

Article

Economic Valuation and Community-Based Management: The Whale Shark Wildlife Tourism in La Paz Bay, Mexico

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Abstract

The whale shark aggregation area in La Paz has given rise to vital wildlife tourism activity since the mid-1990s, which has been consolidated during the first decade of the 21st century. La Paz Bay is one of the three sites in Mexico where whale shark wildlife tourism is practiced. Biological and ecological research of whale sharks in the La Paz Bay is extensive. However, there is a considerable lack of knowledge about the socioeconomic implications of this activity. Understanding the recreational values of whale shark area users is fundamental to formulating an effective management policy. Using the individual travel cost method, we estimate the recreational value of whale sharks in La Paz Bay—the estimated individual's willingness-to-pay ranges from 8 to 27 USD per trip. The recreational value of whale shark wildlife tourism ranges from 304,600 to 1,028,025 USD/season. The recreational value per whale shark ranges from 2361 to 14,083 USD. These results serve as a baseline for implementing economic and environmental policies and/or instruments to collect financial resources, strengthening actions oriented towards site and species conservation. Community-based management options, limitations, and opportunities are also discussed.

Keywords: marine endangered species; *Rhincodon typus*; non-market valuation; gulf of California; marine and coastal tourism; willingness-to-pay

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1. Introduction

1.1. Wildlife Tourism

Wildlife tourism is defined as tourism based on encounters with non-domesticated animals (Higginbottom, 2004). These encounters can occur in either the animals' natural environment or in captivity and are also known as 'non-consumptive' activities, such as viewing or watching, photography, and feeding, as well as hunting and recreational fishing. Wildlife tourism is typically practiced in protected areas (terrestrial or marine), with

the main challenges in emerging economies being sustainable finance and insufficient funding for these areas (Rodger et al., 2010).

Khan (2019) defines sustainable finance as the integration of environmental, social, and governance (ESG) criteria into financial and investment decision-making, with the explicit goal of directing capital flows toward activities that support long-term ecological balance and societal well-being. Besides the financial metric of risk–return, it incorporates a third dimension: positive impact on natural and human systems. Therefore, sustainable finance is the mechanism through which we align the financial system with the biophysical realities of our planet (G. L. Clark et al., 2015). On the other hand, constant financial support is not merely important but the critical determinant of whether a marine protected area (MPA) achieves its ecological and socioeconomic objectives; underinvestment directly correlates with biodiversity loss (Balmford et al., 2004). According to Gill et al. (2017), ongoing financial resources are required for core management activities: enforcement, monitoring, habitat restoration, community engagement, adequate and consistent staffing, and budget, which result in positive ecological outcomes.

Globally, many marine wildlife-watching sites and species opportunities have evolved into a significant and growing industry in some developed and emerging economies. Valentine and Curnock (2001) identified 70 target species for practicing wildlife tourism, including the whale shark. Higginbottom (2004) categorizes marine wildlife watching into three distinct elements: shore-based observations, boat-based observations, and in-water activities, all of which are widely available. The latter includes swimming and diving focused on marine wildlife, including some specialized opportunities (i.e., swimming with sharks, whale sharks, dolphins, and whales).

Whale shark marine tourism is a growing industry and activity in both developed and emerging countries. Since 1986, there has been a considerable increase in recreational diving and boating activity worldwide, with whale shark (WS) aggregations reported in various locations (Stevens, 2007). Often, predictable aggregations foster diving and boating activities with WSs, which are associated with singular productivity events. Locations where aggregations occur include Belize, Central America (Heyman et al., 2001; Graham & Roberts, 2007), Ningaloo Reef, Australia (Taylor, 2007; Norman & Stevens, 2007; Taylor, 1996), the Sea of Cortez and Holbox, Mexico (Whitehead et al., 2020; A. Ziegler et al., 2015; Eckert and Stewart, 2001; Nelson, 2004), the Seychelles, Africa (Rowat and Gore, 2007), KwaZulu-Natal, South Africa (Beckley et al., 1997), and Veraval, India (Vivekanandan and Zala, 1994). Stevens (2007) notes that rising demand and higher prices for WS wildlife tourism have led to increased interest in marine wildlife tourism and in the conservation of the species on which it depends.

Wildlife tourism has economic benefits, including the monetary value of the expenditures wildlife tourists make for travel, accommodation, food, souvenirs, and other related expenses during their visit. Economists have used total economic value (TEV) as a broader concept than these expenses. There are two widely accepted definitions of TEV. Turner et al. (1993) define TEV as a multi-attribute environmental asset that provides both use and non-use values to individuals. D. W. Pearce and Turner (1990) define TEV as the sum of the actual use, option, and existence values. Direct use by tourists is one of the economic values generated by nature tourism destinations (Higginbottom, 2004; Wells, 1998).

These positive impacts from direct uses are only a partial reflection of the total economic value of wildlife tourism, as significant non-use values, such as existence, intrinsic, heritage, and option values (Tisdell, 2002; Emerton, 2001), also need to be considered. Even though the difference between the amount individuals would be willing to pay and the amount they pay constitutes foregone income to the destination (consumer surplus), decision-makers are less interested in these non-use values because of the difficulty of capturing or utilizing them in practice (Wells, 1998).

The Economics of Ecosystems and Biodiversity (TEEB, 2012) argues that a better understanding of marine and coastal ecosystems, biodiversity, and their contributions to societal well-being is required. Demonstrating the economic value of ocean and coastal biomes could reveal new economic opportunities. It could also play a role in policy and management schemes that enhance sustainable development for marine resources and ecosystems. Marine environments provide nearly two-thirds of global ecosystem services (Cooley et al., 2022), yet humans know relatively little about them. Marine ecosystems suffer due to knowledge gaps and governance deficits. Conserving ocean and coastal ecosystems requires managing human activities within them. To address the most critical challenges affecting ocean and coastal environments, the blue economy approach must be integrated into decision-making to enhance ecosystem services and resources and promote sustainable use.

At the beginning of the century, as well as more recently, some research has shown that wildlife tourism valuation has been considered for marine and coastal tourism management (Appendix A). However, to date, no study has estimated the direct use recreational value of wildlife tourism with WSs or the value of individual whale sharks in La Paz Bay, Mexico. Therefore, the research seeks to estimate the recreational value of whale shark wildlife tourism using the travel cost method and to link it to environmental economics policy instruments and community-based management strategies.

1.2. Whale Shark Wildlife Tourism in La Paz Bay, Mexico

The Gulf of California (GOC) is where many WS sightings occur (Wolfson, 1987), especially off the southern Baja peninsula, from Cabo San Lucas to La Paz Bay. Peak WS sightings occur from May to June and from September to November, due to the high abundance of feeding species (Ketchum, 2003).

La Paz Bay, located in Baja California Sur, Mexico, is the biggest coastal water cove in the GOC, with a 2600 Km² area, 450 m deep in its deepest zone. It is the most important bay in the state due to its dynamic biological and fishery activity, which is important to net both primary and secondary production (Reyes-Salinas et al., 2003). La Paz Bay is bounded to the North by Isla San José, to the South by Ensenada de La Paz, and to the East by the Espiritu Santo and La Partida Islands, making the bay a critical refuge area for many marine species, including whale sharks (SEMARNAT, 2021). The southern part of the bay features a protruding sandbar, attached at one end to the mainland and extending approximately 12 km parallel to the bay's mouth, known as El Mogote.

These primary production areas constitute an important feeding zone for filter-feeding marine fauna such as sardines, manta rays, and WSs (De Silva-Dávila & Palomares-García, 1998; Ketchum et al., 2013). Because of the constant presence of WSs, a growing wildlife tourism activity has developed in La Paz, mainly in El Mogote, which is the leading area for WS tourism due to its highly productive coastal zone and its recurrent, seasonally aggregating site for WSs. These feeding and aggregation zones have led to the declaration of several sites as protected areas in different countries. Some key examples of these are Ningaloo Reef, Australia; Baa Atoll and Hanifaru Bay, Maldives; and areas in Bazaruto Archipelago, Mozambique (S. J. Pearce et al., 2021; Thomson et al., 2017; Mulder, 2016; Schleimer et al., 2015; Moriel-Robles, 2009; Mau, 2008).

Diver encounters with WSs in La Paz Bay were relatively rare in the 80s (E. Clark & Nelson, 1997), but there was a marked increase in the number of specialized ecotourism operators in the 90s (Johnson et al., 2019; Hacothen-Domené et al., 2006). Since then, WS wildlife tourism has become an important economic activity in the GOC region and represents an important form of natural capital with high potential to produce economic value for La Paz Bay and Bahía de Los Angeles (Cárdenas-Torres et al., 2005; Rodríguez-Dowdell et al., 2007), where swimming and observation with WSs are conducted,

primarily due to regular and predictable aggregations on a seven-month basis annually (Petatán-Ramírez et al., 2020).

In Mexico, whale shark aggregations frequently occur in Bahía de Los Ángeles, Baja California; La Paz Bay, Baja California Sur; and Banco Gordo, Baja California Sur (Ramírez-Macías, 2005). These aggregations have become an essential seasonal component that supports an important tourist industry. According to Ramírez-Macías et al. (2026), in La Paz City, chartered dive boats conducting WS wildlife tourism have shown rapid growth since the late 90s, increasing from 26 authorized boats in 2006 to 109 in 2013. Tour operators average 56 WS sightings per season. Local authorities are now concerned that more permits will worsen WS issues. Whitehead et al. (2020) noted 1662 WS sightings at La Paz Bay from 2015 to 2018, ranging from 73 to 129.

There have been recent attempts to measure the economic benefits of nature-based tourism in Baja California Sur. One estimated the direct expenditure from nature-based tourism in La Paz at USD 8–18 million (Cisneros-Montemayor et al., 2020). A similar study for gray whale watching in El Vizcaino estimates tourist direct expenditure at about USD 0.70 million (Mayer et al., 2018). There are two studies on elasmobranchs: a study on shark diving estimates the direct expenditure of tourists at USD 12.4 million (Cisneros-Montemayor et al., 2019; Cisneros-Montemayor et al., 2013), and a study on Munk's pigmy devil ray that estimates direct expenditure at USD 0.88 million (Brugès, 2024). Nevertheless, as Pendleton and Rooke (2007) highlight, in the case of wildlife tourism, or recreational activities in general, it is more accurate to use direct use values to measure consumer surplus because they can be measured using revealed-preference methods, like the travel cost method. In this regard, there are no studies that use revealed preference methods to value the recreational activity associated with nature tourism involving whale sharks in La Paz; this research seeks to fill this gap.

It must be highlighted that marine wildlife tourism with WS in La Paz Bay has rapidly increased since the 1990s, raising concerns about its effects on this endangered species and capacity-building (Whitehead et al., 2019; Ramírez-Macías et al., 2026). In this context, capacity building empowers people and organizations to tackle environmental issues by improving skills, knowledge, resources, and governance for sustainability (Visseren-Hamakers et al., 2021; Burch et al., 2019).

One of the recurring problems in the WS area has been injuries caused by boats, primarily affecting the dorsal, caudal, and back fins. Ramírez-Macías and Saad (2016) reported that 54% of WSs had fresh injuries; 60% had abrasions, 30% had both abrasions and lacerations, and 9% had only lacerations. Womersley et al. (2022) indicate that La Paz Bay is considered a high-collision risk site for WSs. These issues are primarily driven by high-maritime-traffic areas and the lack of maritime signaling and surveillance, making them a significant challenge to address, in addition to those arising from growing tourist demand. However, there is no baseline to establish economic forfeits for natural resource damage or penalties and fines for those who cause WS injuries.

Pressures from uncontrolled tourism development, growth, and performance have prompted authorities to establish a protected area and zoning for WS wildlife tourism (Figure 1). Some site-specific regulations were also established to regulate the over-growing industry, such as (i) conditions for obtaining permits, (ii) indications for conducting the activity, (iii) prohibitions inside the area, (iv) emergency and contingency measures, (v) boarding and disembarking sites, (vi) number of authorized boats, (vii) boat operation maneuvers during the activity, and (viii) priority criteria. Lastly, authorities define the whale shark wildlife tourism season as running from September to April annually (SEMARNAT, 2021). However, Thomson et al. (2017) noted that the WS watching season could vary due to factors such as food availability, environmental conditions (sea surface

temperature, ocean currents, and upwelling), migratory behavior, and anthropogenic impacts (tourism pressure, boat traffic, and fishing).

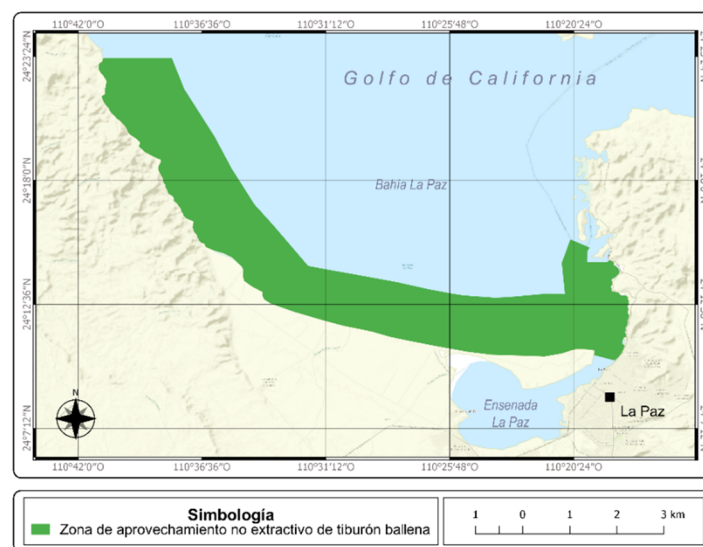


Figure 1. Non-consumptive whale shark area (in green). La Paz Bay, Mexico. Source: Adapted from SEMARNAT (2021).

Nevertheless, to ensure that recreational ecosystem services provided by this species endure, an effective management policy is necessary to prevent damage to WS and its habitat. This management strategy should inform policy decision-makers on how to increase monetary collection and reinforce financial resource allocation for conserving and protecting whale sharks and their habitat, as an economic baseline for potential damages. This policy should be strengthened by understanding the area, the species, user characteristics, and their recreational values.

1.3. Community-Based Management

Bennett et al. (2021) define community-based management (CBM) as the governance and stewardship of natural resources or biodiversity by, for, and with local communities. It emphasizes the coexistence of people and nature, in contrast to protectionist approaches that segregate them. CBM also emphasizes recognizing local rights, knowledge systems, and cultural and ancient practices, as well as identifying leadership in conservation efforts. It seeks to ensure that conservation initiatives align with the area's socio-cultural context and provide tangible benefits to local populations while supporting local livelihoods (Berkes, 2007). However, resource management requires different strategies depending on whether the resources are terrestrial or marine. Managing marine resources is more difficult than managing terrestrial resources, as some of these resources are migratory and move above and below the water column, making their position unpredictable (Maxwell et al., 2015).

Therefore, CBM in marine conservation refers to the involvement of local communities and their knowledge in governance, planning, and management to achieve effective conservation outcomes for marine and coastal resources and protected areas (Sustainable Travel Tech, 2024). Marine CBM is a part of incentive-based conservation (IBC) strategies, which empower and involve communities in the stewardship planning, development, and conservation management for sustainable use of marine resources, looking to promote conservation while ensuring local stakeholders' economic benefits and social welfare from the managed resources (J. Ziegler & Dearden, 2021; Bennett et al., 2021;).

However, there has been a broad debate about the key principles of CBM (Child, 1996; Armitage, 2005; Cox et al., 2010; Dressler et al., 2010; Brooks et al., 2013). In this regard, Gruber (2010) provides an interesting summary of 12 CBM principles. These principles are (i) public participation and mobilization, referring to engaging diverse stakeholders in decision-making and implementation, (ii) social capital and collaborative partnerships, which is building trust, networks, and partnerships to leverage resources and foster innovation, (iii) resources and equity, meaning ensuring fair distribution of benefits and addressing local economic and social needs, (iv) communication and information dissemination, related to transparent and accessible communication to foster trust and knowledge sharing, (v) research and information development, by using integrated scientific, social, and local knowledge for informed decision-making, (vi) devolution and empowerment, which is transferring authority to local communities for better decision-making and commitment, (vii) public trust and legitimacy, by building trust through transparency and participatory approaches, (viii) establishing monitoring, feedback, and accountability systems for evaluation and learning, (ix) adaptive leadership and co-management, which is promoting resilience and collaboration through dynamic leadership and management, (x) participatory decision making, by facilitating inclusive problem-solving and decision-making processes, (xi) optimal environment, meaning that preconditions or early conditions should be assessed to ensure favorable social and resource conditions for success, and (xii) conflict resolution and cooperation, by addressing inherent conflicts and fostering cooperation among stakeholders. These elements collectively aim to empower communities to protect marine resources sustainably while improving their livelihoods.

However, Berkes (2007) argues that CBM is not a “silver bullet” or “one-size-fits-all” solution, but rather a strategy that requires adaptation to the context, adaptive governance, and recognition of the challenges to nature conservation. Linking CBM with economic valuation is not easy, but when done, it creates a powerful alliance in which economic data empowers local governance and validates traditional stewardship (Bennett et al., 2018). Berkes (2021) indicates that on the one hand, CBM often relies on advocating for the resource’s importance; on the other hand, TCM generates socioeconomic and demographic data to provide a monetary estimate of a site’s recreational or cultural value, helping negotiation with external actors and justifying conservation efforts. Also, it helps us to understand how site users value the resource in relation to its precedence, meaning the value visitors receive exceeds what they actually pay. This reveals the potential to generate revenue without drastically reducing visitation. Finally, it assists in setting fees at a level that generates revenue for the community management fund without discouraging visitation (especially by locals) and in prioritizing conservation investments, fostering participatory monitoring and ownership.

2. Materials and Methods

2.1. Sampling

On-site questionnaire surveys were conducted at La Paz City by the research team (lead researcher, supervisor researcher, and two trained interviewers) from December 8th to January 27th. Three hundred and thirty-four surveys were administered to visitors who participated in WS wildlife tourism activities. Surveys were conducted at the local harbor applied to boat passengers. One person was chosen randomly from each boat group. Interviewers answered every interviewee’s questions regarding the questionnaire when the survey was applied.

The sample was estimated by applying unrestricted random sampling without repetition for a known population:

$$N = (Npq)/pq + (i^2(N - 1)) \quad (1)$$

where N : population, p : success probability (0.5), q : $1-p$, and i : estimation error (5.2%). As Scheaffer et al. (2012) and Bethlehem (2008) indicate, the calculated sample is intended to be representative at 95% confidence because this probability sampling method allows the quantification of random error and ensures the sample is unbiased and representative of the population. Therefore, it could be used with visitors who practice wildlife tourism with whale sharks. Information on visitors who engaged in whale shark wildlife tourism at the site during the study period of 2023–2024 was provided by the Natural Resources Administration Agency (SEMARNAT, its Spanish acronym). The agency registered 38,075 visitors that season. The survey response rate was 94.8%. This slight shortfall in the sample was due to some interviewees filling in the survey incompletely, for reasons such as tiredness after swimming with the WSs, a hurry to catch transport, or asking another person to complete it. Nevertheless, Lohr (2022) indicates that a slight shortfall, like 1.6%, is operationally common. Representativity is preserved if the sampling design was probabilistic, as in this case, and the non-response is random; therefore, a small shortfall (low non-response rate) does not automatically imply non-representativeness (Groves & Peytcheva, 2008).

2.2. Travel Cost Method

The travel cost method (TC) was proposed by Hotelling in year 1947 to the National Park Service to establish access fees to protected areas. TC estimates demand for an environmental good or service as the number of visits, given varying travel costs. The first TC studies were conducted in the early '70s (Haab & McConnell, 2002).

TC is based upon seven assumptions (Haab & McConnell, 2002): (1) the number of trips/travels and environmental site quality are complementary with utility function, (2) every individual perceives and responds to travel cost variations in a way that reacts to changes in on-site access fees, (3) only one site is visited, (4) staying time is fixed and exogenous, (5) there are no site substitutes, (6) wage rate represents time opportunity cost, and (7) the individual does not perceive utility or disutility during travel time or working time.

From an economic perspective, aquatic ecosystems (lakes, rivers, estuaries, bays, and seas) and the recreational services they provide are considered public goods. As a result, some of their important environmental attributes and characteristics have no market value. The latter are essential for determining the economic value of natural resources that offer recreational alternatives not assigned by the market system (TEEB, 2012).

As Vásquez-Lavín et al. (2007) note, TC assumes that each individual visiting a recreational site is associated with an implicit transaction linking travel cost to the value (or price) that visitors should pay to access the site. TC has been modeled from two perspectives. One is that a demand function is estimated by associating trips with travel costs, which vary with the traveled distance. The site's economic value is represented by the area under the compensated demand curve and is calculated for all individuals who have visited the site. In the second approach, individuals decide whether they want to visit a site for recreational purposes. If affirmative, models that use this decision scheme are linked to a discrete choice or random utility model (RUM). These models emphasized the choice problem of selecting a site for a predetermined trip; that is, after choosing a number of trips to carry out the recreational activity, the site selection must be modeled.

The same authors indicated that the individual's utility-maximization model could be divided into two stages. First, the site to visit is selected each time. Therefore, there is an interest in how trips are distributed among different sites. In this case, welfare estimates are calculated by using the parameters of an indirect utility function.

The general TC model formalizes individual or group behavior regarding the number of trips to a specific site. These behavioral models are based on a common hypothesis of utility maximization, restricted to the budget constraint (Hueth & Strong, 1984). Assuming that there is a single site available and that all visits are equally distributed, it is feasible to adapt the problem as follows:

$$\begin{aligned} \text{Max } U(x, z), \text{ s.a. } \quad m &= d + wt_w = z + (c_1 + c_2) \\ T &= t_w + (t_1 + t_2)x \end{aligned} \quad (2)$$

where x is the number of trips/travels, z is expenditure on a good which does not involve the time constraint (or Hicksian good), m is individual total income, d is the available income not related to labor journey, w is the wage rate, t_w is the work time, c_1 is travel cost, c_2 are the expenditures on the site (discretionary cost), T is total available time, t_1 is travel time, and t_2 is site staying time.

Assuming individuals could choose discretionally working time (in hours), and assuming that the opportunity cost of travel time is related to w , it is possible to isolate tw as

$$tw = T - (t_1 + t_2)x \quad (3)$$

Replacing (3) in the budget constraint m gives the following:

$$\begin{aligned} m &= d + w[T - (t_1 + t_2)x] = z + (c_1 + c_2)x \\ d + wT &= w(t_1 + t_2)x + z + (c_1 + c_2)x \\ d + wT &= z + x[(c_1 + wt_1) + (c_2 + wt_2)] \end{aligned} \quad (4)$$

From equation (4), it follows that wT is the income if the individual allocates all time to work, $c_1 + wt_1$ is equivalent to the travel cost, and $c_2 + wt_2$ represents the staying cost. Then, Equation (4) could be rewritten as

$$m^* = z + p_x x \quad (5)$$

where $m^* = d + wT$ and $p_x = (c_1 + wt_1) + (c_2 + wt_2)$. Then, the utility maximization problem transforms into

$$\text{Max } U(x, z), \text{ s.a. } m^* = z + p_x x \quad (6)$$

The idea is to estimate $x = (p_x, m^*)$ and $z = z(p_x, m^*)$, where p_x is the price of a trip/travel and p_z is the price of good z ; both equations are the demand functions obtained by solving the primal problem, highlighting the linearity of the budget constraint. Once the primal problem of TC is solved and its assumptions are taken into account, recreational demand estimation becomes possible.

2.3. Recreational Demand Estimation

At the beginning of the 80s, two demand estimation approaches for TC were identified: demand by origin zones and individual demand estimation. Hellerstein (1995) and Clawson and Knetsch (1996) indicate that due to aggregation bias and the lack of micro-data, economists abandoned the demand zone method. Therefore, the individual approach has been frequently used by environmental economists, yielding better parameter estimates for the regressors and more efficient estimates of the demand function for recreational services. The general form of individual TC is

$$X_{ij} = f(C_{ij}, Z_{ij}, e_{ij}) \quad (7)$$

where X_{ij} is the number of visits to site i by individual j in one year, C_{ij} is the individual's travel cost to the site, Z_{ij} is a vector of socioeconomics and other environmental variables associated with the site, and e_{ij} is the stochastic term.

Discrete distributions are used to estimate most TC models in which the dependent variable is the number of trips or travel. These variables are discrete and non-negative, as people generally make only a few trips to the recreational site, typically one or two (Haab & McConnell, 2002). Because of this approach, TC studies commonly use discrete density functions, such as the Poisson distribution, defined by the parameter λ , which represents both the mean (expected number of trips) and the variance. If $\lambda_i > 0$, the expected number of trips (x_i) is positive. However, TC has some issues that must be taken into account, like investment in durable equipment, multi-site/purpose recreational trips, substitutes sites, opportunity cost, lodging, subsistence expenditures and other costs involved, site classification, distance, quality of recreational area and congestion, and travel duration (Czajkowski et al., 2019; Leh et al., 2018; Clawson & Knetsch, 1996; Randall, 1994; Turner et al., 1993; Smith & Kaoru, 1990).

A Poisson regression model is appropriate because of the count data characteristics of WS wildlife tourism visits. The probability of a count is determined by a Poisson distribution (A. C. Cameron & Trivedi, 2007). Count models based on the Poisson distribution avoid regression bias when the dependent variable can only take nonnegative values (Shaw, 1998). One of the main assumptions of the Poisson distribution is that there may be zeros in the observations. However, Haab and McConnell (2002) mention that the estimator in the Poisson model can be biased and inconsistent in the presence of over-dispersion (α), which is considered a type of heteroscedasticity and is defined as the excess conditional variance over the corresponding conditional mean of the dependent variable (when the variance–mean ratio is greater than 1). Under conditions of this nature, it is advisable to use a negative binomial distribution, considered an extension of a Poisson distribution (Hilbe, 2011; A. C. Cameron & Trivedi, 2007).

The net benefits for recreational site visitors can be measured as the consumer surplus (CS). It is the most widely used measure for estimating visitor net benefits (Zhang et al., 2015). Once variable parameters are estimated using count models for the TC, Haab and McConnell (2002) suggest calculating the CS or willingness-to-pay (WTP) by the following formula.

$$WTP = \bar{x} / -\beta_{tc} \quad (8)$$

where $\bar{x}$ is the average number of trips to the site and β_{tc} is the travel cost coefficient. Then, to calculate the WS wildlife tourism recreational value (WS-REV), the estimated WTP must be multiplied by the total number of visitors registered for the studied season for each estimated model. The research adds the economic value per whale shark, estimated by dividing the WS-REV by the high and low monthly WS sightings reported by Whitehead et al. (2020), and the average economic value per WS was also estimated.

Thus, socially, quantifying this considerable economic value strengthens the argument for conserving whale sharks. It transforms them from abstract creatures into recognized economic assets, fostering community pride and stewardship (J. A. Ziegler et al., 2021). On the other hand, economically, the value captured by the method represents a direct financial inflow to the region. It highlights that live sharks generate far more sustained income than a one-time sale if harvested. This provides a powerful economic rationale for governments and businesses to invest in sustainable management instead of short-term exploitation (Cagua et al., 2014).

2.4. Proposed Recreational Demand Models

Following Randall's (1994) recommendation, we use travel price (tp), which each individual paid for the WS wildlife tourism excursion, instead of travel cost. The analysis excludes equipment investment, because appropriate equipment is provided for tour operators. Recreational areas differ due to their cultural, environmental, and historical

characteristics. Because of this, we decided not to include substitute sites; besides, there are no other sites nearby to watch WS in Baja California Sur; La Paz Bay is the only aggregation area for WS in southern Baja. Opportunity and travel duration were included in this research through travel-time costs and other costs, such as on-site meal expenditures. Maximum likelihood estimation was used to estimate the coefficients in the Poisson models.

Four different recreational demand models are proposed to estimate, each with a different type of travel price as the independent variable. First, travel price (*TP*) is the price paid for the WS excursion. The second regression is referred to as *TPOME*, as the travel cost analysis literature lacks consensus on which additional costs, such as accommodation or food, should be included (Rolfe & Prayaga, 2007). Beal (1995) reports that most respondents to travel cost surveys considered fuel, food, and accommodation costs relevant to their recreational trips. Based on this, we decided to include on-site meal expenditures (*OME*) in our analysis, so *TPOME* is defined as

$$TPOME = tp + OME; \text{ where } OME: \text{ On-site meal expenditures} \quad (9)$$

Third, following arguments on travel cost method issues, we decide to include opportunity cost, which is defined as

$$TPTTC = tp + TTC; \text{ where } TTC: \text{ Travel-time cost} \quad (10)$$

Travel-time cost (*TTC*) was estimated using the following expression proposed by Hernández-Trejo et al. (2009); this approach allows for consideration of opportunity cost in welfare estimation.

$$TTC_i = tp_i + (OPC_i \cdot TT_i) \quad (11)$$

where tp_i is the travel price of the excursion for WS wildlife tourism paid by the individual, OPC_i is the individual's opportunity cost, and TT_i is the effective travel time the individual spends from origin to La Paz. *OPC* was estimated by

$$OPC_i = (\eta Y_i) / TWT_i \quad (12)$$

Where Y_i is an individual's annual income, TWT_i is the total worked time a year, and η is a coefficient to convert working time into leisure time. In our analysis, travel time cost was previously defined. However, there is still debate about how to estimate the time cost. One method commonly used is to use a fixed percentage of the respondent's income (Larson & Lew, 2014). Some researchers have suggested using a percentage between 25% and 100% (Hanley et al., 2001); in our analysis, we use 33%. This proportion follows common practice in the recreational valuation literature (e.g., Hanley et al., 2001), though sensitivity to this parameter is acknowledged. An average of 50 weeks and 40 working hours a week was assumed to calculate *TWT*.

Fourth, a model that includes travel price, travel-time cost, and on-site meals (*TPTTCOME*) is defined by

$$TPTTCOME = TPTTC + OME \quad (13)$$

3. Results

3.1. Data

The average age was 34 years old. It took an average of 7.5 h to arrive at the La Paz destination, with six people on board the boat on which the wildlife tourism activity with WSs was performed. Four WSs were spotted before jumping into the water, and visitors declared having done the activity twice in the last five years, staying for six days. The average values (in USD) for monetary variables are as follows: annual income, USD

40,080; excursion with WSs, USD 74; airfare expenses, USD 360; meals, USD 225; fuel, USD 134; car rent, USD 115; lodging, USD 351; total travel cost, USD 540. It must be highlighted that not all visitors incur these exact costs when they visit La Paz. For instance, the annual income of domestic visitors is USD 10,480, and that of foreign visitors is USD 108,748; thus, travel expenses would differ between the two groups.

The sample consists of 49% females, with a majority of males. Visitors are categorized by origin as follows: 74% are domestic, 14% are from Europe, 9% are from North America (including the United States and Canada), and 3% are from other countries. Almost 74% of the sample reported visiting the WS area for the first time. Visitors' opinions about the main reason for conserving the WS area are disaggregated as follows: 87% cite marine biodiversity as the main reason for site conservation, 11% think the area should be conserved for recreational activities, and 2% consider its conservation important for economic reasons. Regarding site conservation status, 54% of the sample considers it deficient, 32% grade it as poor, and only 16% consider the WS area to have a regular conservation status. The guide's performance during the WS activity was graded by visitors as follows: 71% ranked it very good, and 29% ranked it good. A high percentage of visitors, 99%, were informed about the regulations for performing wildlife tourism activities. A high proportion of interviewees (92%) considered it important to inform people that the WS area is a natural sanctuary for WS conservation. Last, 88% consider that a conservation fee must be implemented. Additionally, 44% of visitors traveled with family, 33% with friends, 11% in a couple (spouse/wife/partner), and a minimal percentage traveled alone.

3.2. Econometrics

This research included variables such as travel price, socioeconomic characteristics of WS wildlife tourism visitors, and site environmental conditions in estimations (Table 1). An example of this is variable *dom*, which serves as a proxy for distance; variable *consd* represents site quality; and variable *lggroup* is a proxy for boat crowding. Then, four separate regressions were estimated (Table 2).

Table 1. Variables included in models.

Variable	Description
<i>V</i>	Dependent, number of trips in the last 5 years, ($V \geq 0$)
<i>dom</i>	1: If the visitor is domestic or local, 0: Other
<i>alone</i>	1: If the visitor travels alone, 0: Other
<i>lggroup</i>	Natural logarithm for the number of tourists on board the boat, including the interviewee
<i>sanct</i>	1: If the visitor agrees to inform citizens and visitors that the WS area is a natural sanctuary; 0: Other
<i>gguide</i>	1: If the visitor considers that the guide had a good performance during the tour, 0: Other
<i>rules</i>	1: If the visitor was informed about the rules to carry out the WS wildlife tourism activity; 0: Other
<i>first</i>	1: If the visitor is visiting the WS area for the first time
<i>consd</i>	1: If the visitor considers that the conservation status of the WS area is deficient, 0: Other
<i>couple</i>	1: If the visitor visits the WS area with his couple/wife, 0: Other

Estimation outcomes (Table 2) indicate that the associated coefficients for travel price (*tp*) in demand functions present the expected sign (negative) and are statistically significant at the 1% level. Furthermore, the coefficients of the remaining regressors are statistically significant at conventional levels, and their signs are consistent across all models.

A. C. Cameron and Trivedi (2007) and Wooldridge (2002) note that values for the McFadden R^2 to consider consistent estimations in cross-sectional data should range between 0.20 and 0.40; higher values indicate a better model fit. All McFadden R^2 values on estimated models exceed 0.20 (Table 2). So, models present consistent estimations. According to C. A. Cameron and Windmeijer (1996, 1997), more suitable statistical tests for

measuring model adjustment in similar count models include the R-squared based on deviance residuals and the Pearson R-squared. In this research, Pearson R-squared is used to assess model goodness-of-fit. Considering this argument, the best model is the *TP* model, which has the highest Chi^2 value and the closest pseudo-log-likelihood to zero in absolute terms. Over-dispersion tests were also conducted for each model; the output indicates rejection of the null hypothesis of no over-dispersion.

If *tp* increases by 10%, then visitation will decrease by a percentage equal to the variable coefficient multiplied by 10. Suppose the visitor is local or domestic (*dom*). In that case, visits will likely increase by almost one-third, since these types of visitors can afford to repeat the experience. If a visitor travels *alone*, the number of visits will decrease by a factor associated with the variable's coefficient, while holding the other variables in the model constant. Supposing the number of people on board the boat (*lgroup*) increases by one, meaning that the boat is getting crowded, the visitations will decrease by between 73% and 85% (depending on the model). Suppose that citizens and visitors are informed that the WS area was declared a wildlife sanctuary (*sanct*). Unlike what was expected, in that case, the visits will drop by approximately a factor of 0.48 to 0.60, depending on the estimated model. This last point may be due to three reasons (Abegão, 2019; Manning, 2011; Stankey & McCool, 1984). One, the label "sanctuary" imposes strict rules, affecting visitor expectations and displacing those wanting intense recreation. Two, sanctuary status promotes conservation norms. Highlighting what should be done helps visitors self-regulate and avoid protected areas. Three, the declaration changes the area's focus to species preservation, discouraging visitors with conflicting motivations who may harm wildlife.

If visitors consider the guide's performance during the WS trip to be good (*gguide*), then visits will most likely increase by almost two. Visitors regularly expect good treatment and service from tour operators; if one of the main aspects of the WS wildlife activity is perceived as poor, there is a high likelihood that visitors will not return. When the *rules* for swimming with WS are explained to the visitor, the odds of revisiting the site increase by a factor of 0.14–0.17. If the visitor is practicing the WS wildlife activity for the *first* time, they are likely not to return. Opposite to what could be expected, when the visitor's appreciation of the site's quality is deficient (*consd*) for performing wildlife tourism activities, the visit will increase by almost one-third. The price-quality theory suggests that lower prices are often accompanied by lower quality, and vice versa (Gupta et al., 2025; Carrillat et al., 2024). In contrast to the survey findings, most visitors reported performing the activity with friends or family. The variable *couple* results are statistically more significant than performing the activity with friends or family. This could be interpreted as follows: if the visitor is visiting the site with their partner or spouse (*couple*), then the visitation rate decreases by approximately 0.60.

Table 2. Estimated models for WS wildlife tourism activity.

Dependent: V	Poisson Model			
	<i>TP</i>	<i>TPOME</i>	<i>TPTTC</i>	<i>TPTTCOME</i>
	<i>n</i> = 334	<i>n</i> = 334	<i>n</i> = 334	<i>n</i> = 334
<i>ltp</i>	−0.7198 [0.1229] **			
<i>ltpome</i>		−0.2087 [0.0391] **		
<i>lpttc</i>			−0.6449 [0.1230] **	
<i>lpttcome</i>				−0.5804 [0.1063] **

<i>dom</i>	0.3809 [0.0819] **	0.2588 [0.0822] **	0.3384 [0.0791] **	0.2296 [0.0739] **
<i>alone</i>	−0.7283 [0.1970] **	−0.5016 [0.1435] **	−0.7323 [0.1945] **	−0.5566 [0.1500] **
<i>group</i>	−0.8575 [0.3175] **	−0.7316 [0.3165] *	−0.8428 [0.3201] **	−0.7865 [0.3163] *
<i>sanct</i>	−0.6054 [0.0894] **	−0.4869 [0.0858] **	−0.5753 [0.0890] **	−0.5268 [0.0860] **
<i>gguide</i>	2.0483 [0.2075] **	2.3514 [0.2002] **	2.1046 [0.2041] **	2.2958 [0.1929] **
<i>rules</i>	0.1672 [0.0524] **	0.1492 [0.0696] *	0.1632 [0.0570] **	0.1547 [0.0624] *
<i>first</i>	−1.0402 [0.0631] **	−1.0732 [0.0637] **	−1.0251 [0.0640] **	−1.046 [0.0637] **
<i>consd</i>	0.3212 [0.0841] **	0.3394 [0.0849] **	0.3243 [0.0848] **	0.3299 [0.0842] **
<i>couple</i>	−0.6101 [0.0903] **	−0.5983 [0.0824] **	−0.6131 [0.0867] **	−0.6057 [0.0842] **
<i>_cons</i>	3.0114 [0.4343] **	2.5788 [0.3751] **	3.0912 [0.4792] **	3.0205 [0.4467] **
Chi ² Wald (10)	716.47	712.41	712.64	715.54
Pr > Chi ² Wald (10)	0.0000	0.0000	0.0000	0.0000
Pearson R ²	0.8422	0.8329	0.8390	0.8374
Pseudo LL	−535.09301	−543.5885	−537.0061	−542.02082

* $p < 0.05$; ** $p < 0.01$. Standard errors within brackets. Source: Author's elaboration.

3.3. Willingness-to-Pay

Using a semi-log function on the estimated models precludes the use of equation 8 to calculate WTP. Because of this situation, Christiersson (2003) recommends two steps for WTP estimation. One, the calculation of elasticity for a semi-log model must be calculated by $\varepsilon_{tp} = \frac{-\beta_{tp}}{\bar{x}}$ (Jiménez & Rosat, 1997), where β_{tp} is the travel cost coefficient and $\bar{x}$ is the average visits. Two, estimate WTP by $\bar{x}/-\varepsilon_{tp}$. The estimated elasticities are all inelastic across the four models. Table 3 shows the estimated WTP, WS-REV for the four models, and the economic value per whale shark, and the average economic value per WS was also estimated.

Table 3. WTP, WS-REV, and economic value per WS. La Paz Bay (US\$).

Concept	Poisson Model				Average
	TP	TPOME	TPTTC	TPTTCOME	
WTP	8.00	27.00	9.00	10.00	13.50
WS-REV	304,600	1,028,025	342,675	380,750	514,013
Upper value per WS	4173	14,083	4694	5216	7041
Average value per WS	3016	10,178	3393	3770	5089
Lower value per WS	2361	7969	2656	2952	3985

Source: Author's elaboration. WTP: Willingness-to-pay, WS-REV: Whale shark recreational economic value, WS: Whale shark.

4. Discussion

4.1. Environmental Valuation

Economic valuation aims to transform WTP into environmental policy instruments. Among these are levies, charges, and access fees. The purpose of these instruments is to

become in-demand control instruments for national parks and protected areas, positively or negatively affecting tourism affluence to these sites, reducing anthropic pressure, alleviating operational costs, and strengthening conservation programs and enforcing surveillance (Greiber, 2009). Marine Protected Areas in Mexico and in almost all emerging economies suffer from a lack of federal economic resources to support basic operations.

Conservation efforts and actions demand economic resources. Fortunately, stakeholders are already organized and generating cooperation. Protected Area Managers are considering implementing a Visitors Management Program (VMP) that includes a voluntary fee for area conservation. This research can serve as a baseline for establishing an initial fee. However, Milder et al. (2010) argue that these strategies tend to fail, mainly due to three factors. First, rural communities must have access to scenic and recreational resources, and negotiations must be made with tour companies and other users to purchase access rights. Two, park entrance fees collected by governments of emerging economies are usually not used to support sustainable management in local areas. Third, rural or small communities will require technical support, advice, and capacity-building to manage tourism and recreation demand and capture an equitable share of the revenue generated by these activities. The most notable limitations of a VMP are the high investment required to implement it and its long-term sustainability (Grima et al., 2016).

Based on estimated WTP, a voluntary conservation fee (VCF) for WS wildlife tourism in La Paz can be proposed to strengthen and fund VMP. These kinds of schemes could be applied more generally. It is essential to note that implementing the VCF scheme can reduce demand for wild shark tourism if the threshold is set too high. Remember that this kind of scheme is a voluntary transaction between operators and visitors, oriented to secure WS habitat and, consequently, the flow of its recreational services—i.e., marine biodiversity (Forest Trends & The Katoomba Group, 2010; Wunder, 2005). The VCF could be administered as a small add-on fee to tour prices, with revenues pooled into a community-managed conservation fund. Nevertheless, a critical factor, in addition to financial flow, is maintaining the long-term flow of ecosystem services. It is worth noting that the VCF could yield economic benefits for the site and facilitate management, as well as periodic and autonomous actions that need to be performed on the site (such as verification and surveillance), and assess their impact on ecosystem services. These are actions that might be difficult to manage without this scheme (Chen et al., 2020; Koch & Verholt, 2020). Some authors are convinced that VCF can strengthen the socioeconomic benefits derived from wildlife tourism and protect and promote sustainable uses, while also involving local communities (J. Ziegler & Dearden, 2021).

On the other hand, the economic value per WS could serve as a baseline to establish fines for boats and ships that cause injuries to whale sharks, or for illegal operators within the WS area.

4.2. Community-Based Management

Community-based management is not the optimal solution for conserving marine and coastal ecosystems (Berkes, 2007). Esmail et al. (2023), Cárdenas-Torres et al. (2007), and Rowat and Engelhardt (2007) identify five key limitations of CBM which are as follows: (i) lack of comprehensive policies and legal frameworks for the conservation of marine and coastal ecosystems in emerging economies, making it troublesome to recognize community rights, (ii) lack of surveillance and remote monitoring limiting the quick response to externalities, (iii) unsustainable use of marine resources by illegal operators that damage and deplete marine biodiversity, (iv) access disputes for accessing the resource, since communities do not have ways to achieve agreements without local authorities' help, and (v) emerging economies lacking legal instruments to include traditions and traditional community knowledge in conservation policies.

Nevertheless, there are specific opportunities for implementing CBM. According to Esmail et al. (2023), they can be listed as follows: (i) recognizes the rights and traditions of communities and promotes their integration into resource management, (ii) promotes the use of renewable energy in communities through projects and technologies, (iii) is committed to global initiatives by strengthening local conservation capacities linked to national and international goals, (iv) promotes capacity building to increase community resilience and conservation actions, (v) promotes economic instruments for conservation (such as payments for ecosystem services, offsets, among others) to diversify sources of income.

In addition, it is possible to encourage the collected amount from VCF to promote, strengthen, and enforce some of the principles of the CBM such as the following: (i) development of social capital and collaborative partnerships, (ii) fair distribution of benefits and addressing local economic and social needs, (iii) funding research and information development by using integrated scientific, social, and local knowledge for informed decision-making, development, and monitoring, as well as surveillance, maritime signaling, feedback, and accountability systems, (iv) programs addressed to assess optimal environment, preconditions, or early conditions, and (v) fostering cooperation among stakeholders by promoting public participation and mobilization meetings to engage diverse stakeholders in the decision-making and implementation process, as well as conflict resolution.

These actions could help reduce the open-access problem arising from overlapping legal frameworks. The area falls under federal jurisdiction of the Natural Protected Areas Commission (CONANP, by its acronym in Spanish) and the standard regulation SEMAR-NAT-NOM-059 for protected species, but specific, enforceable instruments for daily tourism operation are lacking (García-Baciero et al., 2025). There are two main issues: disputes over access among different tour operators, leading to overcrowding, and a lack of clear rules on tour limits and revenue sharing.

Economic valuation is an important tool for legal and policy reform, providing the rationale for the establishment and enforcement of limited-entry systems by regulators and resource managers and demonstrating that well-managed and reduced access enables long-term benefits for legal local operators (Pham-Do & Pham, 2020). In addition, it justifies the use of legal instruments such as exclusive rights or prioritized concessions, attached to enforcement protocols (Djunaidi et al., 2021; Cárdenas-Torres et al., 2007). Thus, REV transforms from a point of conflict into the foundation for a legal instrument that could also guarantee equity, sustainability, and community stewardship.

5. Conclusions

The research found that the presence of WS is essential in wildlife tourism activities in La Paz. Perhaps, without WS, the local economy could shrink because visitors might not come or stay as long. Therefore, WS conservation and protection projects are crucial for tourists to continue visiting the WS area, thereby increasing local tour operators' income and generating direct and indirect economic benefits. The estimated WTP for WS wildlife tourism activity ranges from USD 8.00 to 27.00, and the recreational value for this activity oscillates between USD ~0.3 million and USD 1.03 million. Recreational values can be interpreted as the economic value of a potential loss or damage to the whale shark recreational activity. The z-values for the variable coefficients in the four models, along with the Chi-squared and pseudo log-likelihood values, provided statistical validation for our analysis. The travel price variable was statistically significant in the models; therefore, the estimated WTP per trip per person is reliable.

Practicing wildlife tourism with WSs in La Paz Bay has some competitive advantages over other sites where WS wildlife tourism is practiced (such as Holbox in Quintana Roo or Bahía de Los Ángeles in Baja California). These advantages are as follows: (i) WS price

per trip is lower than the other sites, (ii) tour operators' variable costs are lower than in Quintana Roo, and (iii) connectivity is much better than in Bahía de Los Ángeles. Tour operators are not fully seizing these advantages. In addition, according to the service–price–quality paradigm, those tour operators in La Paz who dare to improve and invest in capital, quality service, and human capital can expect increasing economic returns.

Our recommendation would be to use these estimated values for WTP as a baseline to negotiate with stakeholders and together choose the most adequate scheme, as well as to define priority conservation strategies to implement in the WS area, and fund them with the collected amount, which could be taken as the WS area recreational value. Stakeholders involved in WS wildlife tourism should consider implementing a general fee to support conservation, regulatory enforcement, surveillance, management of nature-based destinations, maritime signaling, an online reservation system, WS monitoring, and data generation. Nevertheless, we must emphasize that a nature-based destination management scheme is a great way to support and strengthen other economic instruments aimed at conserving WSs and their habitat in La Paz Bay. The research provides an economic baseline that stakeholders can consider when funding conservation and protection projects in the study area, should an economic instrument be implemented.

Emerging economies have numerous opportunities to leverage global partnerships, innovative technologies, financial tools, and rights-based approaches to enhance CBM attached to financial instruments such as VCF. This will help promote sustainable development and protect biodiversity. On the other hand, limitations show that marine CBM in developing countries has its own set of structural, technological, and socio-political problems. To address these issues, it is crucial to strengthen laws, make it easier for people to use marine technology, regulate fishing by outsiders, and incorporate local knowledge into policy-making.

The presented research is static and limited by the temporal, political, and environmental framework in which it was conducted. The applied methodology cannot incorporate other types of values, such as legacy values or indirect and direct use values. If these values were considered the TEV of WS, it could be even greater. Finally, conservation and protection projects with a lower cost than the recreational value of WS wildlife tourism are economically feasible for a VCF to fund.

At the time this research was carried out, official regulations were barely established by local, state, or federal authorities, but some local tour operators had private agreements regarding prices, how to conduct the tour on site, including the number of boats surrounding a WS, routes of entry and exit to the WS area, and staying time in the area. In the 2018–2019 season, some agreements were incorporated into the management plan and regulations, while others were adapted through consensus. More recently, managers have adopted a capacity-building strategy to help tour operators conduct WS wildlife tourism activities more efficiently.

These are joint efforts by three agencies focused on tour guide training. First, Natural Protected Area Managers focuses on training about WS area rules and zoning; second, the State Tourism Office tour guides training focuses on the NOM-009-TUR, which specifies the information procedures and requirements to protect and conserve environmental, natural, and cultural assets needed to perform the WS activity with visitors; and finally, the Harbor Master trains tour guides on sailing, navigation rules and laws, and sea trade. In addition to these efforts, the Citizen Committee for Whale Sharks voluntarily monitors the WS area to minimize injuries, compiles its own statistics on the number of visitors by season, and reports illegal boats or activities in the area to the environmental authority. Nevertheless, these actions require funding to remain consistent and strengthen the management of whale shark tourism in La Paz Bay, which could be achieved by implementing a VCF as an IBC strategy.

Lastly, when OPC and TTC were included in the estimated models, the goodness-of-fit for the TPOME, TPTTC, and TPTTCOME models decreased. Therefore, adding OPC and TTC costs may introduce noise or collinearity, or the simple additive model may be misspecified. More substantial evidence is needed to reject fundamental aspects of the TCM literature with such certainty. Therefore, these costs should not be included in economic valuation, as they do not contribute to improving models.

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Appendix A

World Economic Valuation Studies for Wildlife Tourism

Author(s)/Year	Area/Site	Specie	USD *
(Wilson & Tisdell, 2002)	South-Eastern Queensland, Australia	W	17,189,043
		ST	509,666
(Enríquez-Andrade et al., 2003)	Bahía de los Ángeles, México	WS	3,554,722
(Topelko & Dearden, 2005)	Belize	WS	4,204,918
(Rowat & Engelhardt, 2007)	Seychelles	WS	7,687,869
(Catlin et al., 2010)	Ningaloo, Australia	WS	5,409,485
(Vianna et al., 2011)	Viti Levu, Fiji Islands	S	4,626,350
(Du Preez et al., 2012)	Aliwal Shoal Marine Protected Area, South Africa	S	473,501
(Vianna et al., 2012)	Palau, Island	S	25,672,131
(Cisneros-Montemayor et al., 2013)	Belize	S	469,054

	Mexico	S	16,127,156
(Cagua et al., 2014)	South Ari, Maldive Islands	WS	5,855,738
(Ruiz-Sakamoto, 2015)	Revillagigedo National Park, Mexico	S	8,377,049
(Schwoerer et al., 2016)	Eastern Pacific Coast Baja California Sur, Mexico	W	281,185
(Huveneers et al., 2017)	Australia	S	43,237,704
(Cisneros-Montemayor et al., 2019)	Mexico	S	12,400,000
(Cisneros-Montemayor et al., 2020)	Baja California Sur, Mexico	SS	47,000,000
(Oropeza-Cortés et al., 2023)	Laguna San Ignacio, Mexico	GW	908,502

Source: Author's elaboration. Note: GW: grey whale, S: sharks, SS: several species, ST: sea turtles, W: whales, WS: whale shark. * Direct expenditure in 2020 updated USD.

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