *Protecting the wild* (pp. 27–35). doi: 10.5822/978-1-61091-551-9_4

Driscoll, D. A., & Watson, M. J. (2019). Science denialism and compassionate conservation: response to Wallach et al. 2018. *Conservation Biology*, *33*(4), 777–780. doi: 10.1111/cobi.13273

Dubois, S., Fenwick, N., Ryan, E. A., Baker, L., Baker, S. E., Beausoleil, N. J., … Draper, C. (2017). International consensus principles for ethical wildlife control. *Conservation Biology*, *31*(4), 753–760. doi: 10.1111/cobi.12896

Edwards, G. P., Zeng, B., Saalfeld, W. K., & Vaarzon-Morel, P. (2010). Evaluation of the impacts of feral camels. *The Rangeland Journal*, *32*(1), 43–54. doi: 10.1071/RJ09037

Essl, F., Erb, K., Glatzel, S., & Pauchard, A. (2018). Climate change, carbon market instruments, and biodiversity: focusing on synergies and avoiding pitfalls. *Wiley Interdisciplinary Reviews: Climate Change*, *9*(1), e486. doi: 10.1002/wcc.486

Essl, F., Hulme, P. E., Jeschke, J. M., Keller, R., Pyšek, P., Richardson, D. M., … Rabitsch, W. (2017). Scientific and normative foundations for the valuation of alien-species impacts: thirteen core principles. *BioScience*, *67*(2), 166–178. doi: 10.1093/biosci/biw160

Fisher, B., Turner, K., Zylstra, M., Brouwer, R., De Groot, R., Farber, S., … Harlow, J. (2008). Ecosystem services and economic theory: integration for policy-relevant research. *Ecological Applications*, *18*(8), 2050–2067. doi: 10.1890/07-1537.1

Gamborg, C., Gremmen, B., Christiansen, S. B., & Sandoe, P. (2010). De-domestication: ethics at the intersection of landscape restoration and animal welfare. *Environmental Values*, *19*(1), 57–78.

Garibaldi, A., & Turner, N. (2004). Cultural keystone species: implications for ecological conservation and restoration. *Ecology and Society*, *9*(3).

Gaston, K. J. (2010). Valuing common species. *Science*, *327*(5962), 154–155. doi: 10.1126/science.1182818

Genovesi, P., & Bertolino, S. (2001). Human dimension aspects in invasive alien species issues: the case of the failure of the grey squirrel eradication project in Italy. *The Great Reshuffling: Human Dimensions of Invasive Alien Species. IUCN, Gland*, 113–119.

Gibbs, L., & Warren, A. (2015). Transforming shark hazard policy: Learning from ocean-users and shark encounter in Western Australia. *Marine Policy*, *58*, 116–124. doi: 10.1016/j.marpol.2015.04.014

Hayward, M. W., Callen, A., Allen, B. L., Ballard, G., Broekhuis, F., Bugir, C., … Wüster, W. (2019). Deconstructing compassionate conservation. *Conservation Biology*, *33*(4), 760–768. doi: 10.1111/cobi.13366

Huang, I. B., Keisler, J., & Linkov, I. (2011). Multi-criteria decision analysis in environmental sciences: ten years of applications and trends. *Science of the Total Environment*, *409*(19), 3578–3594. doi: 10.1016/j.scitotenv.2011.06.022

ICMO2. (2010). Natural processes, animal welfare, moral aspects and management of the Oostvaardersplassen. In *Report of the second International Commission on Management of the Oostvaardersplassen (ICMO2).* The Hague/Wageningen, Netherlands. Wing rapport 039. November 2010.

IUCN-CEM. (2016). *The IUCN Red List of Ecosystems. Version 2016-1. <http://iucnrle.org>.*

IUCN. (2019). *The IUCN Red List of Threatened Species. Version 2019-1. <http://www.iucnredlist.org>.*

Jarić, I., Courchamp, F., Correia, R. A., Crowley, S. L., Essl, F., Fischer, A., … Lenzner, B. (2020). The role of species charisma in biological invasions. *Frontiers in Ecology and the Environment*. doi: 10.1002/fee.2195

Kareiva, P. (2014). New conservation: setting the record straight and finding common ground. *Conservation Biology*, *28*(3), 634–636. doi: 10.1111/cobi.12295

Kareiva, P., & Marvier, M. (2012). What is conservation science? *BioScience*, *62*(11), 962–969. doi: 10.1525/bio.2012.62.11.5

Kuussaari, M., Bommarco, R., Heikkinen, R. K., Helm, A., Krauss, J., Lindborg, R., … Rodà, F. (2009). Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology & Evolution*, *24*(10), 564–571. doi: 10.1016/j.tree.2009.04.011

Lindsey, P. A., Frank, L. G., Alexander, R., Mathieson, A., & Romanach, S. S. (2007). Trophy hunting and conservation in Africa: problems and one potential solution. *Conservation Biology*, *21*(3), 880–883. doi: 10.1111/j.1523-1739.2006.00594.x

Lindsey, P. A., Roulet, P. A., & Romanach, S. S. (2007). Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. *Biological Conservation*, *134*(4), 455–469. doi: 10.1016/j.biocon.2006.09.005

Loreau, M. (2010). Linking biodiversity and ecosystems: towards a unifying ecological theory. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1537), 49–60. doi: 10.1098/rstb.2009.0155

Mace, G. M. (2014). Whose conservation? *Science*, *345*(6204), 1558–1560. doi: 10.1126/science.1254704

MacLean, E. L., Matthews, L. J., Hare, B. A., Nunn, C. L., Anderson, R. C., Aureli, F., … Emery, N. J. (2012). How does cognition evolve? Phylogenetic comparative psychology. *Animal Cognition*, *15*(2), 223–238. doi: 10.1007/s10071-011-0448-8

McIlroy, J. C. (1981). The sensitivity of Australian animals to 1080 poison. II. Marsupial and eutherian carnivores. *Wildlife Research*, *8*(2), 385–399. doi: 10.1071/WR9810385

Miller, T. R., Minteer, B. A., & Malan, L.-C. (2011). The new conservation debate: the view from practical ethics. *Biological Conservation*, *144*(3), 948–957. doi:

10.1016/j.biocon.2010.04.001

Mills, L. S., Soulé, M. E., & Doak, D. F. (1993). The keystone-species concept in ecology and conservation. *BioScience*, *43*(4), 219–224. doi: 10.2307/1312122

Nussbaum, M. C. (1999). Virtue ethics: A misleading category? *The Journal of Ethics*, *3*(3), 163–201. doi: 10.1023/A:1009877217694

Ohl, F., & Van der Staay, F. J. (2012). Animal welfare: At the interface between science and society. *The Veterinary Journal*, *192*(1), 13–19. doi: 10.1016/j.tvjl.2011.05.019

Oommen, M. A., Cooney, R., Ramesh, M., Archer, M., Brockington, D., Buscher, B., … Webb, G. (2019). The fatal flaws of compassionate conservation. *Conservation Biology*, *33*(4), 784–787. doi: 10.1111/cobi.13329

Palazy, L., Bonenfant, C., Gaillard, J.-M., & Courchamp, F. (2011). Cat dilemma: too protected to escape trophy hunting? *PloS One*, *6*(7), e22424. doi: 10.1371/journal.pone.0022424

Palazy, L., Bonenfant, C., Gaillard, J.-M., & Courchamp, F. (2013). On the use of the IUCN status for the management of trophy hunting. *Wildlife Research*, *39*(8), 711–720. doi: 10.1071/WR12121

Palazy, L., Bonenfant, C., Gaillard, J. M., & Courchamp, F. (2012). Rarity, trophy hunting and ungulates. *Animal Conservation*, *15*(1), 4–11. doi: 10.1111/j.1469-1795.2011.00476.x

Palmer, C., McShane, K., & Sandler, R. (2014). Environmental ethics. *Annual Review of Environment and Resources*, *39*, 419–442. doi: 10.1146/annurev-environ-121112-094434

Parker, I. M., Simberloff, D., Lonsdale, W. M., Goodell, K., Wonham, M., Kareiva, P. M., … Goldwasser, L. (1999). Impact: toward a framework for understanding the ecological effects of invaders. *Biological Invasions*, *1*(1), 3–19. doi: 10.1023/A:1010034312781

Perry, D., & Perry, G. (2008). Improving interactions between animal rights groups and conservation biologists. *Conservation Biology*, *22*(1), 27–35. doi: 10.1111/j.1523-1739.2007.00845.x

Peterson, M. N., Peterson, M. J., & Peterson, T. R. (2005). Conservation and the myth of consensus. *Conservation Biology*, *19*(3), 762–767. doi: 10.1111/j.1523-1739.2005.00518.x

Proença, V. M., Pereira, H. M., & Vicente, L. (2008). Organismal complexity is an indicator of species existence value. *Frontiers in Ecology and the Environment*, *6*(6), 298–299. doi: 10.1890/1540-9295(2008)6[298:OCIAIO]2.0.CO;2

Proulx, G., Brook, R. K., Cattet, M., Darimont, C., & Paquet, P. C. (2016). Poisoning wolves with strychnine is unacceptable in experimental studies and conservation programmes. *Environmental Conservation*, *43*(1), 1–2. doi: 10.1017/S0376892915000211

Ramp, D., & Bekoff, M. (2015). Compassion as a practical and evolved ethic for conservation. *BioScience*, *65*(3), 323–327. doi: 10.1093/biosci/biu223

Redpath, S. M., Young, J., Evely, A., Adams, W. M., Sutherland, W. J., Whitehouse, A., … Watt, A. (2013). Understanding and managing conservation conflicts. *Trends in Ecology & Evolution*, *28*(2), 100–109. doi: 10.1016/j.tree.2012.08.021

Rikoon, J. S. (2006). Wild horses and the political ecology of nature restoration in the Missouri Ozarks. *Geoforum*, *37*(2), 200–211. doi: 10.1016/j.geoforum.2005.01.010

Rohwer, Y., & Marris, E. (2019). Clarifying compassionate conservation with hypotheticals: response to Wallach et al. 2018. *Conservation Biology: The Journal of the Society for Conservation Biology*, *33*(4), 781–783. doi: 10.1111/cobi.13274

Rolston III, H. (2003). Environmental ethics. In N. Bunnin & E. P. Tsui-James (Eds.), *The Blackwell Companion to Philosophy, 2nd Edition*. Oxford: Blackwell Publishing.

Russell, J. C., Jones, H. P., Armstrong, D. P., Courchamp, F., Kappes, P. J., Seddon, P. J., … Rocamora, G. (2016). Importance of lethal control of invasive predators for island conservation. *Conservation Biology*, *30*(3), 670–672. doi: 10.1111/cobi.12666

Schuchert, P., Shuttleworth, C. M., McInnes, C. J., Everest, D. J., & Rushton, S. P. (2014). Landscape scale impacts of culling upon a European grey squirrel population: can trapping reduce population size and decrease the threat of squirrelpox virus infection for the native red squirrel? *Biological Invasions*, *16*(11), 2381–2391. doi: 10.1007/s10530-014-0671-8

Shao, Y., Li, M., Zhang, W., Ji, Y., & Hayes, D. J. (2018). World's Largest Pork Producer in Crisis: China's African Swine Fever Outbreak. *Agricultural Policy Review*, *2018*(3), 1.

Shriver, A. (2006). Minding mammals. *Philosophical Psychology*, *19*(4), 433–442. doi: 10.1080/09515080600726385

Singer, P. (1980). Utilitarianism and vegetarianism. *Philosophy & Public Affairs*, 325–337.

Singer, P. (2009). Speciesism and moral status. *Metaphilosophy*, *40*(3-4), 567–581. doi: 10.1111/j.1467-9973.2009.01608.x

Siurua, H. (2006). Nature above people: Rolston and " fortress" conservation in the south. *Ethics and the Environment*, 71–96.

Soulé, M. (2014). The "New Conservation." *Conservation Biology*, *27*, 895–897. doi:

10.1111/cobi.12147

Soulé, M. E. (1985). What is conservation biology? *BioScience*, *35*(11), 727–734. doi: 10.2307/1310054

Stokes, D. L. (2007). Things we like: human preferences among similar organisms and implications for conservation. *Human Ecology*, *35*(3), 361–369. doi: 10.1007/s10745-006-9056-7

Tam, K.-P., Lee, S.-L., & Chao, M. M. (2013). Saving Mr. Nature: Anthropomorphism enhances connectedness to and protectiveness toward nature. *Journal of Experimental Social Psychology*, *49*(3), 514–521. doi: 10.1016/j.jesp.2013.02.001

Taylor, P. W. (2011). *Respect for nature: A theory of environmental ethics*. Princeton University Press.

UNEP CBD. (2010). Strategic plan for biodiversity 2011–2020 and the Aichi targets. *Report of the Tenth Meeting of the Conference of the Parties to the Convention on Biological Diversity*.

Varner, G. (2008). Utilitarianism and the evolution of ecological ethics. *Science and Engineering Ethics*, *14*(4), 551–573. doi: 10.1007/s11948-008-9102-5

Varner, G. (2011). Environmental ethics, hunting, and the place of animals. In *The Oxford handbook of animal ethics*. doi: 10.1093/oxfordhb/9780195371963.013.0032

Wallach, A. D., Batavia, C., Bekoff, M., Alexander, S., Baker, L., Ben-Ami, D., … Ramp, D. (2020). Recognizing animal personhood in compassionate conservation. *Conservation Biology*. doi: 10.1111/cobi.13494

Wallach, A. D., Bekoff, M., Batavia, C., Nelson, M. P., & Ramp, D. (2018). Summoning compassion to address the challenges of conservation. *Conservation Biology*, *32*(6), 1255–1265. doi: 10.1111/cobi.13126

Waytz, A., Iyer, R., Young, L., Haidt, J., & Graham, J. (2019). Ideological differences in the expanse of the moral circle. *Nature Communications*, *10*(1), 1–12. doi: 10.1038/s41467-019-12227-0

Weitz, N., Carlsen, H., Nilsson, M., & Skånberg, K. (2018). Towards systemic and contextual priority setting for implementing the 2030 Agenda. *Sustainability Science*, *13*(2), 531–548. doi: 10.1007/s11625-017-0470-0

Tables

Table1. Glossary of terms as they are used for the purposes of this paper.

Term	Definition
Anthropocentrism	Value system that considers humans to be the sole, or primary, holder of moral standing, and therefore the concern of direct moral obligations. Non-human species are considered only to the extent that they affect humans (Rolston III 2003; Palmer *et al.* 2014).
Anthropomorphism	"The attribution of human personality or characteristics to something non-human, like an animal, object, etc." (Oxford English Dictionary).
Biocentrism	Value system considering all living beings as the concern of direct moral obligations (Rolston III 2003; Palmer *et al.* 2014).
Collectivism	Value system in which a group or collective has a higher value than the individuals that compose it (Wallach *et al.* 2018).
Compassionate conservation	Conservation approach inspired by virtue ethics based on four tenets: i) do no harm; ii) individuals matter; iii) inclusivity (the value of an individual is independent from the context of the population, e.g. nativity, rarity, etc.); and iv) peaceful coexistence (Ramp & Bekoff 2015; Wallach *et al.* 2018).
Consequentialism	"An ethical doctrine which holds that the morality of an action is to be judged solely by its consequences" (Oxford English Dictionary).
Deontology	A normative ethical theory considering that "choices are morally required, forbidden, or permitted" (Alexander & Moore 2016).
Ecocentrism	Value system considering that species, their assemblages and their functions, as well as more broadly ecosystems, rather than individuals, are the concern of direct moral obligations (Rolston III 2003; Palmer *et al.* 2014).
Empathy	"The quality or power of projecting one's personality into or mentally identifying oneself with an object of contemplation, and so fully

	understanding or appreciating it." (Oxford English Dictionary). Empathy will influence the inherent value given to individuals from other species.
Impact (for the purposes of the framework, Eq.1)	Impact refers to any effect that modifies the well-being, health or resilience (for non-sentient beings) of an individual, from physical pain to emotional suffering and death (these notions being interrelated, but not equivalent).
Invasive alien species	"Plants, animals, pathogens and other organisms that are non-native to an ecosystem, and which may cause economic or environmental harm or adversely affect human health" (*Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species*).
Moral community	The group of beings considered to have intrinsic moral value (Shoemaker 2010). The size of the group depends on the value system. For example, the moral community is restricted to humans in case of Anthropocentrism.
Nativism	Value system considering that species that have evolved in a given location have a higher value in this location than species that have evolved somewhere else. In nativism, value varies spatially (Wallach *et al.* 2018).
Nature despite people	Management conceptual approach aiming at conserving biological diversity (focusing on species and habitats) specifically in response to human impacts on the environment, e.g. sustainable use (Mace 2014).
Nature for itself	Management conceptual approach aiming at conserving biological diversity (focusing on wilderness and natural habitats) through human exclusion, for example through the creation of parks and protected areas (Mace 2014).
Nature for people	Management conceptual approach aiming at conserving the components of nature beneficial to humans (focusing on ecosystems and their services) (Mace 2014).

Neoteny	"The retention of juvenile characteristics in a (sexually) mature organism" (Oxford English Dictionary).
New conservation	Discipline aiming at preserving biological diversity through the conservation of natural elements providing services and contribution to human wellbeing (Kareiva & Marvier 2012; Kareiva 2014).
Normative postulate	Value statements that make up the basis of an ethic of appropriate attitudes toward other forms of life (Soulé 1985).
Parochialism	Ideology in which moral regard is directed "towards socially closer and structurally tighter targets, relative to socially more distant and structurally looser targets", and, by extension, to species phylogenetically, cognitively, or in appearance closer to humans (Waytz *et al.* 2019).
People and nature	Management conceptual approach considering that humans and nature are interdependent and therefore aiming at achieving compromises in the conservation of nature and human wellbeing. (Mace 2014).
Sentience	The ability to experience phenomenal consciousness, i.e. the qualitative, subjective, experiential, or phenomenological aspects of conscious experience, rather than just the experience of pain and pleasure (Allen & Trestman 2017).
Sentientism	Value system considering sentient beings as the concern of direct moral obligations (Rolston III 2003; Palmer *et al.* 2014).
Speciesism	Value system in which some species are considered to have a higher value than others, for various possible reasons (Singer 2009). Speciesism is often used to refer to the superiority of humans, which is a specific expression of speciesism as considered in this paper.
Suffering	Negative emotion, sometimes called emotional distress, experienced by sentient beings, and which can result from different causes, including but not limited to physical pain (Dawkins 2008; Farah 2008).
Traditional conservation	Discipline aiming at preserving biological diversity through the management of nature, and based on four value-driven normative postulates: "diversity of organisms is good," "ecological complexity is

	good," "evolution is good," and "biotic diversity has intrinsic value" (Soulé 1985). Traditional conservation is rooted in ecocentrism.
Virtue ethics	Ethical doctrine that emphasizes the virtues, or moral character as the reason for action (Hursthouse & Pettigrove 2018).

1164 **Table 2.** List of factors to consider regarding the effects of environmental management actions
1165 from an environmental ethics perspective.

Factor	**Influence on variables and outputs in Equations 1 to 5**
Biotic interactions	The impact or suffering of individuals from one species can be caused by individuals from another species, either through direct or indirect interactions. Management actions can therefore also have non-trivial indirect impacts on some species.
Capacity to provide ecosystem services	The presence of species may increase the welfare of other animal species through the ecosystem services they provide. Since these effects can be difficult to quantify explicitly, the value of such species may be increased in Equations 1 to 4 to account for them, or an additional term can be included, as in Equation 5.
Discounting rate	Rate at which impacts that occur in the future lose importance.
Impact quantification and commensurability	How the impacts of management actions are quantified is also dependent on value systems, as some impacts (such as death) may be considered incommensurable to others (such as suffering).
Responsibility from previous actions	Previous human actions on certain species, such as reintroduction of domesticated species or the introduction of alien species can change the perception of the public and therefore increase the value attributed to these species, or decrease the morality of an action, in addition to obviously having an impact on these species.
Spatial scale	The spatial scale will change the abundance N and the number of species considered. As a result, a management action that is more beneficial than another at small scale may not be such at a larger scale, and reciprocally.
Temporal scale	The time frame over which the impact or the suffering of individuals is computed can change their values. Management actions may decrease welfare of individuals on the short term, but be beneficial on the long term once the ecosystem has stabilised. Similarly, not culling some population may cause less suffering on the short term, but increase it in the future by disrupting ecosystem

	services, leading to population collapse due to lack of resources, etc.
Uncertainty of impact	The complexity of an ecological system can make the assessment of the impact of management actions on different species difficult to assess precisely, therefore creating potential errors, especially in the presence of multiple biotic interactions. This may lead to an incorrect estimation of the outcome *O*.
Uncertainty of value expressions and preferences	Quantifying the value given by a person or a group of people to an individual is difficult, context-dependent, and highly subjective. Sensitivity analyses on the distribution of values can be used to account for such uncertainty.

Figure

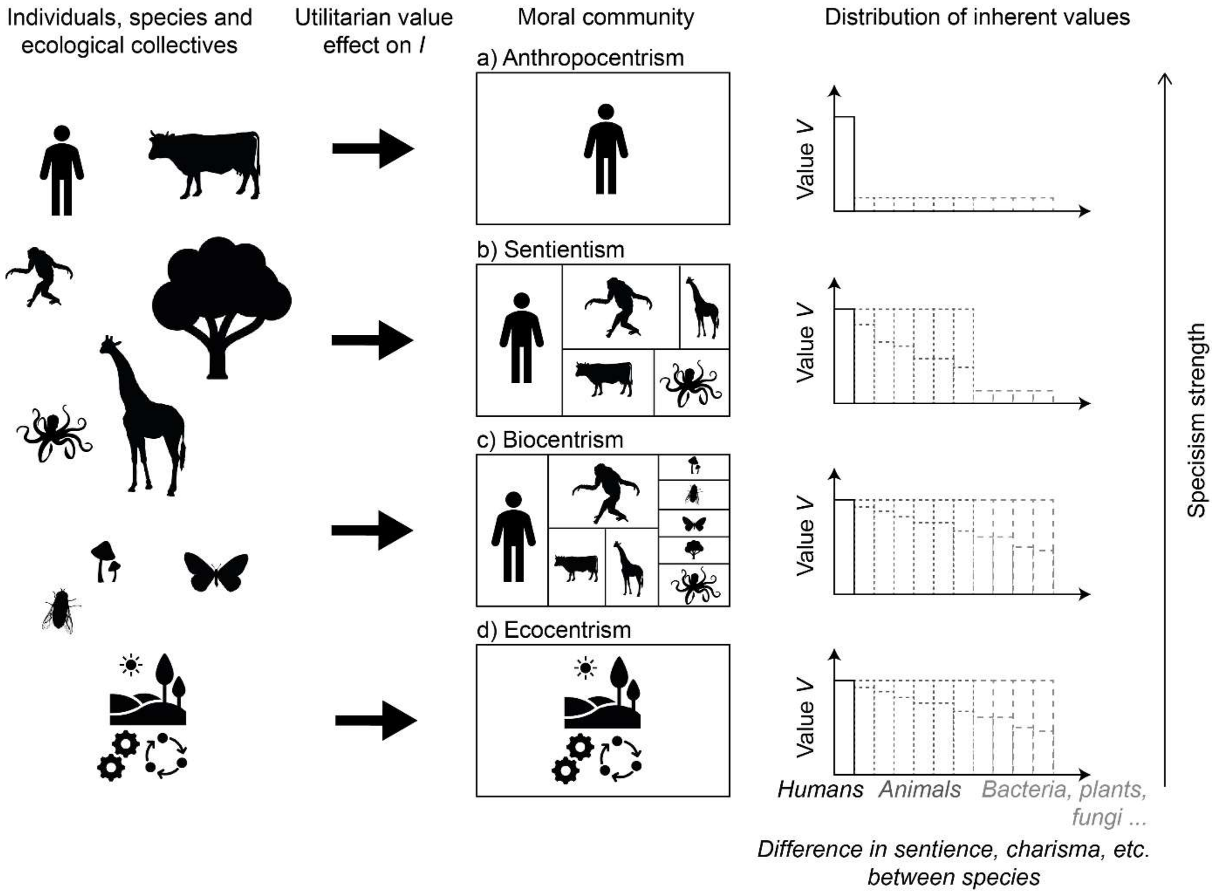

Figure 1. Difference between value systems influenced by a) anthropocentrism, b) sentientism, c) biocentrism and d) ecocentrism. Anthropocentrism, sentientism and biocentrism all value individuals inherently, but consider different moral communities, i.e. their values depend on the category of species they belong to, with {humans} ∈ {sentient beings} ∈ {all living organisms}. Ecocentrism, in contrast, is not based on individuals, but on ecological collectives, i.e. on species or on species assemblages and ecosystems. Species outside of the moral community may have a utilitarian value for species in the moral community (represented by the arrows), which will be reflected by changes in the impact variable. Note that species can have both an inherent and a utilitarian value. Within the moral community, species may have equal inherent values, but subjective perceptions and different value systems can assign different values to different species. The skewness of the value distribution then indicates the degree or strength of

speciesism with respect to the species of references, assumed here to be the human species, and is influenced by many factors, including charisma, cultural context, etc.