Ecosystem Services 67 (2024) 101621

Contents lists available at ScienceDirect

Ecosystem Services

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

Full Length Article

Measuring non-use values to proxy conservation preferences and policy impacts

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A R T I C L E I N F O

Keywords:
Conservation preferences
Conservation policy
Donations
Contingent valuation
Deliberative monetary valuation
Hypothetical bias

A B S T R A C T

The behavior of local natural resource users is not only affected by economic incentives but also by a diverse set of motivations and underlying values. These non-monetary drivers are crucial in safeguarding long-term positive conservation outcomes. However, measuring these factors still constitutes a significant challenge. Building on lessons learned from established methodology such as attitudinal or behavioral measures, we showcase how a contingent valuation method and experimental donation tasks can be used to measure relative changes in non-use values and are a good proxy for conservation preferences. We exemplify this approach within the context of a case study in northern Namibia, where it was employed to investigate whether exposure to a community-based conservation program affects individual conservation preferences. Our findings show that our approach can serve as a complement to established measures for conservation preferences while avoiding some of the existing pitfalls such as demand effects or costly data collection associated with behavioral and attitudinal measures.

1. Introduction

Environmental awareness or inclination toward conservation has a history of being measured via agreement to attitudinal or value-based statements (Nilsson et al., 2020). These approaches have faced criticism, primarily because of growing concerns surrounding phenomena like the value-action gap (Kollmuss & Agyeman, 2002) as well as the reliability and validity of self-reported measures, especially in contexts where respondents face complex trade-offs between environmental protection and economic goals (Kenny, 2021). Researchers have also raised doubts about the accuracy of responses provided by survey participants, as they may align their answers with the perceived social desirability (Milfont, 2009; Vesely & Klöckner, 2020). Likewise, measures of actual conservation behavior are often confounded with incentives, rules, and regulations that are already in place. Thus, people might carry out a desired behavior, but not due to their 'conservation preferences'. In this article, we build on terminology and empirical traditions from economics to showcase an application of preference measurement that allows for circumventing some of the challenges associated with attitudinal or behavioral measures of conservation preferences. We highlight how the contingent valuation method, when focusing on non-use values, can add insights into whether and how a certain policy has influenced the way people relate to conserving ecosystems. Our case study from community-based natural resource management (CBNRM) communities in northern Namibia and neighboring villages that are not part of the policy provides evidence that conservation preferences have not increased despite the conservation policy being in place for more than ten years.

What are conservation preferences? Economists conceptualize preferences as the drivers of individual decision-making and behavior (Falk et al., 2018; D. M. Hausman, 2011). If a person has positive conservation preferences, this person can thus be expected to be supportive of conservation interventions and also to individually act in favor of conservation outcomes (Selinske et al., 2020). E.g., in a choice situation characterized by clear individual trade-offs between environmental and economic outcomes, a person with stronger conservation preferences would be willing to accept more economic losses in order to ensure a positive conservation outcome than a person with lower conservation preferences. As we will further clarify, conservation preferences should be clearly distinguished from preferences for personal economic benefits

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https://doi.org/10.1016/j.ecoser.2024.101621
Received 4 July 2023; Received in revised form 29 March 2024; Accepted 8 April 2024

people may derive or expect to receive from conservation activities. Otherwise, they might reflect a wide array of values that people attribute to nature, which have been conceptualized in various ways (e.g., intrinsic, instrumental, and relational values) (Chan et al., 2016; Pascual et al., 2017, 2023). Within the Total Economic Value framework of nature valuation (Anderson et al., 2022, Fig. 2.13) conservation preferences could thus encompass all types of non-use values as well as direct and indirect use values from natural processes that do not lead to a direct personal economic benefit (such as selling fish or fruits on the market or generating income through eco-tourism). The academic discourse on the potential relevance of non-use values in conservation policymaking, as well as how and to which degree they should be incorporated into economic cost-benefit analyses dates back to the 1960s (Krutilla, 1967; Weisbrod, 1964). Following economic theory, an insufficient acknowledgment of non-use values in decision-making governing conservation policies would lead to an under-provision of the natural resources providing them. While discussions about problems inherent to the exact measurement of non-use values remained a point of contention over the years (see e.g., Carson, 2012; Hausman, 2012), they continue to be an important concept in climate policy on a global scale and their full incorporation into related cost-benefit calculations would have significant policy impacts (Bastien-Olvera & Moore, 2021).

Are conservation preferences important to consider when implementing, managing, and evaluating conservation policies? Conservation practitioners would likely agree that the level of support for conservation actions and the likelihood to act in line with conservation goals are desirable outcomes of many policy interventions. Especially in cases where economic benefits to local communities are only temporary (e.g., via donor support in implementation stages), or fail to materialize at all, an increase of conservation preferences induced by these policies is thus crucial for achieving longer-term positive conservation outcomes (Hayes et al., 2022).[1] Particularly interesting applications of conservation preferences as policy impacts can be found within the context of community-based conservation programs such as the above-mentioned CBNRM or Payments for Ecosystem Services (PES). Policy approaches like this are frequently presented as win–win solutions, in which both biodiversity conservation and economic well-being or development can be promoted simultaneously. However, scenarios that effectively overcome the complex tradeoffs between those two objectives may be relatively rare and difficult to realize in the first place (Hegwood et al., 2022; McShane et al., 2011; Muradian et al., 2013). Table 1 illustrates and describes four theoretical cases, reflecting outcomes of conservation behavior in response to conservation initiatives or policies, that differ both in the realization of economic benefits for the local population and their influence on conservation preferences. Case C represents the case in which CBNRM programs include a bundle of interventions such as awareness raising and information campaigns that strengthen conservation preferences irrespective of any economic benefits (Green et al., 2019). Cases A and B represent incentive-based policies that provide economic benefits to the community. The effect of such policies on conservation preferences can go both ways depending on their design and the socio-cultural context – incentives have been shown to potentially enhance (crowd-in) or decrease (crowd-out) non-economic motivations for conservation (Blanco et al., 2023; Ezzine-de-Blas et al., 2019; Rode et al., 2015).

Table 1
Dependency of Conservation Behavior on Conservation-related Preferences and Economic Benefits. In the presence of economic net benefits (cases A and B), positive conservation actions do not depend on the presence of conservation preferences. However, one would expect communities with strengthened conservation preferences (case A) to be more receptive toward additional conservation actions that do not generate economic returns per se. A decrease in conservation preferences may even result in a net negative effect on conservation behavior (case B). When conservation initiatives lack economic benefits (cases C and D), then positive conservation actions are conditional on enhanced conservation preferences (case C). Finally, one would not expect positive conservation behavior, if no financial benefits are generated nor conservation preferences are enhanced (case D).

Net effect on conservation behavior		**Increase in conservation preferences of resource users (e.g. internalization of ecosystem benefits)**	
		Yes	**No / Unclear**
Economic net benefits to resource users (e.g. payments, labor)	**Yes**	A (++)	B (+, 0, or −)
	No	C (+)	D (0 or −)

In this paper, we conceptualize and measure conservation preferences within a case study on community conservancies in Namibia, one of the most prominent and long-standing CBNRM programs in Sub-Saharan Africa (Boudreaux & Nelson, 2011). Our study investigates potential changes to conservation preferences in a scenario in which the aforementioned economic incentives have widely not materialized, thereby increasing the importance of considering conservation preferences (case C or D).

In our case study, we employed a contingent valuation scenario for a one-time donation to a Namibian conservation project, concerned with monitoring and saving remaining populations of the lappet-faced vulture (Torgos Tracheliotos), an endangered bird species native to almost all of Namibia. We specifically chose this scenario and designed it in such a way that it mostly captures non-use values or indirect use values at most. Within the Total Economic Value framework of nature valuation, non-use values are distinguished from use-values in that they provide no direct benefits to the individual in question. Non-use values are typically further differentiated between those that origin from the satisfaction of knowing that the good in question will be available to others now (altruistic values) or to future generations (bequest values). A third group of non-use values are so-called existence values, which are solely derived from the satisfaction of knowing that an environmental good or natural resource exists (Manero et al., 2022, Fig. 1.; Otrachshenko et al., 2022). While indirect use values are typically listed as a use-value category, they provide no directly perceivable benefits to the individual (e.g., in the form of tradeable goods or recreational activities) but provide important ecosystem functions (Pascual et al., 2010). However, in our case, the specific vulture species is very rarely found in the study region and therefore the role of indirect use values is largely negligible. We thus argue that the amount donated to the vulture conservation cause can be regarded as a valid measure of conservation preferences. This is because the trade-off inherent to the donation decision (giving up the potential utility derived from spending the money for the benefit of nature) is not confounded with any direct benefit from the donation through use values of vultures. Keeping in mind that conservation preferences for all nature within the community land may involve many more value components associated with species and ecosystems, the absolute value of conservation preferences for the particular vulture species can certainly not be equated with the absolute value

[1] Conservation preferences have, to our knowledge, not yet been treated explicitly as policy impacts in economics. One reason for this could be that economics traditionally conceptualized individual preferences to be stable over time, similarly to how personality traits are viewed in psychology. By definition, this would not make them subject of policy evaluations. Individual attitudes, on the other hand, are considered to be less stable (Achen, 1975) and to be changing with experience or depending on the specific context (Gifford et al., 2011). The assumption of stable preferences has always been contested within the social sciences and preference instability has been captured within concepts such as 'endogenous' (Bowles, 1998) or 'adaptive' preferences (von Weizsäcker, 2005). Once preferences are no longer regarded as a stable character trait, they can indeed be considered as outcome variables of an (environmental) policy process.

of conservation preferences for nature. Yet, we argue that it can serve as a conservative and clean indicator for measuring changes in conservation preferences over time and across study sites.

The case study further enables us to scrutinize two methodological key issues related to economic measures of conservation preferences: First, observing both hypothetical willingness to pay (WTP) statements and real donations on an individual level allows us to evaluate potential differences in the hypothetical bias between study sites. Second, we use our case study to experimentally investigate the effect of group deliberation on the measures of conservation preferences.

The remainder of the paper is structured in the following way: Section 2 discusses the concept of conservation preferences, and also provides a critical evaluation of different types of instruments for their measurement. Section 3 presents an application of our proposed measurement approach involving communal conservancies in the Kavango-East Region, Namibia. The section is further subdivided into subsections describing the study context, the design of the contingent valuation scenario as well as an analysis section describing our hypotheses, econometric approach and the study results. Section 4 discusses the case study findings and Section 5 concludes.

2. Measuring conservation preferences and related concepts

Conservation preferences share conceptual similarities to key concepts from psychology such as 'motivations' (Ryan & Deci, 2000), values (Shiell et al., 1997), and 'attitudes' (Nilsson et al., 2020) for conservation. In this paper, we cannot fully alleviate the complexity of delineating these different concepts. Instead, we intend to clarify in particular the role of economic methods among the plurality of related tools from different disciplines to measure conservation preferences. The following paragraphs explain the different approaches for measuring conservation preferences.

2.1. Attitudinal measures

Psychologists traditionally investigate how values, attitudes, and norms change, which are assumed to influence behavioral intentions and actual behavior (de Groot & Steg, 2008; Fishbein & Ajzen, 1975; Stern & Dietz, 1994). In particular, attitudinal measures are a common tool in psychological research on biology conservation and are used to evaluate the effectiveness of conservation programs (Nilsson et al., 2020). Attitudinal statements rely on Likert-scale ratings, using items such as the New Ecological Paradigm (NEP) scale (Dunlap et al., 2000) or via items directly relevant to the specific conservation case, as e.g., individual attitudes on (un)desired conservation behavior, endangered species or wildlife management practices (Nilsson et al., 2020).

The main virtue of eliciting attitudes to evaluate conservation preferences is that it is quick, cheap, easily implementable, and, depending on the use of standardized scales, also comparable across different study contexts. Widely applied scales, such as e.g., the NEP, have been typically tested for reliability and validity, however often only in specific cultural (i.e., western) contexts. While attitudes have been shown to predict pro-environmental behavior, on average they only do so imperfectly (Armitage & Conner, 2001); the phenomenon commonly referred to as "attitude-behavior-gap" in environmental economics and psychological literature (Farjam et al., 2019). Experimental economists have also pointed out that attitudinal scales are also especially prone to demand effects (Zizzo, 2010). In this case, respondents would consciously or unconsciously provide answers in line with their conceptions about the researcher's expectations. For example, respondents may provide systemically biased answers if they expect that these answers increase the chances of attracting or retaining conservation projects and associated funding.

2.2. Behavioral intentions

As a way to get closer to conservation behavior, many studies have used self-reported behavior (e.g., "Do you typically engage in wildfire control measures?") or behavioral intentions (e.g., "In the coming month, do you plan to engage more regularly in wildfire control measures?") to approximate people's conservation preferences. Survey items of this type can be regarded as advantageous over scale-based attitudinal statements as they can be adjusted to the specific context of the conservation intervention in question. However, as purely hypothetical statements, they suffer from similar drawbacks as the attitudinal measures, namely being prone to demand effects and an unclear correspondence to real behavior (Webb & Sheeran, 2006). These issues may be particularly pronounced when the behavioral intentions relate to potentially incriminating practices such as poaching or illegal logging. Additionally, in contrast to purely attitudinal statements, behavioral intentions can be already influenced by economic constraints, i.e., material pay-offs associated with specific behavior. Conservation programs may alter these parameters and thus affect behavioral intentions without changing underlying conservation preferences.

2.3. Stated preferences: Willingness-to-pay (WTP)

Economists have a long tradition of measuring preferences based on monetary values, either by observing actual behavior (e.g., purchases on markets) or by eliciting stated preferences, e.g. in the form of willingness-to-pay (WTP). WTP is commonly understood as the maximum monetary amount an individual is willing to forego in order to e.g., buy a product or service. These approaches are thought to be superior as respondents face a trade-off between conservation and something else. The measurement of conservation preferences should not automatically assume win–win constellations between environmental and economic outcomes. Instead, it should acknowledge that the conservation of natural resources, at least initially, can in fact be detrimental to economic goals. Attitudinal measures or behavioral intentions neglect such trade-offs or at least have difficulties in disentangling individual motives (Kenny, 2021), while economic approaches enable researchers to take them into account.

While the discrete choice experiment is nowadays the more popular method to elicit WTP estimates for natural resources (Chaikaew et al., 2017; Dias & Belcher, 2015), the 'contingent valuation' approach (Carson, 2012b; J. Hausman, 2012b; Kling et al., 2012), where study participants are directly asked to state their hypothetical WTP for an environmental good, remains in use as well (e.g. Otrachshenko et al., 2022). While in the most typical case, this approach is used to derive an economic value of environmental goods (Carson et al., 2003; Otrachshenko et al., 2022) to inform environmental policy, individual WTP can also be regarded as a proxy for individuals' conservation preferences. In the latter case, the valuation scenario should be carefully designed. For example, the WTP question could be framed as a one-off payment to promote the conservation of a nature reserve or an endangered species. Importantly, all respondents (both targeted and not targeted by the policy or intervention at hand) should face similar economic (dis)incentives towards the respective conservation good. Rather than focusing on the absolute monetary levels WTP - in this application - is a relative, comparative measure for conservation preferences between populations (e.g., targeted and not targeted by a conservation project or policy) or over time.

Many economists have been wary of mere statements of preferences, which they consider "cheap talk" (Farrell and Rabin, 1996), and have favored 'revealed preferences' measures based on real-world observations of actual behavior (Frey and Stutzer, 1999). Nevertheless, another line of economics research, especially related to the valuation of environmental goods, relies on contingent valuation methods, where individuals' hypothetical "willingness-to-pay" statements indicate their 'stated preferences' (Carson, 2012; Hausman, 2012). The fact that such

WTP statements are purely hypothetical renders them conceptually similar to the previously discussed behavioral intentions – a person states the intention to act in a certain way (making a payment in this case) – and they, therefore, are prone to similar problems. The contingent valuation method, as well as the closely related discrete choice experiment methodology are under constant criticism for its susceptibility to a phenomenon frequently referred to as 'hypothetical bias', i.e., that studies applying such methods would systematically lead to the overestimation of environmental values (J. Hausman, 2012b; Hensher, 2010). A number of *meta*-analyses from recent years investigate this phenomenon and provide evidence that the magnitude of hypothetical bias can at least be mitigated to some degree by the use of so-called cheap talk scripts or consequentiality scripts, where survey respondents are informed about the hypothetical bias or consequentiality of WTP bids before the value elicitation (Andor et al., 2017; Foster & Burrows, 2017; J. Penn & Hu, 2019; J. M. Penn & Hu, 2018). Other means proposed in the literature to potentially tackle such biases are to ensure task familiarity among respondents applied (Schläpfer and Fischhoff, 2012), using oath scripts prior to the value elicitation (Stevens et al., 2013), or controlling for certain respondent characteristics in the analysis of hypothetical WTP studies (Wuepper et al., 2019).

Other criticisms of the contingent valuation method revolve around the insensitivity to the scope of WTP (Lopes & Kipperberg, 2020), that respondents oftentimes do not have well-defined preferences for goods that are not commonly exchanged in market settings (Hsee & Rottenstreich, 2004; Spash, 2007), or that they might even protest against monetary attribution to nature goods altogether (Spash et al., 2009). Financial constraints may also render money a questionable unit for conservation preferences, e.g., when poor communities do not have the ability to support conservation activities financially (Martínez-Alier, 2002). Thus, absolute levels may be understated and income differences between study sites may render comparisons of hypothetical WTP measures in absolute terms difficult. However, in contrast to the other approaches, WTP statements allow for a relative comparison of conservation preferences between different study sites, while clearly reflecting the trade-offs between financial and conservation outcomes inherent to conservation. The ease of comparability in combination with the high degree of flexibility in designing WTP scenarios enables researchers to easily enhance this approach with additional experimental interventions, which can yield new insights into how and what might drive changes in conservation preferences.

2.4. Behavioral measures (revealed preferences)

Intuitively one may presume that the evolution of conservation preferences over time would ideally be monitored by observing real-world conservation behavior and assessing the direct links to conservation outcomes, such as a reduction of deforestation or poaching within a randomly assigned trial. Monitoring such outcomes, however, is in many cases practically infeasible, or difficult and costly to implement (Nilsson et al., 2020). While preference measures based on observed behavior provide more objective and reliable behavioral data than the aforementioned approaches based on self-reported statements and intentions, they come with downsides. They are relatively costly to implement, especially when conservation behavior is supposed to be monitored over a longer period of time (Floress et al., 2018). Moreover, conservation projects commonly aim to alter the costs and benefits associated with conservation actions. Differences in conservation behavior between intervention and control populations are likely to reflect changes in monetary (dis)incentives, regulations, or social norms rather than a fundamental shift in the underlying conservation preferences. In other cases, conservation interventions may have already been proven successful in improving environmental quality to an extent that additional behavioral adjustments can hardly be expected, even though the intervention might have in principle strengthened conservation preferences.

2.5. Donations

Real monetary donations towards a conservation cause conceptually represent a specific behavioral measure. Such measures are typically assumed to measure altruism coming at a real personal cost and they have been used in multiple studies from the field of experimental economics to approximate altruistic preferences towards social or environmental causes (e.g. Blanco et al., 2012; Champ et al., 1997; Rode et al., 2008). The cause for which people are asked to donate can be selected in a targeted way such that donations capture underlying conservation preferences as opposed to economic dis(incentives) from conservation. As for WTP measures above, financial constraints may however render money a questionable measure for conservation preferences, e.g., when poor communities do not have the ability to support conservation activities financially (Martínez-Alier, 2002). Recently, studies have employed donation tasks in combination with contingent valuation scenarios in experiments to investigate how WTP bids to support public causes are driven by behavioral factors (Bouma & Koetse, 2019). In the remainder of this article, we aim to showcase how a combination of both WTP measures and donations can be applied to measure and investigate underlying conservation preferences based on a case study on CBNRM in Namibia (Table 2).

3. Case study

3.1. Context of CBNRM in Namibia

CBNRM in Namibia is often considered a success story of nature conservation in Africa (a detailed description of CBNRM in Namibia and our selected conservancies can be found in the Supplementary Online Material (SOM), Section 1). Conservancies within Namibia's national CBNRM program cover a total area of 166,045 km^2, amounting to 52.9% of all communal land in Namibia. However, although some conservancies have successfully developed income sources from tourism or hunting, substantial economic benefits have not materialized for the majority of conservancies. As of 2017, 15 (18%) officially registered conservancies generate no cash income at all, and 45 (54%) are unable to fully cover their operating costs (NACSO, 2018, pp. 56, 65). The financial incentives in conservancies may further decrease if donor and NGO support is reduced over time (Boudreaux & Nelson, 2011). Also, rising wildlife numbers are expected to go hand-in-hand with an increase in human-wildlife conflicts (NACSO, 2018, p. 45).

There have been claims in the literature that conservancies can alter the way people relate to the environment, that they buy into the conservation narrative and start protecting nature out of self-interest (Blaikie, 2006). While this is easily imaginable in scenarios where biodiversity conservation can generate income through e.g., ecotourism, it is worthwhile to investigate whether a deeper change of underlying conservation preferences happens in response to CBRNM policies. As stated earlier, using attitudinal items comes with the risk of demand effects and these items are often not validated but constructed ad hoc. Asking for behavioral intentions is problematic as there are clear laws against certain anti-environmental activities like e.g., fire clearing, or illegal hunting practices. Also, people may be reluctant to talk truthfully about certain activities e.g., those they undertake to prevent their crops or even homes from being damaged by elephants. Thus, one may not get to the underlying conservation preference either. Our preferred measure thus is one that asks people to weigh the costs and benefits of conservation which is unrelated to their local conservancy goals and predominantly generates indirect and negligible ecosystem benefits.

3.2. Design of the contingent valuation survey

We designed the survey instrument with the main purpose of eliciting two different measures for conservation preferences. In particular,

Table 2
Methods for Eliciting Conservation Preferences.

Method	Attitudes	Stated Preferences		Revealed Preferences	
		Self-reported behavior and behavioral intentions	WTP	Observed Behavior	Donations
Advantages	- Fast and easy to collect - Potentially applicable in different contexts (e.g., New Environmental Paradigm Scale, but not always given) - Previously tested (reliability/ validity)	- Change in human behavior is often the targeted outcome of interventions - Relatively easy and fast to collect	- No financial constraints since WTP is hypothetical - Highly flexible and adjustable to specific conservation contexts	- 'First best' solution to measure actual behavioral changes	- Decisions have direct, financial consequences (trade-off) - Donation can be chosen to explicitly rule out any direct, economic benefits
Disadvantages	- Demand effects - Link to actual conservation behavior not always given	- Interventions may change (dis) incentives to engage in a specific behavior (monetary payoffs, social norms, level of env. quality) - Recall bias for self-reported behavior - Demand effects - Difficult to disentangle motivational drivers (conservation preferences and economic benefits)	- Hypothetical Bias - Demand effect - Hypothetical valuation might be unfamiliar (could be reduced with group deliberation and sufficient time)	- Difficult to disentangle motivational drivers (conservation preferences and economic benefits) - Difficult & costly to monitor - Lack of experimental control	- Limited resources reduce the scope to donate even though conservation preferences exist - Costly implementation

we elicited a) the WTP through a hypothetical donation scenario, and b) real donations through including an opportunity to donate at the end of the survey. The additional inclusion of two different deliberation methods (individual vs group-based) as experimental treatments allows us to assess whether potential differences in conservation preferences between conservancy and non-conservancy villages are sensitive to this design dimension. The sequence of our data collection is illustrated in Fig. 1. A more comprehensive description can be found in the SOM, Section 2.

First, we employed a contingent valuation scenario for a one-time donation to a Namibian conservation project, concerned with monitoring and saving the remaining populations of the lappet-faced vulture (*Torgos Tracheliotos*), an endangered bird species native to almost all of Namibia. Exploiting the flexibility of the contingent valuation method, we chose to focus our scenario on a species that is not directly related to the income of village inhabitants in this area by any means (as opposed to e.g., a scenario featuring any of the "Big Five" animals attractive for eco-tourism or other aspects related to residents' livelihoods, such as wildfire protection). This design choice directly relates to our argument regarding behavioral changes driven by altered cost/benefit ratios due to conservation interventions. We thereby eliminate potential issues relating to WTP statements being affected by economic self-interest outside of personal budget constraints. The elicited WTP statements (and donations) should be associated mainly with the non-use values (in particular, ecosystem function and existence values) of the lappet-faced vultures. Furthermore, the WTP measure and the actual donations should focus on a species that is known to respondents within and outside of conservancies, but they should not already contribute through other means to the conservation of the specific species. Given the low awareness and knowledge about the lappet-faced vulture among respondents, it is unlikely that people living in the surveyed conservancies might feel that they already contribute to vulture conservation by means other than donating to the project. Neither of the conservancies has explicitly implemented measures to protect vulture populations or any other endangered species. Targeting a species that is more common and directly associated with CBNRM could have influenced the valuation, as conservancy members may already consider their individual contribution to the protection of the species while deciding on their WTP bids and/or donations.

After respondents were provided with basic information on the lappet-faced vulture, they were asked to answer a set of Likert-scale questions regarding their attitude toward the animal, as well as their personal experience with it. Although no one reported ever seeing a lappet-faced vulture, all respondents indicated that they had seen other kinds of vultures. Thus, in line with our reasoning above, respondents could be expected to value the vultures only for their existence and ecosystem service provision as a contributor to natural pest control (Ogada et al., 2012). We then presented detailed information on the vulture conservation project to the respondents. Immediately before we asked for the WTP amount that respondents hypothetically would be willing to donate to the conservation project, we confronted them with the following cheap-talk script which, as already discussed in section 2.3., have been shown to be an effective tool in mitigating the issue of hypothetical bias in stated preference studies (see e.g., Foster and Burrows, 2017 or Penn and Hu, 2019):

"Before you answer, please think about your response carefully. Keep in mind your personal income situation and alternative possibilities to spend your money. Similar studies showed that people tend to overstate their willingness to pay in hypothetical scenarios, because they do not consider properly how much their spending would affect their personal budget. It is important that you answer as honestly and realistically as possible."

One-time donations allowed us to directly compare the magnitude of the WTP with the donations we elicited at the end of the survey. Naturally, a one-time donation is easier to implement in an area where people have barely any means to make recurring payments, e.g., via mobile-payment applications. To facilitate the respondents' decisions, this was done with the help of so-called payment cards.[2] We tested for the effect of group deliberation on our different measures of conservation preferences by allocating half of the respondents to small groups of three (see Fig. 1) in which they were asked to jointly discuss their hypothetical donation to the vulture conservation project. Participants were allowed to deliberate for a minimum of three to five minutes. After the deliberation, each respondent reported their individual WTP privately. Stated preference studies increasingly apply group settings in which

[2] Payment cards elicit individual WTP responses with the help of a range of values presented to the respondents in the form of a list. Following recommendations from established literature on this topic (Bateman et al., 2002), this elicitation format was chosen over other alternatives because it provides a credible context for the possible donation amounts, avoids starting point bias and reduces unrealistic outlier responses. After pretesting, we decided to present respondents with values ranging from 0 NAD to 40 NAD.

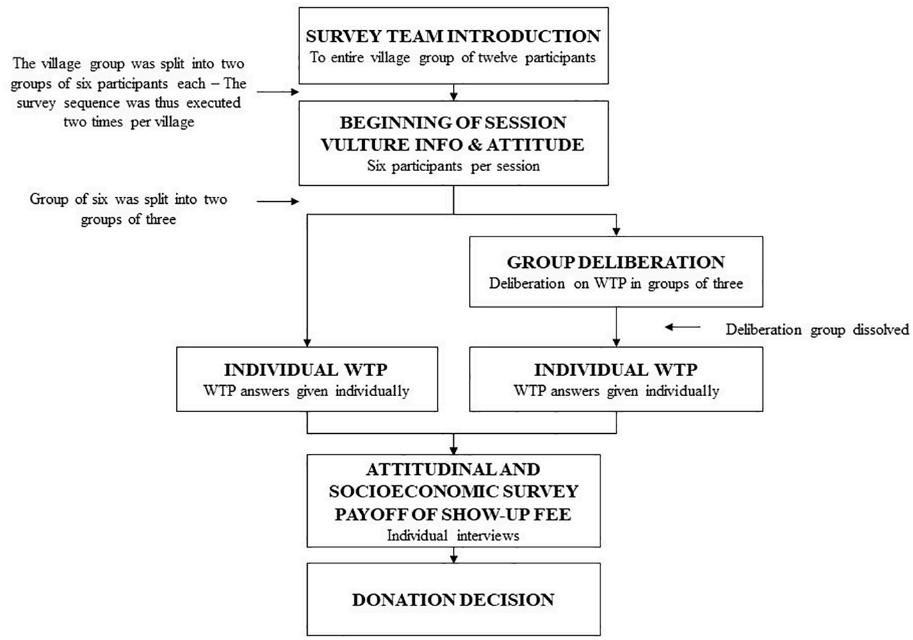

Fig. 1. Illustration of Study Sequence.

respondents are given the opportunity to discuss the costs and benefits associated with the good for which WTP is elicited (Kenter, 2016; Spash, 2007). Group deliberation may remedy some of the potential problems inherent to hypothetical valuation methods, including the often-unrealistic assumption of well-defined and informed preferences among respondents (Schaafsma et al., 2018; Spash, 2007). After the group sessions, the survey interviews continued individually until the end of the session.

After finishing the survey with a set of socio-economic questions, respondents received 30 NAD (4.8 US$ PPP)[3] for their participation at the end of the session and were given the opportunity to donate to the previously presented vulture conservation project (identical to the one in the hypothetical WTP scenario). Respondents donated anonymously by putting money in an envelope and dropping it in a separate box, out of sight of the remaining survey participants and the enumerators. The respective study participant and donation amount could later be linked by an interview ID that was written on the envelope.

The comparison of WTP bids and actual donations allows us to approximate the potential hypothetical bias in WTP bids. As indicated previously, the hypothetical bias states that individuals tend to overstate their WTP in hypothetical scenarios (Carson, 2012b), e.g., due to demand effects (Tourangeau et al., 2000). Such demand effects could be argued to arise both within and outside conservancy villages. In conservancies, respondents might have seen an opportunity to provide a rationale for any kind of government funding (either of monetary nature or via in-kind benefits) in case they indicate positive conservation outcomes by responding accordingly to respective survey items. Outside, villagers on the other hand might have had an interest to respond positively to these survey items to support potential future considerations of their village to become part of a conservancy, as CBNRM conservancies and their alleged benefits are well-known throughout the study area. To mitigate the impact of these demand effects on our conservation preference measures, we clearly stated to not be associated with any kind of governmental agency or NGO that could provide funding and/or help in the establishment of new conservancy areas.

3.3. *Study sites for our comparative case study*

Our research was concentrated on two conservancies located in northeastern Namibia: The Joseph Mbambangandu Conservancy (JMC), established in 2004, and the George Mukoya Conservancy (GMC), established in 2005. These conservancies were chosen due to being located within the study area of a larger research project but are not atypical when compared to other Namibian case studies. Based on the available data, the GMC and especially the JMC belong to the lower-achieving categories of conservancies in terms of financial revenues (see SOM, Section 1.2). We consider both conservancies prime examples for showcasing the importance of conservation preferences as positive conservation outcomes will likely rely on these underlying drivers due to the absence of tangible financial incentives (corresponding to case C or D in Table 1).

Sampling was conducted in two stages. First, we sampled all four settlements located in the JMC, along with two of the eight villages located in the more remote, barely accessible, and less densely populated GMC[4]. In order to identify "control villages" outside of the conservancies, 58 villages in relative proximity to the main road and the two conservancies were identified using official records. Five localities situated close to the JMC, and two localities situated near the GMC were then randomly selected for the final sample. Twelve participants per village were randomly recruited during village meetings, which were organized by the respective local headman, resulting in a total sample size of 156 observations. Following local ethical research standards, the field team visited all villages a few days before the actual survey work. This was done to acquire approval from the respective village headman, thereby providing them with time to inform all village members about the opportunity to participate and the random nature of the sampling process. All respondents provided informed consent before entering the survey sessions, and they were filled in on data confidentiality and anonymization. Each respondent was remunerated with a fixed participation fee of 30 NAD, which roughly translated to the daily wage for regional farm workers at the time of the study. All interviews took place

[3] At the time of the data collection, the monthly minimum wage for farm workers was 900 NAD.

[4] Of the eight villages in the GMC, two were not accessible by the research team. The two sampled villages were randomly selected from the remaining six villages.

in September 2017 and were held in the local RuKwangali language with the help of three trained research assistants. The socio-economic characteristics of participants by conservancy status are provided in the Supplementary Online Material (SOM Table S5).

3.4. Hypotheses and analytical approach

In line with the performances of the two surveyed conservancies, we expect potentially two mechanisms to affect conservation preferences. On the one hand, the establishment of conservancies may have strengthened overall conservation preferences through training and awareness campaigns as well as deliberation within villages about the value of nature. On the other hand, the fact that the promised benefits of conservancies did not materialize may lead to frustration, potentially reducing conservation preferences. We thus formulate an open hypothesis, acknowledging the two directions of effects:

H1: The average conservation preferences of residents residing within conservancies differ from those of residents living outside of conservancies.

Collecting data on the stated, hypothetical WTP and eliciting actual donations, allows us to approximate the hypothetical bias at the individual level. If we are able to detect systematic differences in the hypothetical bias between study sites, it could be argued that the hypothetical WTP statements cannot provide reliable approximations of relative changes in conservation preferences on their own. Differences in hypothetical bias between study sites could arise from potential demand effects. E.g., within CBNRM conservancies residents may be more susceptible to a social desirability bias. They may feel obliged to answer to the hypothetical questions in line with a positive impact of conservancies and/or may be motivated by reciprocal motives (stating a high valuation of nature as they received funding for the establishment of conservancies by donor agencies and NGOs). This difference in hypothetical WTP would then result in a larger hypothetical bias. Second, residents within conservancies could exhibit more well-defined preferences for ecosystem services. This could be a result of educational interventions and/or collective discussions regarding the significance of conservation activities, both in terms of use and non-use values. Respondents from conservancies could therefore better estimate their true WTP in a hypothetical task. We therefore formulate the following hypotheses:

H2: The hypothetical bias of residents within nature conservancies differs from those living outside of conservancies.

Lastly, we randomly assigned half of our sampled respondents to a treatment condition in which they could discuss their statements with a randomly selected group of fellow community members. Following the argument brought forward in (Spash, 2007), stated preference techniques are under scrutiny for assuming well-informed preferences among respondents, and deliberative monetary valuation (DMV) is presented as a potential remedy to this issue. In the context of our case study, the average effect of group deliberation on the conservation preference measures may differ between study sites. After years of being directly exposed to pro-conservation education and incentivization, people living in conservancies might have more well-defined conservation preferences when compared to people living outside conservancies. If the establishment of conservancies led to more well-defined preferences, group deliberation has potentially a smaller effect on stated WTP and donations in conservancies. Thus, while we expect group deliberation to have a positive effect on both WTP and donations in both village types, we expect these effects to be stronger in magnitude outside of conservancies.

H3: Group deliberation has a positive effect on respondents' WTP and donations, but one that is stronger outside than within conservancies.

We report both non-parametric tests (Mann-Whitney-U tests for continuous outcomes, chi-squared tests for categorical outcomes) and multivariate regressions as hypotheses tests. For Hypotheses 1 we focus on two related, yet unique outcomes: a) the binary decision whether to state a WTP greater than 0 or donate anything, tested with a Probit model (Eq. (1), and b) the WTP and donation amount as a continuous measure of preferences, tested with a Tobit model (Eq. (2).[5]

The probability of respondent i stating a positive WTP or donation is expressed as a function of various explanatory variables, where $\Phi(\cdot)$ denotes the cumulative standard normal distribution:

$$P_i(Y_i > 0|X_i) = \Phi(\beta_0 + Conservancy_i\beta_1 + X_i\beta) \tag{1}$$

where $Conservancy_i$ is a dummy variable (taking 1 if the respondent i lives in a conservancy and 0 otherwise), and X_i is a vector of control variables including gender, age, age^2, number of children, household size, education (years of schooling), monthly expenses, and deliberation method (individual vs. group). We report the marginal effects of the Probit model that can be interpreted as the change in the probability of Y $>$ 0 due to a one-unit increase in the explanatory variable. Standard errors are clustered for all regression models at the session level.

As a continuous outcome measure, we use the WTP and donation amount of respondent i respectively (y_i) and estimate Tobit models with 0 as lower bound and 40 (30 for the donation) as upper bound to account for the censoring of the data. The Tobit model can be expressed as a latent variable regression model with upper limit a, and lower limit b (Cong, 2001). The latent continuous variable y_i^* cannot be observed over its entire range, but only within its upper and lower limit:

$$y_i^* = Conservancy_i\beta_1 + X_i\beta + \varepsilon_i \tag{2}$$

$$y_i^* = \begin{cases} y_i if b < y_i < a \\ b if y_i \le b \\ a if y_i \ge a \end{cases} \tag{3}$$

where $Conservancy_i$ is a dummy variable (taking 1 if the respondent i lives in a conservancy and 0 otherwise), and X_i is a vector of control variables including gender, age, age^2, number of children, household size, education (years of schooling), monthly expenses, and deliberation method (individual vs. group). We report the marginal effects that estimate the change in the mean of the latent dependent variable y_i^* due to a one-unit increase in the explanatory variable.

To test Hypothesis 2 we focus on two different measures for the hypothetical bias at the respondent level, namely a) the difference between the stated WTP and donation (y_i^{bias}, see Eq. (4), and b) the absolute difference between the stated WTP and donation ($y^{abs-bias}{}_i$, *see* Eq. (5). We estimate the corresponding Tobit models with these two different dependent variables censored at the respective limits ($-$30 and 40; 0 and 40).

$$y_i^{bias} = WTP_i - donation_i \tag{4}$$

$$y^{abs-bias}{}_i = |WTP_i - donation_i| \tag{5}$$

Lastly, we test Hypothesis 3 by estimating the following Tobit regression model for WTP and donations:

$$y_i^* = Conservancy_i\beta_1 + Group_i\beta_2 + Conservancy_i{}^*Group_i\beta_3 + X_i\beta + \varepsilon_i \tag{6}$$

where $Conservancy_i$ is a dummy variable (taking 1 if respondent i lives in a conservancy and 0 otherwise), and $Group_i$ is a dummy variable (taking 1 if respondent i participated in group deliberation and 0 otherwise). The interaction $Conservancy_i{}^*Group_i$ tests, Hypothesis 3, whether group deliberation has a different effect within conservancies relative to outside. X_i is a vector of control variables including gender, age, age^2, number of children, household size, education (years of schooling), and monthly expenses.

[5] Both of these model classes have a rich history of application in conservation research, especially in studies dealing with WTP estimates. Tobit models account for the censored nature of WTP data (Halstead et al., 1991; Norris and Batie, 1987), while Probit models have previously been used to investigate zero and/or protest responses to WTP elicitation questions (Cho et al., 2008).

3.5. Results

3.5.1 WTP and donations

Fig. 2 illustrates the distribution of WTP and donations by conservancy status (see Fig. 2, Panel A and B) as well as the corresponding averages (Fig. 2, Panel D and E; Hypothesis 1). The average WTP in non-conservancies and conservancies are almost identical (Mean = 4.81 vs. 4.75 NAD, SD = 7.55 vs 7.96 NAD, Mann-Whitney-U (MWU) test p = 0.23). However, the average donation from respondents living outside of conservancies is larger (Mean = 3.16 vs. 2.39 NAD, SD = 4.38 vs. 3.55 NAD, MWU p = 0.08). Next, we focus on the categorical outcome whether respondents stated a WTP larger zero and donated anything. Respondents in conservancies are more likely to state a WTP of zero (31%) and donate nothing (44%) compared to the control villages (11% and 26%, respectively). These discrepancies are significantly different, as indicated by chi-squared tests (WTP: p = 0.00, donation: p = 0.02).

In Table 3, we present Probit and Tobit regression models where we control for additional socio-economic covariates (gender, age, household size, education, and monthly expenses). Here, we find that while respondents in conservancies are less likely to state a positive WTP and donate something, average WTP and donations are not significantly different between conservancy and non-conservancy villages.

Finding 1: There is evidence for lower conservation preferences within conservancies based on the hypothetical WTP and actual donation decisions. We observe larger shares of zero donations and zero WTP in conservancies, thus, partially confirming Hypotheses 1.

3.5.2 Hypothetical bias

We can compute the hypothetical bias at the individual level by subtracting the donations from the WTP measure. A positive hypothetical bias thus indicates that respondents overstated their hypothetical WTP in relation to their actual decision. In our sample, 98 out of 156 (63%) respondents show no hypothetical bias. Overall, 45 respondents (29%) have a positive hypothetical bias and 13 (8%) respondents have a negative bias. See Fig. 2, Panel C for the distribution of this variable by conservancy status and Fig. 2, Panel F for the mean hypothetical bias by conservancy status.

On average however, actual donations are substantially lower than the stated WTP (2.80 vs. 4.78 NAD), and the share of respondents who donated nothing is larger than the share of respondents who stated a zero WTP (34.6% vs. 19.9%). The difference between average WTP and donations is statistically significant (Wilcoxon signed-rank test, $p < 0.01$).

Using WTP as a measure for conservation preferences could be particularly contested if the hypothetical bias would be more or less pronounced in conservancy villages (Hypothesis 2). Here, this difference is not statistically significant (Mean = 1.66 vs 2.36 NAD, SD = 8.42 vs 8.07 NAD, MWU p = 0.75). These findings are also confirmed by Tobit regressions, including socio-economic controls and taking the absolute hypothetical bias as an alternative dependent variable (see Table 4).

Finding 2: There is a significant hypothetical bias, which does not differ significantly between conservancy and non-conservancy villages. We thus do not find support for Hypotheses 2.

3.5.3 Group deliberation

Next, we assess the impact of group deliberation on outcomes. A zero WTP is slightly more common under group than individual deliberation (23 vs 16.7%), even though this difference is not statistically significant (chi-squared test, p = 0.316). The share of respondents who do not donate anything is similar across treatments (group: 35.9%, individual: 33.3%, chi-squared test p = 0.736). We find that prior group deliberation has a negative effect on the average stated individual WTP and donation. The stated WTP is on average 2 NAD lower for the group deliberation treatment (3.76 vs. 5.8 NAD), and donations are on average 1.27 NAD lower for group compared to individual deliberation (2.17 vs. 3.44 NAD). However, these two differences are only statistically significant at the 0.1 level (MWU, WTP: p = 0.09, donation: p = 0.10) based on non-parametric tests and insignificant in regression analysis (see Table 3).[6] The hypothetical bias (measured as the difference between WTP and donations) is slightly lower under group deliberation (1.60 vs. 2.36 NAD), but this difference is not statistically significant (MWU, p = 0.52).

Through Tobit regressions, we also assess whether group deliberation has different effects in- and outside of conservancies (Hypotheses 3). In the respective models (see Table 3), we include an interaction between the group deliberation and conservancy variable that measures the additional impact of group deliberation in conservancies relative to the effect in non-conservancies. Despite a tendency that group deliberation lowers both WTP and donations in conservancies more than outside of conservancies, these effects are not significant.

Finding 3: We observe a tendency for lower conservation preferences if participants can deliberate in groups before the elicitation of WTP and donations. We do not find evidence that group deliberation has a different effect within and outside of conservancies (Hypotheses 3).

4. Discussion

This article conceptualizes conservation preferences as the drivers of individual decision-making and behavior (Falk et al., 2018; Hausman, 2011) that render people supportive of conservation interventions and also to act in favor of conservation outcomes (Selinske et al., 2020). While they may reflect a wide array of values that people attribute to the conservation of nature (Pascual et al 2023), we argue that they should be distinguishable from preferences for personal economic benefits people may expect to receive from conservation. The article describes and tests hypothetical WTP and real donations within a contingent valuation scenario as a way to measure relative changes in conservation preferences in the context of conservation projects. The major advantage of these two measures is that they are insensitive to demand effects or the altered cost/benefit ratio affecting conservation behavior that is inherent to such projects (Silva & Mosimane, 2014). Our case study in Namibia provides evidence that respondents in conservancies are both less likely to state a positive WTP and less likely to donate any money to the conservation cause. Based on the arguments presented throughout this article, these findings suggest lower average conservation preferences in CBRNM conservancies or at least no increase in conservation preferences due to the existence of the conservancy. We have already alluded to a potential explanation for this finding in the introduction to this paper: it might well be that the conservancy members' average conservation preferences have been undermined by the fact that initially promised economic benefits to the respective communities have failed to materialize over a timeframe of more than ten years (Hegwood et al., 2022; McShane et al., 2011; Muradian et al., 2013).

The comparison between hypothetical WTP and real donations further shows that the hypothetical bias in conservancies is not different from the average bias we measured outside conservancies. From a practical perspective, this finding suggests that WTP elicitation within hypothetical scenarios (i.e., without additional real donation tasks) might be sufficient to assess relative differences in conservation preferences between members and non-members of community-based

[6] Group and individual deliberation treatments were randomly assigned at the individual level. Respondents in the two treatments had similar observable socio-economic characteristics (see SOM Table S6 for details).

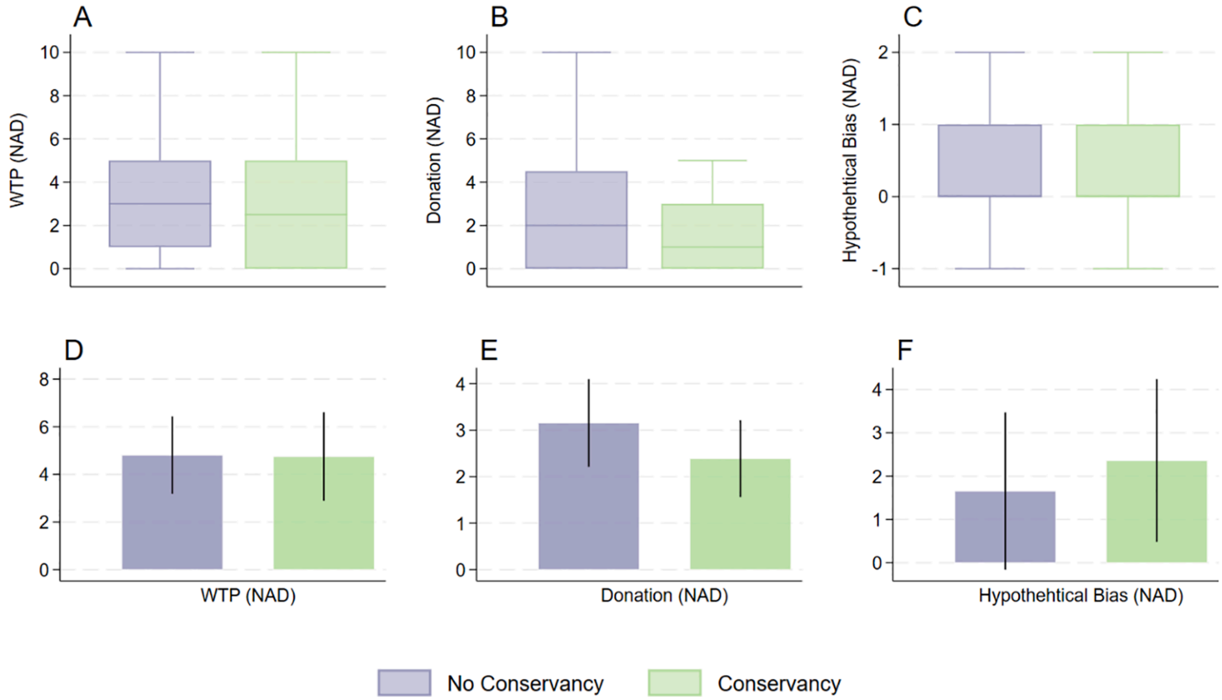

Fig. 2. A) Boxplot of WTP by conservancy status (outliers not displayed), B) Boxplot of donations by conservancy status (outliers not displayed) C) Boxplot of hypothetical bias by conservancy status ($\boldsymbol{WTP_i} - \boldsymbol{donation_i}$), D) Mean WTP by conservancy status (with 95% CIs), E) Mean donation by conservancy status (with 95% CIs), F) Mean hypothetical bias ($\boldsymbol{WTP_i} - \boldsymbol{donation_i}$) by conservancy status (with 95% CIs).

Table 3
Probit (DV: binary outcome positive WTP/donation yes/no; reported as marginal effects) and Tobit model (DV: donation amount) results.

	WTP			Donation		
	(1) Positive WTP (y/n)	(2) Amount (NAD)	(3) Amount (NAD)	(4) Positive Donation (y/n)	(5) Amount (NAD)	(6) Amount (NAD)
Conservancy	−0.21**	−1.73	0.97	−0.21*	−1.49	−1.04
	[−0.34,−0.07]	[−4.44,0.98]	[−3.41,5.36]	[−0.39,−0.02]	[−3.56,0.59]	[−3.93,1.85]
Group Deliberation	−0.05	−2.48	−0.11	−0.02	−1.63	−1.25
	[−0.21,0.10]	[−6.13,1.17]	[−4.50,4.28]	[−0.14,0.10]	[−3.42,0.16]	[−3.47,0.98]
Conservancy × Group Deliberation			−5.45			−0.90
			[−12.99,2.09]			[−4.98,3.19]
Obs.	155	155	155	155	155	155
Left-censored		31	31		54	54
Right-censored		5	5		0	0
F-Stat		1.75	1.68		2.80	3.82
Chi2	42.41			17.90		
p-Value	0.00	0.08	0.09	0.04	0.00	0.00
Pseudo R2	0.20	0.02	0.02	0.07	0.03	0.03

Robust standard errors clustered at the session level; $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$; controls: gender, age, age^2, number of children, household size, education (years of schooling), monthly expenses; Model 1 and 4 report the marginal effects of Probit regressions, while the other models are Tobit regressions. Full models reported in the SOM Tables S1, S2 and S4.

conservation projects. The fact that we observe a positive hypothetical bias in line with the broader contingent valuation literature, suggests that the donation task probably produces the more reliable measures and that attitudinal items are similarly prone to demand effects.[7]

Our results regarding the effects of deliberation on WTP and donation estimates are mixed, and further research is warranted to come to robust recommendations. While we observe a tendency that group deliberation leads to both lower average WTP bids and donations, the effect goes in the opposite direction as we have initially anticipated. While empirical studies have shown that groups can be more selfish relative to individuals (Vollstädt & Böhm, 2019), we additionally look at another circumstance to provide a potential explanation for this finding: Group deliberation does not seem to reduce the hypothetical bias between stated WTP and the real donations. It could thus be that individuals both inside and outside of conservancies may indeed struggle to form and express ad hoc preferences for unfamiliar goods, which is argued to be potentially remedied by group deliberation (Spash, 2007). Our results regarding the slightly lower WTP bids and donations following group deliberation might be a result of the realization that other group members in fact have relatively weak conservation

[7] Meta-studies on hypothetical bias report that individuals overstate their WTP by a factor of 3 on average, although differences between studies are substantial depending on the type of good studied and the elicitation method applied (List and Gallet, 2001; Murphy et al., 2005; Schläpfer and Fischhoff, 2012). We find that respondents overstate their WTP by a factor of 1.71.

Table 4
Tobit regression results with the hypothetical bias ($WTP_i - donation_i$) as dependent variable.

	Hypothetical Bias (1)	Absolute Hypothetical Bias (2)
Conservancy	0.459	−0.317
	[−1.592,2.510]	[−5.571,4.938]
Obs.	155	155
Left-censored	0	98
Right-censored	5	5
F-Stat	2.961	1.235
p-Val	0.003	0.278
Pseudo R2	0.009	0.019

Robust standard errors clustered at the session level; $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$; controls: gender, age, age^2, number of children, household size, education (years of schooling), monthly expenses; Full models reported in the SOM Table S3.

preferences. If deliberation would be effective in mitigating demand effects, we would likely have observed a larger hypothetical bias within the group deliberation treatment.

Naturally, all of these findings assume that our approach to measuring conservation preferences can be regarded as internally valid in the first place. At first glance, our finding that the measurements of individual monthly expenditures are neither correlated with WTP nor donations may indicate a lack of internal validity. For two reasons, we believe however that this must not necessarily be the case: First, the expenditure measurement might not be a valid indicator of internal validity Gathering income estimates in developing countries is known to be challenging, as households are typically characterized by varying income levels from multiple sources. Because of that, employing consumption expenses instead of income has been considered more reliable (Haughton & Khandker, 2009). Given the limited timeframe to elicit socioeconomic characteristics of respondents in our survey, we relied on a crude measure of expenses through a straightforward mode of questioning. We thus believe that our measure includes additional statistical noise compared to a more detailed approach of eliciting expenses for different categories. Second, even though we observe some degree of variation for the expense variable, the range of observed values remains relatively narrow, likely due to the relatively homogenous nature of our sample. The lack of correlation might therefore be just a result of inelastic demand at the observed variation of income in our sample (Greenstone & Jack, 2015). We would also like to highlight that we collected data on general attitudes towards nature and nature conservation. To fulfill internal validity criteria, one would expect WTP and donations to be correlated with such measures. We do not report average responses to these attitudinal statements in the main text of this paper as three out of these four statements show very little variation (see SOM Table S7), potentially because self-reported agreement with statements is prone to demand effects. To our knowledge, there is no validated and widely accepted attitudinal scale for attitudes towards nature for our study region, which poses a challenge for studies that aim to elicit such attitudes as covariates and do not have the resources to develop and validate contextually adapted measures.

Another reason to treat the results of our case study with at least some degree of caution is the relatively small sample size and hence limited statistical power. Drawing robust conclusions on design considerations (WTP vs. donation; group vs. individual deliberation), would ideally be based on a larger sample and multi-site studies covering different contextual variations. Apart from that, one would ideally have applied a random assignment of the conservation intervention or employed longitudinal data that would allow the application of more robust (quasi)-experimental methods (Ferraro, 2009; Greenstone & Gayer, 2009). Nonetheless, we find a few reasons to assume pre-existing differences between villages with respect to our outcome variables: First, the control villages are located close to the conservancy villages. On average, the control villages are located 18.4 km away from the nearest conservancy village (min. 6.3 km; max. 36.2 km). As a result, villages inside and outside of conservancies likely share similar socioeconomic and ecological conditions and thus have equal potential for CBNRM success and economic development in general. Second, both our surveyed conservancies were initiated by community members rather than external stakeholders, such as NGOs or the Namibian government, who could have targeted certain communities selectively. To the best of our knowledge, there have been no (failed) attempts to establish conservancies in any of our control villages. Thus, there might if at all be a positive selection bias stemming from the initial group of conservancy founders. Assuming the non-random establishment of conservancies (i.e., if conservation preferences were initially higher in the pre-CBNRM conservancy villages), the true conservancy effect would be biased upwards and thus be lower than what we observe based on our data.

On the subject of the external validity of our case study, we would again like to point out that we did purposefully choose the two conservancies because they do not report noteworthy financial revenues through e.g., ecotourism, so we assume the formation of conservation preferences especially important (see cases C and D in Table 1). Against this backdrop, we could only reasonably generalize our findings for conservancies with comparable financial performance. Many of the Namibian conservancies, especially in the Kunene and Erongo regions of northwestern Namibia, report significantly higher financial revenue streams (NACSO, 2018). Within the subgroup of conservancies comparable to the ones we have sampled, we do however believe that are results can be regarded as externally valid, as we have implemented a random sampling mechanism. We have also identified two studies in which researchers have elicited a one-time WTP for the conservation of threatened vulture species using contingent valuation, which we therefore regard at least somewhat comparable to our approach. While one Israeli study using payment cards has produced comparable average WTP bids for vulture conservation when adjusting WTP bids for inflation and national per capita income (Becker et al., 2009), another study reports significantly higher average WTP bids (Zambrano-Monserrate, 2020). However, this study elicited the WTP via dichotomous choice with randomly allocated bids including relatively high monetary figures when compared to our study. WTP elicitation methods of this type are also known to yield higher WTP bids than payment card approaches (Blaine et al., 2005).

We can also not rule out that our WTP estimates might have been affected by the participation fee of 30 NAD which was, of course, known to the respondents prior to the start of the individual surveys. Naturally, this circumstance could have the potential to negatively affect the external validity of the participants' stated WTP bids. However, such effects should be leveled between respondents from both conservancy and non-conservancy villages. Thus, with regards to our study's main research objective, which was to investigate mean WTP differences between village types, we do not regard this as a major cause for concern. However, we did try to mitigate such effects in different ways: We addressed potential WTP anchoring effects caused by the participation fee by offering WTP bids higher than the fee on our payment cards. The observed distribution of our WTP estimates towards the lower end of the payment card spectrum does indicate that anchoring towards the 30 NAD value did not seem to be a prevalent problem. Also, respondents were made aware of the payment a few days prior to the study (during the recruitment phase). This time horizon likely reduced the "windfall gain" effect compared to a "surprise" earning at the time of the interview, as participants had a longer time to internalize the money and make plans for its use (Arkes et al., 1994). Still, future studies could further try to circumvent problems of this nature by adjusting the survey sequence, tweaking the elicitation method of the contingent valuation method (e.g., open-ended, bidding game, or dichotomous choice), or the mode of reimbursement to the study participants.

5. Conclusion

In this article, we showcased how the combination of a contingent valuation scenario and a real donation task can be applied to measure and investigate conservation preferences. We argued that knowledge about such preferences can help conservation policymakers better evaluate the long-term impacts of conservation projects, which is especially crucial in cases where financial incentives to affected communities do not materialize as initially promised. Our proposed approach can serve as a complement to other preference measurement instruments in conservation contexts, as it avoids various pitfalls associated with other methodological approaches, which have been highlighted in section 2 of the paper at hand. The outcomes of the case study surrounding the Namibian CBNRM program, in which we have tested the proposed measurement approach, suggest that conservation preferences can indeed fail to materialize or even be undermined.

We argue that a careful design of the (hypothetical) donation cause is of crucial importance. Following our arguments brought forward specifically in Sections 1 and 2 of this article, future studies should aim to further minimize the potential role of direct use values (in the sense of a direct economic benefit of the donation cause to the respondents) in contingent valuation scenarios aiming to elicit conservation preferences. It should furthermore be ensured that the species or ecosystem central to the valuation scenario is comparably familiar to respondents in the relevant sites but not subject to conservation efforts already. We ultimately hope that the conceptualization of conservation preferences and the methodological contribution brought forward in this article can inform and inspire future studies that want to measure conservation preferences as an outcome of policy intervention.

CRediT authorship contribution statement

Marco Nilgen: Writing – review & editing, Writing – original draft, Project administration, Methodology, Formal analysis, Data curation, Conceptualization. **Julian Rode:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Conceptualization. **Tobias Vorlaufer:** Writing – review & editing, Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Björn Vollan:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2024.101621.

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