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A comparison of latent class and mixed logit models

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Using Network of Species Interactions to Value Biodiversity Conservation in a Megadiverse Country: A Comparison of Latent Class and Mixed Logit Models

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Abstract

This study examine whether different biodiversity proxies—flora, fauna, habitat, and functionality—satisfy scope sensitivity and plausibility criteria in willingness to pay (WTP) estimation using a choice experiment in Manu National Park, Peru. We introduce the *network of species interactions* as a proxy for functionality and apply latent class (LC) models, including attribute non-attendance (ANA), to account for preference heterogeneity. Our results indicate that interaction networks are the only proxy consistently meeting both validity criteria across all specifications. LC analysis reveals two segments: one (74.4 %) displaying coherent, scope-sensitive WTP across biodiversity attributes and another (25.6 %) less engaged, disregarding standard proxies but still valuing interaction networks. Even under ANA constraints, networks remain salient for less attentive respondents, underscoring their cognitive accessibility in complex ecological contexts. These findings highlight the methodological and policy relevance of functionality-based proxies for biodiversity valuation in megadiverse environments, where conventional measures may fail to elicit behaviorally consistent responses.

1. Introduction

From an economic perspective, biodiversity conservation can be considered a public good since its consumption does not entail rivalry or exclusion (Bishop & Welsh, 1992; Dasgupta, 2021; Strange et al., 2024). Public goods lack markets that could provide information regarding their relative scarcity (Arrow et al., 1996; Hanley & Perrings, 2019; Dasgupta, 2021), which makes it difficult to estimate their economic value and evaluate policies oriented to their conservation (Fraser et al., 2023).

Since biodiversity is an essential input for many ecosystem services, its economic value can be estimated based on the value of such ecosystem services (Bateman & Mace, 2020). Nevertheless, several attempts have been made to value biodiversity directly (Jacobsen et al., 2008; Jacobsen et al., 2011; Bakhtiari et al., 2014; Remoundou et al., 2015; Strange et al., 2024). In many of these studies, biodiversity conservation is a source of nonuse value (existence and bequest); therefore, economic value estimation has relied on stated preference (SP) techniques, such as contingent valuation (CV) or choice experiments (CEs).

SP techniques require the implementation of surveys that present hypothetical markets in which respondents declare preferences for changes in the quantity or quality of biodiversity proxies (Kahneman & Knetsch, 1992; Hanemann, 1994; Rolfe & Windle, 2012; Martin-Ortega et al., 2015). Furthermore, CE, which is a more recent technique compared with CV, can identify tradeoffs between the attributes of alternatives (Jin et al., 2018) and has been used more profusely in recent investigations (Christie & Rayment, 2012; Czajkowski et al., 2014; Emang et al., 2020; Dechasa et al., 2021). Discrete CEs allow for the structured evaluation of complex, non-market environmental attributes, making them particularly suitable for testing ecological constructs such as species interaction networks.

Previous research using SP to value biodiversity conservation is extensive and diverse (Jorgensen et al., 2001; Jin et al., 2006; Olar et al., 2007; Bakhtiari et al., 2014; Borzykowski et al., 2018; Vedogbeton et al., 2020). Rather than attempting an exhaustive review, this study concentrates on two critical themes within this literature: (i) how biodiversity is represented in SP studies, and (ii) how different proxies affect the capacity of SP designs to meet validity criteria such as sensitivity to scope and plausibility.

A growing consensus in the literature stresses that how biodiversity is represented in valuation studies affects the estimated economic value (Strange et al., 2024). Frameworks such as the essential biodiversity variables (Turak et al., 2017) promote the inclusion of functional and holistic indicators of biodiversity change. Among these, the network of species interactions—a proxy capturing ecosystem functionality and resilience—remains underexplored in SP studies, despite its theoretical relevance.

However, different representations of biodiversity in SP will present some challenges. SPs are based on the random utility model (RUM) proposed by McFadden (1974), which allows respondents to express their preferences in monetary terms (i.e., willingness to pay (WTP) and willingness to accept (WTA)). These values have been used to evaluate public policies for biodiversity conservation (Rudd, 2009; Lew et al., 2010; Wallmo & Lew, 2016; Spencer-Cotton et al., 2018). Nevertheless, SP applications have been criticized for generating results inconsistent with the economic theory (Diamond et al., 1993; Adams et al., 2008; Nijkamp et al., 2008; Szabó, 2011; Ferrini & Turner, 2018).

Arrow et al. (1993), in a report for the National Oceanic and Atmospheric Administration (NOAA), and Johnston et al. (2017) suggested a series of recommendations to ensure that the results of an SP study can be used to inform public policy. This study focuses on two of these issues: ***rationality*** and ***plausibility***.

Rationality refers to the results consistent with the assumption that an individual should pay more for more units or higher quality of goods. This is also called *sensitivity to scope* (Pouta, 2005; Amiran & Hagen, 2010; Lew & Wallmo, 2011; Ojea & Loureiro, 2011; Hjerpe et al., 2015). Some researchers have claimed that when interviewees do not perceive changes in a good, it is a symptom of conflict with behavioral economic theory that weakens the results of SP studies (Loomis et al., 1993; Ojea & Loureiro, 2009; Morse-Jones et al., 2012). Others have argued that a reduction in marginal utility (a sign of satiety) can explain insensitivity to scope (Rollins & Lyke, 1998; Jorgensen et al., 2001; Wheeler & Damania, 2001; Olar et al., 2007; Jacobsen et al., 2011; Ressurreição et al., 2011). This satiety could occur after passing a certain threshold; for instance, a minimum viable number of individuals in a species' population (Ojea & Loureiro, 2009).

Insensitivity to scope has been presented in many SP studies in different contexts and is particularly pervasive in biodiversity conservation valuation (Kahneman & Knetsch, 1992; Desvousges et al., 2012; Martin-Ortega et al., 2015; Borzykowski et al., 2018; Tonin, 2019). The problem appears to have been exacerbated by the complexity of representing biodiversity in SP studies (Nunes & van den Bergh, 2001; Jacobsen et al., 2008; Bartkowski et al., 2015).

For instance, *species* and *habitat* are two common representations of biodiversity. Economic valuation studies referencing biodiversity with *species* have used different subcategories, including the number of flora or fauna species affected by an intervention (Christie et al., 2006; Christie et al., 2007; Morse-Jones et al., 2010; Cerda & Losada, 2013; Hausmann et al., 2017), iconic or emblematic species, and broad categories of fish (McDaniels et al., 2003) and mammals (Tanguay et al., 1993; White et al., 2001; Giraud & Valcic, 2004; Boxall et al., 2012; Frontuto et al., 2017). Other studies have used a small number of species in a specific location or even a whole country (Boyle et al., 1994; Jakobsson & Dragun, 2001; Hanley et al., 2003; Olar et al., 2007; Hoyos et al., 2012; Vargas & Díaz, 2014; Yao et al., 2014). Regarding *habitat,* studies have characterized biodiversity by the size of an ecosystem (length or surface) (Rolfe et al., 2000; Rolfe & Windle, 2012; Rogers et al., 2013; Shoyama et al., 2013; Estifanos et al., 2018); iconic regions such as wilderness areas (Diamond et al., 1993; McFadden & Leonard, 1993; Gilbert et al., 1994; McFadden, 1994), rivers (Brown & Duffield, 1995), wetlands (Powe & Bateman, 2004; Whitehead et al., 2009; Pattison et al., 2011), and forests (Adams et al., 2008; Borzykowski et al., 2018); and places that species traditionally inhabit (Meyerhoff et al., 2009; Thiene et al., 2012; Hausmann et al., 2017; He et al., 2017; Pakalniete et al., 2017).

Notably, these common proxies seem to be insufficient for representing biodiversity. A common concern about these definitions is that they are not well linked to a more ecological definition of biodiversity.[1] *Species* representations have often been equivalent to a single and plain component of biodiversity or the number of individuals of one or more species. *Habitat* has usually referenced a broad area with a nonspecific definition of biodiversity (Ring et al., 2010; Bartkowski et al., 2015). Both approaches can generate representations of levels of biodiversity that could be indistinguishable from interviewees' perspectives and become a source of insensitivity to scope. For example, biodiversity representations from uncharismatic and unrecognized species or places (habitats) could include variations that are perceived to be equal by interviewees (Boyle et al., 1998; Jacobsen et al., 2008; Ojea & Loureiro, 2009; Morse-Jones et al., 2012). In this sense, limitations in interviewees' adequate appreciation of biodiversity and hypothetical markets could translate into null or even negative marginal utility, obtaining responses that contradict a well-being perspective (Carson & Mitchell, 1995; Morrison, 2014; Mwebaze et al., 2018).

In this paper, in addition to common proxies of the number of threatened flora and fauna species, we include the *network of species interactions* measured as the percentage of persistent species encounters in the community network. Although several authors have suggested using functionality to measure biodiversity (Czajkowski & Hanley, 2009; Rambonilaza & Brahic, 2016; al., 2018), no consensus has emerged regarding its definition in the valuation literature. To the best of our knowledge, this is the first study to use this proxy of biodiversity based on the foundations of ecology and ecosystem architecture of Jordano (2016). Loss of key ecological interactions may precede local extinction of partnering species that depend on the key ecological services provided, signaling the loss of ecological functions (Jordano, 2016). We argue that network of species interactions represents ecological community structure and biodiversity functionality, as described by Rajmis et al. (2009) and Bakhtiari et al. (2014). The concept of functionality reaches beyond isolated groups of species or areas, aiming to capture agents' interrelationships that generate stability and resilience in ecosystems

[1] According to the Convention on Biodiversity (CBD, 1992), biodiversity is defined as "the infinite variety of life forms; genetic diversity–a variation of genes within individual species, species diversity–the variety of species in flora and fauna, and ecosystem diversity–the variety of ecosystems, such as rainforests, coral reefs, and deserts, that exist on our planet. This biological diversity is the sine qua non for the resilience of ecosystems and life forms and their ability to prevent and to recover from disasters and adverse conditions."

to represent changes and impact on human well-being more adequately (Bartkowski et al., 2015). Whether described qualitatively or quantitatively, functionality can be a better option for representing biodiversity, although interviewees might not be familiar with the concept (Ring et al., 2010; Jordano, 2016). Some previous evidence has suggested otherwise (Rajmis et al., 2009; Bakhtiari et al., 2014; Bartkowski et al., 2015; Lavado et al., 2021). More importantly, representing a network of species interactions could be useful, particularly in valuation applications concerning megadiverse environments[2]. In these contexts, biodiversity loss may be perceived differently by respondents due to the overwhelming number of species involved, which can challenge their cognitive ability to process changes based on species counts alone. We hypothesized that this definition of functionality would respond better to scope and plausibility tests.

Arrow et al. (1993) proposed that reliable SP estimates must exhibit sensitivity to scope and an *adequate degree of responsiveness* to the magnitude of change being valued. In a later clarification, Arrow et al. (1994) acknowledged that the term *plausible responsiveness* would have better conveyed the behavioral credibility expected of WTP estimates. However, the NOAA panel did not offer formal criteria for evaluating plausibility. Whitehead (2016) and Burrows et al. (2017) suggested using the scope elasticity of WTP (E_{WTP}) as an empirical measure of plausibility. In the presence of diminishing marginal utility, most environmental goods are expected to yield E_{WTP} values below one. Whitehead (2016) conducted simulations under linear and quadratic utility forms, suggesting plausible E_{WTP} typically ranges between 0.2 and 1.0. Although Burrows et al. (2017) proposed a lower plausibility bound of 0.2, they noted that this threshold is somewhat arbitrary and lacks strong theoretical justification. Overall, the literature underscores the importance of context-specific judgment when evaluating the plausibility of scope effects.

Dugstad et al. (2021) further formalized E_{WTP} within the CE framework, emphasizing that assuming constant (unitary) elasticity—common in linear models—can obscure insufficient scope sensitivity.

[2] Countries with at least 70 % of the planet's biodiversity are known as megadiverse countries (CBD, 2016).

Estimating arc-elasticities across discrete attribute levels allows researchers to evaluate whether WTP responds in a behaviorally consistent way, particularly when marginal utility may decline. This approach complements conventional model fit statistics and provides a more nuanced validity test.

In conclusion, this study evaluates whether the network of species interactions passes scope sensitivity and plausibility tests using a CE approach. We compare respondents' WTP for three proxies of biodiversity conservation, including species richness (i.e., number of flora and fauna species), habitat, and network of species interactions. This study makes four novel contributions. First, we value biodiversity conservation in a megadiverse environment. Second, we linked our concept of functionality to the recent definitions of a network of species interactions that is derived from ecological literature (Rajmis et al., 2009; Bakhtiari et al., 2014; Jordano, 2016). Third, we estimate a latent class (LC) model to determine interviewees' heterogeneity based on socioeconomic characteristics. Finally, we evaluate scope sensitivity and plausibility using different econometric models, with particular emphasis on the LC model—both in its standard and attribute non-attendance (ANA) specifications—due to their ability to uncover preference heterogeneity and cognitive processing relevant for biodiversity policy.

Our case study concerns Manu National Park (hereafter Manu), which was established as a UNESCO Natural Heritage Site in 1987 and the Protected Natural Area with the greatest biodiversity in Peru (SERNANP, 2014).

The remainder of this paper is organized as follows. Section 2 presents the materials and methods and the study's design. We present our results in Section 3. Section 4 of the discussion compares our benchmark results with those of previous literature. Section 5 presents the conclusions.

2. Materials and methods

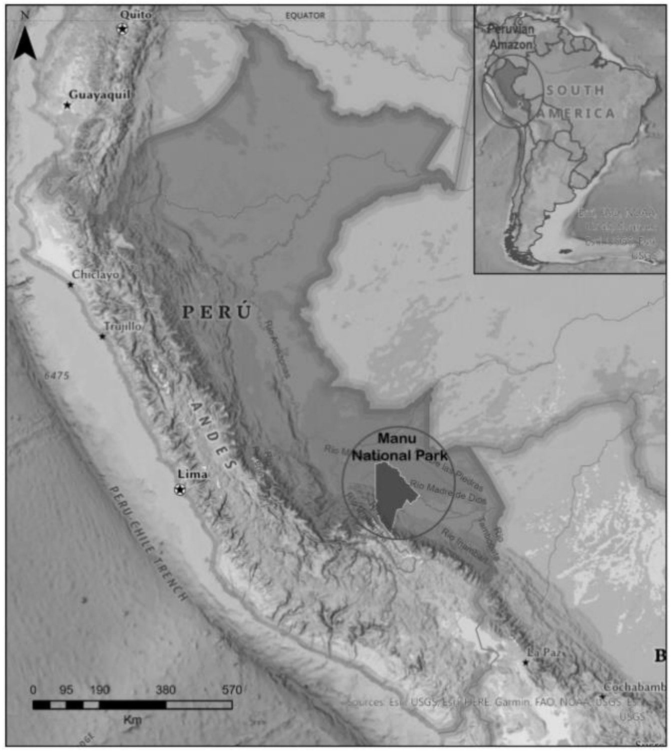
2.1. Study area

Manu has an area of 1.8 million hectares (ha) and is located in southeastern Peru, encompassing part of two regions, Cusco and Madre de Dios (Figure 1). This Protected Natural Area has the greatest biodiversity in Peru, containing a large proportion of the Peruvian Amazon's flora and fauna. A diverse number of wildlife species inhabit Manu, with nearly 160 species of mammals, more than 1000 species of birds (mostly residents), about 140 species of amphibians, 50 species of snakes, 40 species of lizards, 6 species of turtles, 3 species of alligators, and 210 species of fish. Furthermore, the number of insects in the Manu is very high; estimated at about 30 million species, including more than 1300 species of butterflies, 136 species of dragonflies, at least 300 species of ants (more than 40 species were found in a single tree), and more than 650 species of beetles. As for flora, the records indicate at least 162 families, 1191 genera, and 4385 identified species (SERNANP, 2014). Up to 250 varieties of trees have been identified in a single ha[3]. Likewise, in the Manu hosts 10 % of the world's bird species and is the third largest Protected Natural Area in Peru, after Alto Purús National Park and Pacaya Samiria National Reserve, and its tropical forests are among the most undisturbed, only inhabited by indigenous populations in isolation, with a notable diversity of Amazonian ethnic groups (SERNANP, 2014).

Figure 1

Manu location

[3] Some of them include cedar (*cedrela sp.*), cetico (*cecropia sp.*), tornillo (*cedrelinga catenaeformis*), chestnut (*bertholletia excelsa*), lupuna (*chorisia sp.*), and rubber (*hevea brasiliensis*) (SERNANP, 2014).



Source: SERNANP (2014)

2.2. Modeling approach

The CE is a model based on attributes of alternatives, and it has been frequently used in nonmarket valuation since the 1990s (Adamowicz et al., 1994; Boxall et al., 1996; Hanley et al., 1998; Holmes & Adamowicz, 2003). The random utility function is described as follows:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} \tag{1}$$

Where U_{ijt} is the utility of individual i when choosing alternative j in the choice occasion t, V_{ijt} is the deterministic component, and ε_{ijt} is the stochastic term. The respondent selects the alternative that they perceive as providing the maximum benefit.

Different probabilistic models can be used depending on the assumptions concerning the distribution of the error terms. A widely used technique has been the conditional logit (CL) model, which assumes an extreme Value Type I distribution for error terms (McFadden, 1974). In this model, the estimation of the coefficients (β) is conducted using the maximum likelihood method, with the following likelihood function:

$$L(\beta) = \prod_{i=1}^{N}\prod_{t=1}^{T}\frac{e^{\beta' x_{ijt}}}{\sum_{l} e^{\beta' x_{ilt}}} \tag{2}$$

where β is a vector of coefficients associated with individuals' attribute levels and characteristics, and x_{ijt} represents the explanatory variables, i denotes individuals, j and l denote choice sets, and t denotes decision occasions. The term inside the product is the probability of individual i choosing alternative j in decision occasion t.

This study uses CL, mix logit (MXL), and LC models (McFadden & Train, 2000; Crespo-Cebada et al., 2020) to calculate the mean WTP for biodiversity conservation. In the CL model, the preference structure is the same for all respondents, which may suffer from the independence of irrelevant alternatives problem. The MXL model corrects this limitation by estimating a distribution for the coefficients (random parameters) that captures respondents' heterogeneity (Train, 2009).

Following standard practice in stated preference valuation, we specified all non-monetary attribute coefficients as normally distributed. This distribution accommodates both positive and negative preferences and captures attitudinal heterogeneity in biodiversity valuation. In contrast, the price coefficient was held fixed to preserve utility monotonicity and avoid convergence issues or implausibly large WTP estimates—problems commonly observed when using random price terms (Train & Weeks, 2005; Scarpa & Thiene, 2005; Daly et al., 2012). This specification enhances both interpretability and theoretical coherence (Hole, 2007; Hess et al., 2005)..

Assuming this normal distribution, represented as $f(\beta|b, W)$, where β is a vector containing the random coefficients (including those of alternative intercepts), b corresponds to the vector of the means, and W indicates the covariance matrix. Because it is not possible to obtain an analytical expression for the unconditional choice probability, the estimation process uses simulation methods to assess the integral of the probability for given values of *b* and W. In this case, P_{ijt} is the probability of individual *i* choosing alternative *j*, which is defined as follows:

$$P_{ijt} = \int \frac{e^{\beta'_i x_{ijt}}}{\sum_{j=l}^{J} e^{\beta'_i x_{ilt}}} f(\beta|b, W) d\beta \quad (3)$$

In contrast, the LC model captures heterogeneity using a discrete distribution over unobservable endogenous (latent) classes of respondents (Ben-Akiva et al., 1997; Ryan & Amaya-Amaya, 2005). Respondents' preferences are assumed to be homogeneous within each class but can differ across a finite number of classes, which are determined by the data, while membership to a segment depends probabilistically on respondents' observable socioeconomic or attitudinal and behavioral characteristics (Barreiro-Hurlé & Gómez-Limón, 2008; Birol et al., 2009; Lundberg et al., 2020).

The probability of choice varies among classes in this case; that is, the probability of individual *i* in class q choosing alternative *j* is given by (P_{ijt}^{q}), which is determined as follows:

$$P_{ijt}^{q} = \frac{e^{\beta'_q x_{ijt}}}{\sum_{l=1}^{J} e^{\beta'_q x_{ilt}}} \quad (4)$$

q classes exist, in which the probability of choosing each alternative differs from other classes since the coefficients differ (β_q). We also estimate the probability of belonging to each class using a logit model that uses socioeconomic variables as explanatory variables.

We estimate the CL, MXL, and LC models, and end up focusing on the LC specification due to its flexibility in capturing unobserved preference heterogeneity. Unlike MXL, which assumes a continuous distribution of tastes across individuals, the LC model partitions the population into a finite number of classes with distinct, internally homogeneous preference structures. This discrete segmentation is particularly useful in environmental valuation, where respondents may interpret biodiversity attributes differently depending on their ecological awareness, personal values, or familiarity with conservation issues (Boxall & Adamowicz, 2002; Greene & Hensher, 2003; Scarpa & Thiene, 2005).

To determine the appropriate number of classes, we estimated LC models with two to five segments and compared them using the Akaike Information Criterion (AIC) and Bayesian Information Criterion (BIC). To further validate the robustness of preference estimates, we estimated an additional LC model incorporating ANA, following Scarpa et al. (2009) and Lew and Whitehead (2020). This specification allows for the possibility that respondents may have systematically ignored one or more attributes when making their choices. Accounting for ANA helps distinguish between genuine preference stability and lack of attribute salience, thus improving the behavioral credibility and interpretability of class-specific estimates.

Each model also enables the estimation of average marginal WTP for a change in the attribute x_{ijt}. Under the random utility framework, WTP is calculated as the negative ratio between the attribute coefficient and the marginal utility of income (Louviere et al., 2000):

$$WTP = -\beta_x/\beta_{price} \tag{5}$$

where β_x is the estimated coefficient for the attribute and β_{price} is the coefficient for the cost attribute. By comparing respondents' WTP for different levels of biodiversity attribute, we can test whether they are statistically different and determine scope sensitivity (rationality). Economic theory indicates that if an individual is willing to pay something to obtain a certain environmental good (i.e., biodiversity conservation), then the individual should be willing to pay more to obtain a higher quantity and/or quality of that good (Pouta, 2005; Lew & Wallmo, 2011; Ojea & Loureiro, 2011; Hjerpe et al., 2015); otherwise, scope insensitivity prevails, which is a type of embedding effect (Kahneman & Knetsch, 1992; Hanemann, 1994).

Our null hypothesis is that insensitivity to scope exists for each chosen attribute, which can be expressed as $H_0: WTP_{level1} = WTP_{level2} = WTP_{level3}$. The levels refer to the different attribute amounts included in the survey. For instance, *level 1* may represent the lowest level of the attribute, while levels 2 and 3 represent the medium and highest levels. If this hypothesis is rejected and $WTP_{level1} < WTP_{level2} < WTP_{level3}$, we can conclude that respondents' preferences demonstrate scope sensitivity. However, statistical significance alone may not be sufficient.

Whitehead (2016) and Burrows et al. (2017) suggested using the scope elasticity of WTP (E_{WTP}) as an empirical measure of plausibility. This elasticity is defined as:

$$E_{WTP} \equiv \frac{\%\Delta WTP(q,z)}{\%\Delta q} = \left(\frac{\partial WTP(q,z)}{\partial q}\right) \cdot \left(\frac{q}{WTP(q,z)}\right) \tag{6}$$

For a non-marginal improvement in environmental quality and/or quantity, say from q^0 to q^1, where $q^1 > q^0$, with associated change in WTP from WTP^0 to WTP^1 ($WTP^1 \geq WTP^0$), the midpoint formula can be utilized to define a scope arc-elasticity ($\overline{E}_{WTP}$) as follows:

$$\overline{E}_{WTP} \equiv \frac{\%\Delta WTP(q,z)}{\%\Delta q} = \left(\frac{\partial WTP(q,z)}{\partial q}\right) \cdot \left(\frac{\overline{q}}{\overline{WTP}}\right) \tag{7}$$

where $\Delta q = q^1 - q^0 > 0$, $\Delta WTP = WTP^1 - WTP^0 > 0$, and $\overline{q}$ and $\overline{WTP}$ are, respectively, average environmental quality and/or quantity ($\frac{q^0+q^1}{2}$) and average ($\frac{WTP^0+WTP^1}{2}$).

Although E_{WTP} can also be estimated using log-log utility specifications—where the elasticity corresponds to the ratio of coefficients on logged attributes and prices—this approach requires continuous, strictly positive variables. In this study, we rely on dummy-coded categorical levels, and therefore adopt Dugstad et al.'s arc-elasticity method. This allows for flexible modeling of non-linear preferences and clearer interpretation of behavioral responses. Nevertheless, log-log formulations remain a promising avenue for future robustness checks.

This provides a unit-free metric to evaluate whether the WTP changes are economically meaningful and consistent with diminishing marginal utility. Elasticities within the range 0.2 to 1.0 are typically interpreted as plausible under standard microeconomic assumptions. To assess whether elasticity estimates exceeding unity reflect statistically significant deviations from theoretical expectations, we calculated standard errors and 95 % confidence intervals for all E_{WTP} values using the delta method. This approach enables a formal evaluation of whether scope elasticities are statistically distinguishable from one, or whether they fall within plausible ranges due to estimation uncertainty, as suggested in the literature (Whitehead, 2016; Dugstad et al., 2021).

2.3. Choice experiment design

To describe the likely changes in the biodiversity of the Manu, we identified the most important elements of biodiversity in cooperation with ecologists who have investigated the biodiversity and environmental conditions of the Manu.[4] As this study focuses on biodiversity conservation, we present diversity indicators that capture conservation-sensitive factors, including specified values for the number of threatened species or the percentage of network of species interactions maintained as a proxy of the network of species interactions (i.e., thresholds for the loss of species interactions in the network of species interactions). We pretested the list of potential attributes using two focus groups that included the general public and a multidisciplinary team, resulting in a list of five attributes with three or six levels each. The selected attributes in Table 1 represent potential changes in different biodiversity proxies that our focus group respondents identified as meaningful.

Table 1 Attributes and levels of biodiversity of the Manu for the optimal CE design

[4] Most of the information was obtained through personal communication since limited academic research has been published regarding biodiversity in the Manu, and most of it has been in gray literature.

Attribute	Definition	Proposed levels	Name
Threatened flora biodiversity	Number of threatened flora species in the Manu	Low (8) Medium (16) High (24)	flora_8 flora_16 status quo (SQ)
Threatened fauna biodiversity	Number of threatened fauna species in the Manu	Low (8) Medium (16) High (24)	fauna_8 fauna_16 status quo (SQ)
Network of species interactions	Interrelation of agents that generate stability and resilience in Manu ecosystems (in percentage)	High (100 %) Medium (80 %) Low (60 %)	network of species interactions _high network of species interactions _medium status quo (SQ)
Habitat	Annual average deforestation (ha) in the Manu	Low (286) Medium (700) High (1,400)	def_286 def_700 status quo (SQ)
Price	Monthly economic contribution in Peruvian soles (PEN) to avoid the loss of the remaining Manu biodiversity attributes	PEN 32, 24, 16, 12, 8, 0 (SQ)	Price

The first and second attributes are the number of threatened flora and fauna species. For example, cedar and mahogany are two threatened flora species, and spectacled bears and jaguars are two species of threatened fauna. These two attributes have three levels, where the loss of 24 species is the *status quo* (SQ), representing a worsening situation of biodiversity than 16 or 8 threatened species.

The third attribute is the novel biodiversity proxy: the network of species interactions. This construct captures key ecological dynamics such as keystone species loss, interaction richness, and network connectivity—critical for stabilizing food webs and preventing secondary extinctions (Tylianakis et al., 2010; Jordano, 2016). Although empirical thresholds for network stability are difficult to define (Allesina & Tang, 2012), expert input and literature suggest that losing 30–40 % of species interactions may trigger systemic ecological disruptions. Based on this, we defined three levels of interaction retention: 60 %, 80 %, and 100 %, representing degraded, intermediate, and intact network conditions, respectively. These levels were refined through expert consultation and cognitive pre-testing during the survey piloting phase. Importantly, these percentages reflect aggregate interaction retention at the landscape scale of Manu, rather than spatially homogeneous values. While local areas with lower species richness may still exhibit high functionality due to keystone species or interaction hubs, the modeling approach aimed to convey overall network resilience and systemic degradation.

The fourth attribute represents habitat loss from deforestation, referring to the number of ha deforested annually. Increased deforestation implies more habitat loss and biodiversity. We presented three possible levels, including annual losses of 286 ha, 700 ha, and 1400 ha (SQ), to represent the amount of felled trees. The SQ for deforestation was based on historical averages observed in buffer zones of the park.

The final attribute is the contribution, determining respondents' WTP for the four combined biodiversity attributes. Logically, if individuals want to reduce the negative impact of biodiversity loss to zero, they should be willing to contribute more money. The alternative payments ranged from PEN 0 (SQ) to PEN 32 (1 USD = 4.10 PEN, on average, between July and August 2021) per month. Contributions were framed as an additional charge on the household electricity bill, collected monthly over a 12-month period, to fund a real biodiversity conservation initiative in Manu. Six possible levels were presented (PEN 0, 8, 12, 16, 24, 32).

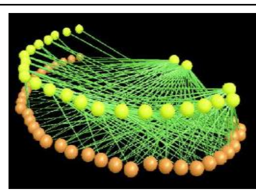
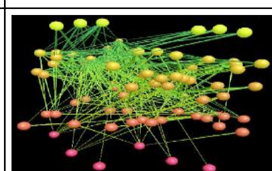
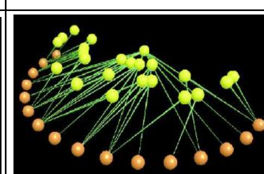
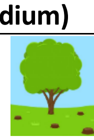
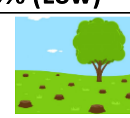
Attributes were explained to respondents using audiovisual material, pointing out that the Universidad Nacional Agraria La Molina from Peru (UNALM) and Servicio Nacional de Áreas Naturales Protegidas por el Estado (SERNANP) are evaluating project implementation to reduce negative impacts on Peruvian biodiversity that includes a real-time surveillance system using drones and increased security personnel and park rangers. Our video included a narrative cheap-talk-script that noted, *"It is VERY IMPORTANT that you take into account your home budget and contributions you would make."*

Although the survey was framed as hypothetical and did not include an explicit consequentiality script or follow-up questions regarding perceived policy relevance, several features were incorporated to enhance perceived consequentiality. First, the initiative was framed as a real and plausible government-supported program. Second, its institutional affiliation with UNALM and SERNANP—two nationally recognized public agencies in biodiversity and conservation—provided credible implementation support. Third, consistent with the recommendations of Carson and Groves (2007), consequentiality was reinforced through institutional framing, realistic policy mechanisms, and contextually grounded scenarios. These design elements aimed to reduce hypothetical bias and increase respondent engagement with the biodiversity conservation choices.

The survey design included 16 choice sets blocked into four questionnaires. An example choice card is presented in Figure 2. We applied a D-optimal design using Ngene software (Puckett & Rose, 2010). The third alternative is a constant SQ, which was added to every choice set, with no variation in the attribute levels for this alternative. Each respondent was presented with four choice sets, each of which included three alternatives.

Figure 2

Choice card example

	OPTION A	OPTION B	OPTION C (STATUS QUO)
Number of threatened flora species	16	8	24
Number of threatened fauna species	16	8	24
Functionality	 80% (Medium)	 100% (High)	 60% (Low)
Annual deforestation (ha)	700 	286 	1,400 
Contribution (PEN)	8	12	0
Choice			

Note: A network of species interaction representations was obtained from Jordano (2016).

2.4. Survey implementation

The survey was conducted online in July and August 2021, in response to restrictions imposed by the COVID-19 pandemic. Participants were recruited through an opt-in online panel, which does not constitute a probability-based sample. We acknowledge this as a methodological limitation and interpret our findings accordingly. Despite this constraint, the survey achieved a relatively high response rate of 65 %, and we collected detailed sociodemographic information to characterize respondents and assess their comparability with the broader urban population of Lima.

The target population comprised residents of Metropolitan Lima, which includes both Lima, the capital of Peru, and the neighboring constitutional province of Callao—together accounting for more than one-third of the national population. Of the 1501 individuals who initiated the survey, 1005 completed all sections, yielding 4020 usable choice observations, as each respondent answered four discrete choice sets. This completion rate (65 %) exceeds that reported in comparable SP studies (e.g., Jacobsen et al., 2011; Jacobsen et al., 2013; Weller & Elsasser, 2018).

Among respondents, 60 % were male, with an average age of 32 years. Approximately 69 % were single, and 83 % reported having completed higher education—primarily women. Only 18 % reported monthly incomes exceeding PEN 6000. A large majority (86 %) indicated they were born in coastal regions. Regarding environmental attitudes and behaviors, 98 % of respondents agreed that biodiversity conservation is important, 36 % had visited a protected natural area within the past five years, and 2 % reported membership in an environmental organization.

Compared to official statistics, our sample overrepresents individuals with higher educational attainment and income levels. In Metropolitan Lima, 42 % of the labor force has completed higher education, and most earn less than PEN 3000 per month (LimaCómoVamos, 2022). Internet access—required to participate in the online survey—is available in only 72 % of households in the region (INEI, 2021a). Thus, our sample is not representative of the full urban population but rather reflects the preferences of a more affluent and environmentally engaged segment. Table 2 provides a comparative summary of sample characteristics and population benchmarks.

Although 27 respondents expressed protest responses—stating that biodiversity conservation is a responsibility of the government and should not be financed by citizens—they were retained in the final sample, as their inclusion did not materially alter the results.

Table 2 Comparison between the sample and the Metropolitan Lima population

Variable	Sample	Metropolitan Lima population
Gender: Male	60 %	55 %
Average age	32 years	38 years (Economically active population)
Marital status: Single	69 %	50 %[1]
Higher education completed	83 %	42 %
Income > PEN 6,000	18 %	66 % earn less than PEN 3000
Internet Access	100% (due to online survey method)	72%
Visited a Protected Natural Area in the last 5 years	36 %	Not available[2]
Member of an environmental group	2 %	Not available[2]

[1] Estimate based on the urban young-adult population structure in Metropolitan Lima (INEI, 2021b)
[2] No official statistics available for visits to protected areas or environmental group membership for Metropolitan Lima

3. Results

3.1. Model results

Table 3 summarizes the key results using the MLX and the LC. Full CL model results are reported in Appendix Table A3 for reference. MXL serves as a benchmark for preference heterogeneity prior to introducing the LC specification. In the MXL model, all biodiversity-related coefficients are statistically significant except for flora_16. Regarding standard deviations (SDs) of the random parameters, most are statistically significant (indicating heterogeneity), except for habitat_286. Importantly, none of the SDs for the network of species interaction levels are significant, suggesting greater agreement among respondents on the importance of this attribute. The cost coefficient remains negative and statistically significant, as expected.

Regarding the sociodemographic interactions, higher income is associated with a lower probability of choosing the SQ. Similarly, older respondents are less likely to choose the SQ, while higher educational attainment also decreases the likelihood of opting for the SQ alternative.

The LC model outperforms both the CL and MXL models in terms of AIC and BIC. We estimated a two-class solution. Although the four-class model yielded the lowest AIC and BIC values (AIC = 10336.96, BIC = 10695.71) (see Appendix Table A1), we selected the two-class model because it offered superior interpretability, particularly because some price coefficients in the three-class and four-class models displayed unexpected positive signs (see Appendix Table A2). The two-class model also produced well-proportioned segments (74.4 % and 25.6 %). Therefore, the two-class specification was considered the most robust, interpretable, and policy-relevant.

Class 1 (LC1), representing the majority of the sample (74.4 %), is characterized by relatively low sensitivity to price and consistently strong preferences for biodiversity conservation attributes. Salary is the only statistically significant sociodemographic predictor of class membership, with higher-income respondents more likely to belong to LC1. In contrast, in class 2 (LC2), the coefficients for flora and fauna attributes are not statistically significant. However, the network of species interactions remains significant across both classes—with a stronger effect in LC1—confirming its salience and robustness across specifications.

We further explored heterogeneity by estimating a LC model with ANA. Compared to the standard LC model, this specification slightly reduced the proportion of respondents assigned to LC2—from 25.6 % to 25.1 %—and marginally improved model fit, as reflected in lower AIC and BIC values. The ANA LC model confirms that respondents in LC2 tend to disregard standard proxies such as flora and fauna richness—reflected in zero or non-significant coefficients—while still assigning significant positive value to the network of species interactions and habitat conservation. This finding underscores that interaction networks are perceived as more salient or understandable even by respondents with lower engagement or cognitive attention to biodiversity detail.

Table 3 Econometric results for mixed logit, and latent class models

VARIABLES	MXL		LC		
	Coefficient	SD	Coefficient Class1	Coefficient Class2	Coefficient Class2 (ANA)
flora_16	0.0469	0.741***	0.0975***	0.1909	0.178
flora_8	0.109*	0.853***	0.1959***	-0.00002	0 (ANA)
fauna_16	0.0845*	0.414***	0.1253***	−0.1058	0 (ANA)
fauna_8	0.384***	0.833***	0.4289***	−0.4291**	-0.387**
network of species interactions _medium	0.395***	−0.132	0.2716***	0.6848***	0.675***
network of species interactions _high	0.551***	−0.0325	0.4051***	0.4701**	0.462**
habitat_700	0.180***	0.468***	0.0922*	0.5052**	0.519**
habitat_286	0.257***	−0.0592	0.0978**	0.4797***	0.492***
Price	−0.0395***	-	−0.0236***	−0.1024***	-0.104***
SQ	1.285**	-	−1.9797***	0.4257*	0.474*
Class probability			74.40 %	25.60 %	25.10 %
SQ*gender	0.1304	0.191		−0.1668	
SQ*age	−0.0591***	0.0901***		−0.0025	
SQ*educ	−0.258**	−0.0422		0.0977	
SQ*salary	−0.366***	0.0274		0.1745***	
Constant				0.6237*	
AIC	11052		10963.89		10957.7
BIC	11254.77		11158.86		11113.67

Note: *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$

3.2.1. Scope sensitivity

Based on the benchmark econometric results, we estimate the marginal WTP for each attribute (Table 4) to examine sensitivity to scope. A statistically significant result in the first row of each model indicates the scope sensitivity between the lowest and medium or highest levels of each category. A statistically significant coefficient in the second row of each category shows the scope sensitivity between the medium and highest levels. The findings reveal sensitivity to scope for all attributes ($p < 0.10$) in the MXL model in the first row with exception of flora_16. Evaluating the scope sensitivity moving from the medium to the highest level (second row) of each attribute, the sensitivity to scope persists for fauna and for the network of species interactions ($p < 0.01$).

The LC model reveals additional insights. Scope sensitivity is evident for the LC1 in almost all cases, while LC2 shows it only for the network of species interactions and habitat ($p < 0.01$). This heterogeneity highlights that scope sensitivity is not uniform across respondents, but class-dependent—an important insight for refining policy targeting and testing the behavioral validity of SP designs in biodiversity valuation.

Table 4 Scope sensitivity tests

			flora_16	**flora_8**	**fauna_16**	**fauna_8**	**network of species interactions _medium**	**network of species interactions _high**	**habitat_700**	**habitat_286**
MXL		**Row 1**	1.19	23.7591**	2.14*	9.74***	10.00***	13.96***	4.56***	6.51***
		Row 2	1.57		7.60***		3.96***		1.95	
LC	**LC1**	**Row 1**	4.0977**	8.2460***	5.177**	17.8826***	11.4802***	17.0320***	3.8805*	4.1196**
		Row 2	4.1483*		12.706***		5.5518**		0.2391	
	LC2	**Row 1**	1.8466	−0.0704	−0.9448	−4.2177**	6.6718***	4.5574***	5.0749***	4.8447***
		Row 2	−1.9170		−3.2728		−2.1144		−0.2302	

Note: *** p < 0.01, ** p < 0.05, * p < 0.1

3.2.2. Plausibility

Table 5 illustrates the attributes and levels presenting statistically significant and economically plausible scope elasticity of willingness to pay (E_{WTP}), defined as values between 0.2 and 1.0 (Whitehead, 2016; Dugstad et al., 2021). This criterion is best fulfilled by the network of species interactions and, to a lesser extent, by fauna attributes. Although the MXL model indicates scope sensitivity for some attributes, WTP does not consistently increase for the highest quality levels—particularly for flora—suggesting limited behavioral plausibility under standard assumptions of diminishing marginal utility.

In contrast, the LC model reveals substantial heterogeneity. LC1, comprising 74.4 % of the sample, demonstrates strong behavioral coherence: respondents are willing to pay significantly more for higher levels of three out of four biodiversity measures (flora, fauna, and species interactions). This suggests a more than proportional response to improvements and underscores the presence of a conservation-minded segment. Notably, network of species interactions consistently satisfy both scope sensitivity and plausibility across all levels in LC1. Meanwhile, LC2 shows limited engagement: only the medium level of species interactions meets the plausibility threshold, while upper levels yield implausibly low or even negative elasticities.

Table 5 Plausibility test

<table>
<tr><th></th><th>flora_16</th><th>flora_8</th><th>fauna_16</th><th>fauna_8</th><th>network of species interactions medium</th><th>network of species interactions high</th><th>habitat_700</th><th>habitat_286</th></tr>
<tr><td rowspan="2">MXL</td><td>N/S</td><td>0.17*</td><td>0.27*</td><td>0.61***</td><td>0.50***</td><td>0.35***</td><td>0.01</td><td>0.01</td></tr>
<tr><td colspan="2">N/S</td><td colspan="2">1.92</td><td colspan="2">1.09**</td><td colspan="2">N/S</td></tr>
<tr><td rowspan="2">LC1</td><td>0.51**</td><td>0.52***</td><td>0.65***</td><td>1.12***</td><td>0.57***</td><td>0.43***</td><td>0.01</td><td>0.004</td></tr>
<tr><td colspan="2">2.01</td><td colspan="2">1.65</td><td colspan="2">1.05**</td><td colspan="2">0.09</td></tr>
<tr><td rowspan="2">LC2</td><td>N/S</td><td>N/S</td><td>N/S</td><td>−0.26</td><td>0.27***</td><td>0.18</td><td>−0.01</td><td>−0.02</td></tr>
<tr><td colspan="2">N/S</td><td colspan="2">N/S</td><td colspan="2">−1.69</td><td colspan="2">−0.06</td></tr>
</table>

Note: *** p < 0.01, ** p < 0.05, * p < 0.1, N/S: Plausibility from nonsignificant WTP

4. Discussion

Our findings inform the ongoing debate on the validity of SP methods for valuing biodiversity conservation in ecologically complex settings. Building on Arrow et al. (1993), we assess whether respondents' choices satisfy scope sensitivity and plausibility criteria (Whitehead, 2016; Dugstad et al., 2021). Although SP techniques like CV and CE are widely used to estimate nonuse values (Boyle et al., 1998; Hanley et al., 2003; Christie et al., 2007; He et al., 2017; Borzykowski et al., 2018; Weller & Elsasser, 2018), meaningful biodiversity representation remains a challenge in megadiverse contexts (Bartkowski et al., 2015; Lavado et al., 2021).

In our CE, the network of species interactions was the only biodiversity proxy consistently aligned with rational and plausible behavior. It generated the highest WTP values and showed strong scope sensitivity, especially in LC1, aligning with prior work on functionality-based indicators (Bakhtiari et al., 2013; 2018). In megadiverse contexts, species or habitat counts may lack behavioral salience, while functionality-based proxies are more cognitively accessible (Ojea & Loureiro, 2009).

The LC model outperformed traditional specifications and revealed marked heterogeneity. Roughly 75 % of respondents (LC1) exhibited scope-consistent and plausible WTP for most biodiversity conservation attributes, while a second group (LC2) displayed weaker or implausible responses. To test robustness, we estimated a LC model with ANA, which slightly reduced LC2's share (to 25.1 %) and improved model fit. Results confirmed that LC2 respondents systematically disregarded standard proxies like flora and fauna richness but still assigned significant value to the network of species interactions. This consistency across classes—even under ANA constraints—suggests that functionality-based proxies hold greater cognitive salience and perceived legitimacy, even among less engaged respondents, echoing concerns in the literature about the limited recognizability of species counts (Boyle et al., 1998; Jacobsen et al., 2008).

Socioeconomic patterns were consistent with previous studies (Barreiro-Hurlé & Gómez-Limón, 2008): LC1 had higher income, education, and concern for biodiversity; LC2 showed lower engagement and scope insensitivity. This segmentation has policy implications in megadiverse countries, where a portion of the population may not perceive additional conservation benefits beyond a threshold.

Despite its strong performance, the network of species interactions may remain abstract for ecologically less literate respondents. While it elicited internally consistent WTP, this does not preclude hypothetical bias.

We note several modeling limitations. Random parameter independence may overlook interdependencies among ecological proxies (Jordano, 2016; Turak et al., 2017). Although we included an ANA specification, more flexible models (Hole, 2011; Lew & Whitehead, 2020) could better distinguish genuine indifference from heuristic-driven simplification.

Lastly, we did not apply formal nested model comparisons (Johnston et al., 2024) to test whether functionality-based proxies improve explanatory power. Future work should examine whether such proxies enhance both model fit and internal validity relative to conventional biodiversity measures.

5. Conclusion

This study tested whether different biodiversity proxies satisfy scope sensitivity and plausibility criteria in an ecologically complex setting. Using a CE in Manu National Park, Peru, we compared four proxies: threatened flora, threatened fauna, habitat, and the network of species interactions. Among these, the network of species interactions—grounded in ecological theory and indicative of ecosystem functionality—elicited the most consistent and plausible responses. It passed internal scope tests and produced elasticity estimates within expected theoretical ranges, even under a LC specification with ANA. This suggests that functional ecological relationships may enhance SP validity, particularly in megadiverse contexts where conventional proxies can overwhelm or disengage respondents.

Our results reveal substantial preference heterogeneity: a majority exhibited coherent WTP patterns, while others showed price sensitivity or inattentiveness to biodiversity attributes. The ANA model confirmed that functionality-based proxies remained salient even for less engaged respondents.

Overall, the study advances methodological and applied knowledge in biodiversity valuation. It demonstrates that novel ecological constructs—such as network of species interactions—can be effectively integrated into SP frameworks, guiding future valuation efforts in biodiversity and cognitively demanding settings. Nonetheless, several limitations remain, including discrete attribute levels, limited public familiarity with ecological functionality, and the non-probabilistic urban sample. Future studies should broaden their geographic and cultural scope to include rural and Indigenous communities, whose ecological relationships and knowledge systems are essential to understanding biodiversity values more comprehensively. Addressing these challenges will be key to improving the validity, inclusivity, and policy relevance of stated preference methods.

Appendix Table A1. LC model selection criteria

Number of classes	Log-Likelihood	AIC	BIC	Smallest class share (%)
2	-5458.637	10963.89	11158.86	25.6
3	-5314.426	10696.85	10962.01	14.8
4	-5122.48	10336.96	10695.71	4.1
5	Convergence not achieved			

Appendix Table A2. Estimated parameters by class in the LC model

Variable	Class1	Class2	Class3
flora_16	0.16	0.19	23.025
flora_8	0.133	-0.232	-33.9
fauna_16	0.358	-0.297	-31.945
fauna_8	0.606	0.591	-84.251
network of species interactions _medium	0.296	0.721	-111.289
network of species interactions _high	0.535	0.842	-85.741
habitat_700	-0.046	-0.678	207.779
habitat_286	-0.084	-0.558	204.488
price	0.003	-0.163	-3.63
Sq	-1.422	0.002	-205.656
Class probability	**56.20 %**	**29 %**	**14.80 %**

Variable	Class1	Class2	Class3	Class4
flora_16	4.899	0.015	-1.294	0.11
flora_8	-26.225	0.247	1.062	-0.342
fauna_16	-1.346	0.103	2.213	0.026
fauna_8	-31.156	0.671	2.939	-0.029
network of species interactions _medium	-2.928	0.08	2.579	0.434
network of species interactions _high	-3.542	0.358	2.464	0.244
habitat_700	30.293	-0.458	2.903	0.288
habitat_286	31.565	-0.192	2.015	0.264
price	0.084	0.006	-0.307	-0.081
Sq	-28.571	-1.341	-3.813	0.826
Class probability	**9.5 %**	**40.5 %**	**27.8 %**	**22.2 %**

Appendix Table A3. CL model and sensitivity to scope (WTP)

VARIABLES	CL Coefficient	Sensitivity to scope	
flora_16	0.0795*	2.49*	
	(0.0418)		1.37
flora_8	0.1232***	3.87***	
	(0.0456)		
fauna_16	0.1097**	3.44**	
	(0.0425)		6.93***
fauna_8	0.3305***	10.37***	
	(0.0429)		
network of species interactions _medium	0.3420***	10.73***	
	(0.0421)		1.27
network of species interactions _high	0.3825***	12.00***	
	(0.0461)		
habitat_700	0.1088**	3.41**	
	(0.0430)		1.13
habitat_286	0.1448***	4.54***	
	(0.0427)		
Price	−0.03187***		
	(0.0017)		
SQ	−0.6109		
	(0.0587)		
AIC	12265.12		
BIC	12343.10		
Observations	6004		

Note: Standard errors are in parentheses. *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$

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