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Regional Governance Framework for the CLME project

Deliverable 4:

The Valuation of Marine Ecosystem Goods and Services in the Caribbean

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List of abbreviations

BAU	Business as usual
BVI	British Virgin Islands
BMP	Bonaire Marine Park
CA	Cost avoidance method
CBA	Cost benefit analysis
CLME	Caribbean Large Marine Ecosystem
CM	Choice modeling
CS	Consumer surplus
CTO	Caribbean Tourism Organization
CVM	Contingent valuation method
GDP	Gross domestic product
GSMP	Gladden Spit Marine Park
HP	Hedonic price method
MBMP	Montego Bay Marine Park
MEY	Maximum economic yield
MSY	Maximum sustainable yield
MP	Market price method
NPV	Net present value
PF	Production function method
PS	Producer surplus
RC	Replacement cost method
SVG	St. Vincent and the Grenadines
TCI	Turks and Caicos Islands
TCM	Travel cost method
TEV	Total economic value
USVI	United States Virgin Islands
WCR	Wider Caribbean Region
WRI	World Resources Institute
WTA	Willingness to accept
WTP	Willingness to pay

Executive Summary

This report provides a summary of economic analyses of marine ecosystem services in the Wider Caribbean Region for the three major marine ecosystem types being addressed by the Caribbean Large Marine Ecosystem (CLME) Project: reef, pelagic and continental shelf. Particular attention is given to empirical valuation studies. An overview of existing valuation methodologies is provided, along with recommendations for applications in the in the CLME Project area. Advantages and disadvantages of alternative methods are discussed. Market and non-market valuation studies from peer-reviewed journals, proceedings, government archives and university databases are reviewed. Attention is also given to other empirical work at the nexus of economics and marine resources in the Caribbean such as the use and effects of economic incentives, economic impact studies, and conservation finance. Studies reviewed for this report include analyses of primary and secondary data, benefits transfer applications, meta-analyses and case studies. Gaps, deficiencies and unknowns in the extant literature are highlighted and discussed. The summary and analysis of gaps together form a framework for valuation in the Wider Caribbean Region so that future work can be directed toward areas of policy importance.

Approximately 200 individual value estimates were reviewed for this study. To date, marine economic valuations in the WCR have focused on only a limited number of benefits derived from marine ecosystem goods and services, primarily benefits that are relatively easy to measure and convey to the public, such as recreation opportunities in near-shore protected areas, and benefits that are ascribed to easily measured market indicators such those derived from real estate and capture fisheries. Values associated with reefs have received far more attention than those associated with the pelagic or shelf ecosystems, no doubt due to the ease of access to associated user groups by researchers and the relatively straightforward linkages between changes in resource quality and well-being.

Despite a plethora of market data and evidence of overfishing in the WCR (CARSEA, 2007), the economic impacts of overfishing remain largely unexplored. These include effects on national economies, employment, food security and tourism. Likewise, the economic practicality of fisheries subsidies in terms of the relative values of contemporaneous support livelihoods and future economic costs overfishing remains unknown.

As overfishing could potentially lead to the loss of historic fishing heritage as well as an important source of protection against exogenous economic shocks, understanding the values from the cultural and security benefits of small scale fisheries deserves of attention. Other cultural service values that remain largely unknown include the value of WCR marine ecosystems to research and education and the amenity value of reefs to coastal property owners. This latter value, estimable via the HP method, would appear to be an important partner to studies that estimate the value of reefs for coastal protection.

Supporting and regulating services provided by the marine ecosystems of the WCR that have been recognized as important in the context of natural sciences, have not be linked to valuation. Examples include the contribution of Caribbean reefs and other coastal ecosystems to fisheries production, climate regulation and habitat provision. With appropriate modeling, the PF method could be applied to improve our understanding of these values.

Finally, despite the continued improvement of appropriate methodologies, estimates of non-use values for WCR marine ecosystem goods and services are in short supply.

The matrix of ecosystem services by major marine ecosystems serves as a framework for reviewing the status of evaluation studies in the Wider Caribbean Region. It is suggested that future work on valuation be coordinated among countries and agencies so that gaps can be prioritized and addressed through conceptual models and valuation studies that will lead to a more comprehensive understanding of the full value of the goods and services provided by all three major ecosystems.

1 The economics of marine ecosystem goods and services

1.1 Introduction

Marine ecosystems supply a variety of goods and services that provide direct and indirect contributions to human well being. These include goods traded in formal markets such as food and raw materials as well as non-market goods and services such as nutrient cycling, climate regulation, coastal protection and opportunities for recreation. A broad literature provides classifications of ecosystem services. Well-cited examples include Daily (1997), DeGroot et al. (2002), Boyd and Banzhaf (2007), Moberg and Folke (1999) and Wallace (2007). The 2005 Millennium Ecosystem Assessment (WRI, 2005) also provides a typology, categorizing ecosystem services as supportive, regulating, provisioning or cultural. That report enumerates a variety of examples and descriptions of ecosystem services. Table 1 gives some common examples and is the typology that will be used for this report.

Assessing ecosystem services requires measurement of stocks or flows of ecosystem processes and structures and the level of services that are provided (Farber et al, 2006). That is, in order to characterize the state of an ecosystem, units of measure must be clearly defined (Boyd and Banzhaf, 2007). Consistent measurement and accurate portrayal of environmental conditions are prerequisites for economic analysis of ecosystem goods and services that is intended to inform policy.

Table 1: Ecosystem Services

Supportive Services	Regulating Services
Nutrient Cycling Net Primary Production Pollination and Seed Dispersal Habitat Hydrological Cycle	Gas Regulation Climate Regulation Hazard Protection/ Disturbance Regulation Biological Regulation Water Regulation Soil Retention Waste Regulation Nutrient Regulation
Provisioning Services	**Cultural Services**
Water Supply Food Raw Materials Genetic Resources Medicinal Resources Ornamental Resources	Recreation Aesthetics Science and Education Spiritual and Historic

Adapted from Plantier Santos (2010).

1.2 Services, Benefits and Value

Understanding and measuring the associated contributions to human well being is the domain of economic valuation, hence it is important at the onset of this report to link notions of ecosystem services with concepts of human well being and economic value. As noted in Boyd and Banzhaf (2007) and echoed by Fisher and Turner (2008), ecosystem services (processes and functions) are not synonymous with ecosystem benefits (outcomes). While the former may be especially difficult to quantify, measurements of the latter are more often of interest for informing policy. More importantly for the purposes of this review, it is these benefits that provide the basis of economic valuation studies. For example, consider the case of a coral reef ecosystem. The processes and functions of the reef include biological production, biochemical processing, waste assimilation and maintenance of biological diversity. These processes and functions provide the benefits of food, recreation, aesthetics and damage/cost avoidance. The delineation between ecosystem services and ecosystem benefits helps us to avoid the problem of double counting when estimating economic values as we only seek to measure distinct benefits (Fisher and Turner, 2008).

Economists define the value of a particular good or service as what it is *worth to people*, in terms of the contribution of the good or service to well-being (Bockstael et al., 2000). Because this definition of value is best measured by what people are willing and able to pay for a good or service, value is often confused with cost. Cost, or what people have to actually pay for a good or service, is considered expenditure and may differ greatly from the value of the good or service. For example, a beach renourishment project may involve $1 million in physical and engineering costs, but may generate considerably more (or less) than that in actual economic value. Likewise, subsidies to developers or commercial fishers may cost society thousands of dollars but result in net economic losses. Such actions should be recognized as having positive costs but negative economic values.

It is also important to recognize that economic value extends beyond the marketplace to "*nonmarket*" goods and services such as clean water and diverse ecosystems, and may include benefits derived without any direct use or interaction with the natural environment whatsoever. These "non-use values" include benefits derived from simply knowing that a species or ecosystem exists, benefits from the knowledge that resources may be available for future generations or for potential future use or research and discovery. That people are willing to give up time or other resources (including money) for the opportunity to consume these goods and services lends evidence to this notion.

Distinguishing the notions of ecosystem services and ecosystem benefits allows us to partition the concept of economic value in a way that allows for a straightforward and commonly accepted classification scheme. For example, in addition to categorizing values in terms of whether or not they are revealed in formal markets, it is common to differentiate between values associated with use and those not associated with use. *Use values* include benefits derived from environmental goods that are associated with direct or indirect interaction with the environment. Direct benefits can be derived via extraction (e.g. fish harvests, raw materials) and would be associated with provisioning services in the WRI (2005) typology, while benefits derived from non-extractive direct interactions such as recreation, research and aesthetics would be associated with cultural services. Indirect use values include the benefits from damage or cost avoidance (e.g. protection of coastal real estate) and would be associated with regulating services.

Non-use values are benefits not associated with use, and include existence, option and bequest values. *Existence value* is value derived from simply knowing that a natural resource or environmental good exists. *Option value* is benefit derived today from knowing that a resource is available for potential future use. *Quasi-option value* is similar to option value but has an extra degree of uncertainty regarding the nature of the resource itself. This value is best described as the contemporaneous benefit derived from preserving the resource for potential discovery of future uses, and is most often associated with genetic material and medicine. *Bequest value* is value associated with an altruistic motive. That is, the benefits derived by one group from knowing that the resource is available for other groups. The majority of these values are associated with WRI notion of cultural services, though one could argue that option and quasi-option values have clear ties to provisioning services.

Table 2 presents a common categorization of the components of total economic value. It should be recognized however that the total value of a given species or ecosystem may be greater than the arithmetic total of these individual values, as the value to society of a healthy and functioning system may be more than the sum of the individual components of value (Turner et al., 2003). Conversely, if values categories are non-complementary, the total value of an ecosystem may be less than the sum of individual values.

Table 2: Categories of Economic Value

Total Economic Value	
Use values	**Non-use values**
Extractive use Non-extractive use Direct non-extractive use Indirect non-extractive use (damage avoidance)	Existence value Option value Quasi-option value Bequest value

1.3 Valuation

Economic valuation simply means estimating what something is worth to a group of people or to society at large. In short, valuation is the monetization of the benefits or costs associated with a good or service. We can understand the value of a good or service is by observing what most people are willing to give up (i.e., trade) to attain it. There are many situations where measuring and understanding the value of particular natural resources can be useful. In general, anytime there is a potential for tradeoff between market values and non-market values, economic valuation can serve as a means of facilitating this comparison by expressing all impacts in monetary units. This is based on the fact that alternative uses of natural resources create a range of impacts, which are usually not in comparable units (changes in fish stocks, water or air quality changes, or reef degradation).

Valuation of any sort requires an understanding of how changes in environmental goods and services affect human well-being, and then determining how much individuals are willing to pay (WTP) for beneficial changes, or willing to accept (WTA) as compensation for unfavorable changes (Bockstael, et al. 2000; Barbier, et al. 2011). Thus, estimating values allows a comparison of two alternative states of the world (e.g. with policy and without policy). When links between changes in the environment and human well-being are obscure or uncertain, valuation naturally becomes more difficult. Valuation may be precluded due to scientific complexity, human cognitive limitations, or when the monetary measures are deemed morally intractable because of prevailing social norms (Turner et al., 2003). Given the classification scheme for values outlined above, it is not surprising that the majority of valuation studies derive estimates for benefits associated with provisioning and cultural ecosystem services, and relatively few address supportive and regulating services. Indeed, while values linked to supportive and regulating services are clearly non-market in nature, their estimation takes place outside of the realm of traditional non-market valuation (described below), and requires a different set of empirical approaches.

1.4 The importance of valuation

Despite the fact that the entire market economy depends on the existence of natural systems, values derived from environmental goods and services are often ignored by policy makers. Part of the explanation for this stems from the fact that people and governments most often respond to monetary price signals which may differ from economic values (Dixon, 1998). Without an understanding of the monetary worth of natural resources, conservation efforts may be stymied because they are viewed as costly in terms of precluding activities that have large immediate financial rewards (Schuhmann et al., 2011). Simply put, the true value of natural resources, in the Caribbean and around the world, is largely unknown, and as a consequence may not be given due attention in the policy process.

Valuation studies can fill this void by generating information on the costs associated with species and habitat loss, the benefits of conservation and restoration efforts, or economic dependence on natural systems. When reported in a common and easily understood monetary metric, such information can serve as a valuable input into decision-making processes attempting to manage the allocation of scarce resources among competing demands (Turner et al., 2003). At the national level, economic valuation can support decision-making by ministries, the private sector, and organizations involved in the use and management of natural systems. Incorporating the value of natural assets into national income accounts, though difficult, may allow for a more accurate indication of economic performance or national wealth (Dharmaratne and Strand, 1999). Public and private sector organizations that indirectly affect natural systems via actions or budgetary decisions can also benefit from knowledge gained through valuation studies, but may be less convinced of the importance of valuation than direct stakeholders (Lange and Jiddawi, 2008).

2 Valuation Methodologies

2.1 Methodologies based on market data

2.1.1 The market price approach

Approaches to valuation can take many forms, coincident with the many ways that humans can interact with the environment and the array of benefits that result. The method chosen often depends on what is being valued and the intended use or policy purpose of the values. When value is easily revealed through market transactions (such as benefits associated with direct extractive uses), monetization may be accomplished via the relatively straightforward *market* *price approach* (MP). Sometimes referred to as *financial analysis*, this approach values environmental goods and services based on profits or market value-added (Huber and Ruitenbeek, 1997). In its simplest form, the MP method constructs estimates of total expenditure by purchasers (equivalent to revenue received by producers) based on market prices and quantities. Such estimates ignore costs of providing the goods and services and as such are not reflective of net gains to market participants and therefore should not be considered true estimates of economic value. Common applications of the basic form of the market price approach include estimates of the gross value of commercial fisheries (e.g. those produced by FAO). When time series market data are available, demand and supply relationships can be estimated that allow measurement of total expenditure and consumer and/or producer surplus (net gains). Taking the market price approach to this level of detail is sometimes referred to as the *net factor income approach* (Brader et al, 2006). Consumer surplus is the difference between what consumers are willing to pay for a good and what actually is paid (market price). Similarly, producer surplus is the difference between what sellers are willing to accept (typically associated with the marginal costs of production) and what they actually accept (market price). Total economic benefit is the sum of consumer and producer surplus, and represents net gains from the good in question. Figure 1 below illustrates consumer surplus (CS) and producer surplus (PS).

An advantage of the MP approach is the relative ease of calculation, especially when market price and quantity data are available from secondary sources such as fisheries divisions or national statistics offices.[1] Further, the fact that values are revealed through actual transactions lends undeniable credibility to these estimates. Disadvantages include the difficulty in deriving estimates of net gains (producer and consumer surpluses) from market prices, which can only be considered lower bounds on true value. Small scale fisheries data are inconsistently measured over time and space across the Caribbean, making the MP approach difficult to apply (Salas et al., 2007). Even with reliable data, market valuation based on annual average prices may obscure seasonal or geographic variations in value (Abaza and Rietbergen-McCracken, 1998).

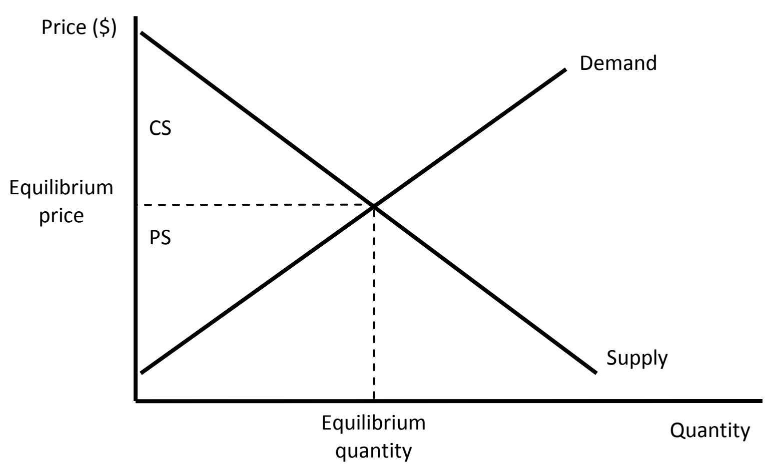

Figure 1: Market gains (producer and consumer surplus)

[1] Unlike prices and quantities which are typically revealed at the market level, cost data may be specific to individual market participants and are therefore more difficult to obtain.

2.1.2 The replacement cost approach

The *replacement cost approach* (RC) is based on the idea that some goods and services provided by the natural environment can be replaced with manmade goods and services. Estimates of the costs of providing these replacement services are used as the value of the associated naturally provided services. For example, the costs associated with constructing an artificial breakwater may be used as a proxy for the value of a reef that provides the same service, or the costs associated with building and maintaining a water treatment plant may be used as an estimate of the value of the water cleaning service provided by wetlands. This method is applicable only in situations when the natural service can be suitably and equivalently replaced with a manmade alternative, the costs of that substitute are known or estimable (WRI, 2009) and represent the least-cost means of providing the service, and when society is willing and able to incur the costs associated with the replacement (Bockstael et al., 2000). When these conditions are not met, use of the replacement cost approach is not valid (EPA, 2009).

Advantages of the replacement cost approach include relative ease of calculation based on market data. The values conveyed by this method are also easily understood by policy makers and the public at large as representing the opportunity costs associated with failure to protect natural assets. The principle disadvantage of the method is that it is not a true means of measuring the value of ecosystem goods and services in the sense of gross or net benefits to people. The method provides a measurement of costs, which may not reflect the true value of in-situ environmental goods and services. Indeed, it seems unlike that man-made alternatives will provide the full range of benefits provided by natural assets. Application of the replacement cost approach may therefore leave policy makers with insufficient information regarding the true gains from outlays or interventions designed to prevent damage to natural assets (Abaza and Rietbergen-McCracken, 1998).

2.1.3 The cost (damage) avoidance approach

Similar to the replacement cost approach, the *cost (damage) avoidance approach* (CA) uses estimates of the expenditures that would be incurred to prevent, diminish or avoid harmful effects associated with degradations to natural resources. This perspective views the cost savings associated with reduced spending on human and environmental health as benefits of maintaining ecosystem services or preventing their decline. For example, the cost of replacing coastal real estate may be used as an estimate of the storm protection service derived from healthy reefs, or the value of reducing bacterial counts in surface water may be quantified by estimating the associated reductions in costly incidents of diarrhea (Farber et al. 2006).

It is important to reinforce the idea that the replacement cost approach and cost avoidance approaches ascribe estimates of costs to notions of value, which may be an inherently flawed means of understanding the benefits derived from changes in natural resources. For example, the cost of building an artificial reef may be wholly unrelated to the benefits derived from a natural reef. Using the value of coastal real estate as a proxy for the value of reefs may lead analysts to conclude that degraded reefs adjacent to highly developed coastlines are worth more than healthy reefs where coastal development is limited. These methods should be only used with the appropriate cautions and caveats and should not be used in isolation (i.e. without considerations of other benefits derived via alternative methodologies).

2.1.4 The production function approach (productivity method)

The *production function approach* (PF), which is sometimes referred to as the *productivity method*, links the impact of a change in environmental conditions to the provision of particular goods or services by using a model describing the production relationship. The resulting change in the output of the good or service is then valued via other methods such as those described above and below. For example, if commercial fishery yields can be modeled as a function of mangrove acreage or reef quality, then changes in the quality or quantity of reefs or mangroves can be valued using the market price method using estimates of their impact on fisheries output. Cartier et al. (1999) suggest that production function approaches should attempt to focus on valuation of a limited number of locally important use values under different impact or conservation scenarios in order to provide insight into the relative scale of benefits as well as a comparative basis or benchmark for values that may be more difficult to measure or involve greater uncertainty.

A potential obstacle in implementing the PF approach is the need for an integrated multidisciplinary approach. Application of this method requires an appropriate understanding of the relationship between the environmental resource and the resulting impacts on the production of the good or service of interest (WRI, 2009). In short, outputs from ecological models of bio-physical relationships are needed as inputs to economic valuation models. As noted in EPA (2009), indentifying the need for such collaborations amounts to identifying ecological processes that produce responses that affect human well-being. Producing the outputs that are amenable to valuation may involve an additional level of modeling or measurement that is beyond the typical scope of analysis for researchers in a particular discipline, and may necessitate education of the general public in terms of the importance of ecological change (EPA, 2009). Clearly, such efforts must take place early in the valuation process. Despite this limitation, the PF approach holds great promise for the valuation of a more complete range of ecosystem services.

2.2 Non-market approaches

The estimation of values that occur outside of markets or those that are only partially associated with markets (especially those associated with non-use), while much less known outside the economics profession, are facilitated by a variety of valuation techniques. A brief discussion of the more popular and practical techniques are presented below. For an in depth discussion on the full range of methods and the history of non-market valuation, see Bockstael, McConnell, and Strand (1989), Bockstael, Hanemann, and Kling (1987) and Braden and Kolstad (1991).

2.2.1 Revealed preference methods

Revealed preference methods include the *travel cost method* (TCM) and *hedonic pricing* (HP). These methods examine decisions that individuals make regarding market goods that are used together with non-market goods to reveal the value of the non-market good (Kahn, 1998), and require that a link be established between changes in the environmental resource and changes in the observed behavior of people. For instance, changes in beach width or reef quality may result in tourists moving to another location or taking fewer trips. With this information, a demand or marginal willingness to pay (WTP) function can be estimated, which allows one to estimate the value of particular changes in the natural resource. The principle advantage of revealed preference methods is that resulting values are grounded in actual behavior and are therefore empirically defensible. However, these methods are not suitable for monetization of non-use values.

2.2.2 The Travel Cost Method

TCM, one of the most widely used revealed preference valuation techniques, uses information on actual behavior to estimate a trip demand curve from which the value of the resource can be derived. This method is most commonly employed to value natural resources associated with recreation, and can be applied to changes in the quality or quantity of environmental attributes at recreation sites, changes in accessibility or number of sites and willingness to pay user fees (Birol et al., 2006).

A trip demand curve is estimated using visitation data, including travel costs and the number of trips taken by each individual to a particular site. Using distance traveled as a proxy for the price of a trip, and the number of trips as the quantity, individual or group demand curves can be estimated for a site or destination. The net benefits of a particular site or the value of the resources within each site can then be estimated. When modified for international travel, this method can be employed to value the flow of recreation services from tourist activities related to the coastal and marine environment, though accuracy may be sacrificed with increased travel distance (Abaza and Rietbergen-McCracken, 1998). The TCM has been used by government and non-government agencies alike around the world to value a wide variety of non-market goods and services, including ecotourism and wildlife viewing in Costa Rica (Menkhaus and Lober 1996), diving in Honduras and Bonaire (Pendleton, 1994 and Pendleton, 1995), MPA tourism in Jamaica (Reid-Grant and Bhat, 2009) and reef tourism in Bermuda and Puerto Rico (Van Beukerin et al., 2009 and ETI, 2007).

Application of the TCM requires a detailed survey of individuals who travel to a particular location and use the associated resources. These data are typically acquired via survey instruments administered to travelers or recreationists. Travel log data reporting the dates and frequency of travel by individuals or households are ideal, but can be costly. On-site, phone or mail data pertaining to a single travel experiences are often sufficient, but may limit valuation to conditions existing at the time of travel (Birol et al., 2006). Exit surveys administered by tourism offices (e.g. CTO) often contain the variables appropriate for TCM estimation. Difficulties in valuation arise when the purpose for travel includes activities other than interaction with natural resources, as travel expenses, the price surrogate, must be partitioned among multiple purposes. Treatment of the opportunity cost of time is an important issue and has received attention in the literature (e.g. McConnel and Strand, 1981; Smith et al., 1983).

2.2.3 The Hedonic Pricing Method

The HP method recognizes that natural resource values will be reflected in the prices people pay for composite goods such as housing. For example, housing prices along the coastline tend to exceed the prices of inland homes because beaches provide recreational and amenity values to coastal property owners. Hence, when people buy a house, the price they pay not only reflects the materials that went into constructing that house, but also the number of bedrooms, square footage, whether there is a garage, neighborhood characteristics, and environmental characteristics and amenities. By collecting data on house characteristics and associated environmental attributes, the value of environmental amenities or changes in them can be estimated. Because this method relies on market real estate data, value results are highly defensible provided that associated environmental characteristics can be accurately quantified and have not undergone significant change since the time of housing transactions. Further, the HP method is limited to direct use values that easily be ascribed to real estate as perceived by the housing consumer (Birol et al., 2006). Examples of the HP method applied in the CLME include van Beukering et al. (2009) who estimate the contribution of reefs to the amenity value of real estate in Bermuda.

2.3 Stated Preference Methods

Stated Preference Methods include the Contingent Valuation Method (CVM) and Choice Modeling (CM, also referred to as Conjoint Analysis and Choice Experiments). While the revealed preference methods outlined above allow for the estimation of the value associated with recreation activities and property, such analyses do not permit the estimation of changes in values not associated with direct use (i.e. the non-use values described above). For example, tourists and residents may place value on the knowledge that the reef ecosystem and its wildlife are preserved in a particular way. To elicit such values, stated preference methods must be employed. CVM relies on direct survey questions to elicit values, while CM asks people to make hypothetical choices across bundles of goods or through ranking alternatives with "price" being one alternative or characteristic in the bundle.

Both of these techniques are well-accepted methods for valuing non-market goods and services and have been used around the world. As early as 1994, the CVM method has been used in over 1600 studies and 40 countries (Carson et al. 1994). The U.S. Department of Interior (DOI) has adopted CVM to measure non-market values associated with damages under CERCLA 1980 (DOI 1986), while NOAA has endorsed the use of this method for damage assessment under the Oil Pollution Act of 1990 (Arrow et al. 1993).

2.3.1 The Contingent Valuation Method

The goal of CVM is to create a realistic, albeit hypothetical, market where peoples' values for a good or service are expressed. A CVM survey constructed for deriving non-use values consists of four main elements: a description of the program the respondent is asked to value or vote upon (e.g, a conservation project); a mechanism for eliciting value or choice (e.g., a simple referendum type question that asks the respondent to vote "yes" or "no" to a specified price); a "payment vehicle" describing the manner in which the hypothetical payments are collected (e.g., higher taxes or a payment into a trust fund); and information on respondent attitudes and characteristics (e.g., socioeconomic characteristics and environmental attitudes). This exercise could be undertaken via an in-person or mail survey of residents and tourists to ascertain the value of a particular resource. CVM estimates are subject to numerous biases including strategic bias, hypothetical bias, starting point bias and information bias, which have made the method controversial and the subject of great debate in the literature (see Portney, 1994). Because of the extensive pretesting and groundwork required to develop survey instruments that surmount these difficulties, the CVM method can be prohibitively expensive (Birol et al. 2006). Whitehead (2000) and Huber and Ruitenbeek (1997) provide excellent guidelines and best practices for valuation via CVM. CVM is perhaps the widely used non-market valuation method in the CLME, with more than 20 applications detailed in this report.

2.3.2 Choice Modeling (Choice Experiments)

While CVM can be a powerful and useful tool in deriving value estimates for natural resources, a CM approach may be more useful in terms of determining the value associated with factors that contribute to tourists' destination choice (Forster et al. 2011) and as such may be more appropriate for valuation of coastal and marine resources in the Caribbean. The CM approach is increasingly gaining favor in the literature as it avoids many of the inherent biases associated with CVM and is more consistent with economic welfare theory than conjoint analysis (Hanley et al, 2001a). Unlike other valuation methods, CEs allow multidimensional attribute changes to be valued simultaneously, and can be used to generate estimates of the relative value of multiple attributes (Huybers, 2004). The choice modeling approach can therefore be used to analyze tradeoffs that individuals are willing to make between environmental factors and as such can be an important tool in guiding the allocation of scarce conservation resources. Data for a CM analysis are obtained from a survey designed to elicit preferences by guiding the respondent through a series of paired choice alternatives, each described in terms of different levels of attributes that comprise the product. On the basis of an experimental design, the descriptions of the alternatives vary across scenarios. By observing the changes in stated choices due to the variation in the alternative's characteristics, the effect of the attributes on the choices can be derived (Huybers, 2004).

2.4 Methods that rely on existing valuation estimates

2.4.1 The Benefits Transfer Approach

Developed for situations in which the costs of primary data collection for valuation are prohibitive, the benefits transfer approach spatially and/or temporally transfers summary estimates of environmental benefit from other case studies (i.e., the study site) to the policy case study (i.e., the policy site) (Dumas et al. 2005). The two principle approaches to this practice are benefit estimate transfer and benefit function transfer. The former directly applies summary estimates of environmental benefits (e.g. estimates of WTP) from the study site to the policy site, while the latter applies an empirical model of benefits to the policy site (e.g. coefficient estimates from a WTP model where independent variables include demographics and measures of environmental quality). By allowing characteristics of the policy site to be substituted into an empirical function of value, benefit function transfer may allow for greater accuracy of transfer (Loomis, 1992). Necessary conditions for a valid benefit transfer include a theoretically and methodologically valid application at the study site, similar population sizes and characteristics in the study and policy sites, similarity between pre-policy and post-policy environmental conditions at the sites, and similar distributions of property rights (Brouwer, 2000, Dumas et al. 2005). Value is also dependent upon the availability of substitutes (Bockstael et al., 2000), which often vary across sites. While this method provides a convenient and low-cost tool for valuation, when the study site is inappropriately matched to environmental or socioeconomic conditions at the policy site, benefit transfer estimates are subject to error (Plummer, 2009). Because of the significant variation in ecosystem values across sites (Steiner, et al. 2004), this method should be used with caution.

2.4.2 Meta analysis

Meta analysis is a method that synthesizes results from a collection of existing studies by regressing value estimates from similar studies on study and site characteristics (Woodward and Wui, 2001). Dependent variables can include measures of the resource being valued, valuation method, time, and sample characteristics (Loomis and White, 1996). The advantage of this method is the ability to empirically account for factors that cause variation in estimates of value (Smith and Kaoru, 1990). Fitted values can be generated and used as value estimates for the policy site by applying appropriate values of the independent variables.

2.4.3 Economic Impact Analysis

Related to valuation studies is the notion of *economic impact analysis*, which recognizes that the contribution of market transactions to an economy may substantially larger than is revealed by the market price method. For example, commercial fishery earnings serve to generate additional revenues, incomes and employment in national and regional economies. Hence, *economic impacts* include the values associated with output and revenues that flow from a particular market transaction as a portion of each dollar spent by a consumer represents revenue earned by someone else in the economy. As some of that generated income is used to purchase other goods and services, each new dollar spent and earned ripples through numerous other businesses and households creating an "economic multiplier effect". As only a portion of each dollar that is earned is spent, the amount of money from a particular transaction that continues on in the national economy tends to get smaller. If the linkages between economic sectors are known or estimable, the total economic impact of earnings or spending in a particular market can be estimated. These impacts are commonly partitioned into direct effects, indirect effects and induced effects. *Direct effects* are market contributions to the national economy, and are typically measured by gross total revenues, total employment or gross incomes. *Indirect effects* are impacts on the incomes and wages of the suppliers of inputs used in the industry in question when those earnings are subsequently spent on other goods and services. Finally, *induced effects* are the economic impacts of spending of generated income by households who are either directly or indirectly employed in the industry. Indirect and induced effects taken together are often referred to as *value added* effects (Fedler, 2010).

The estimation of these effects is typically facilitated by input-output models, which empirically delineate forward and backward linkages between a particular economic sector and the rest of the economy. These linkages form the basis for *multipliers* which measure the extent to which a given economic activity (direct effect) propagates other economic activity (Schuhmann et al., 2010). Value added effects can be represented with a multiplier that converts direct expenditures total economic impact (Fedler, 2010). For example, if the estimated value added multiplier for tourism is 1.5, then each $1 of direct spending by tourists results in an additional $1.50 of indirect and induced effects, for a total of $2.50 in economic impact. For more details on input-output analysis see Miller and Blair (1985). Economic impact analysis and the use of input-output models should not be considered a substitute for the calculation of total economic value or economic surplus, as net gains to market participants are not estimated, and non-market values are excluded (Hoagland, et al, 2005).

2.5 Guiding valuation in the CLME: Appropriate methods for different categories of values

As noted above, the valuation method chosen often depends on what is being valued as well as the intended purpose of value estimates. Table 3 presents sources of benefits from ecosystem goods and services as well as an assessment of the ease of valuation and recommended valuation procedures. Generally speaking, if the intended purpose of value estimates is to inform specific resource management decisions at the local or national level (e.g. whether to close an area to fishing, whether or not to permit development of a coastal area), then the valuation study should be carefully directed so as to permit estimation of the costs and benefits of the proposed change relative to the status quo. If the purpose of valuation is broader, such as to call attention to otherwise unrecognized values, then valuation exercise need not be as precisely targeted toward specific changes from the status quo.

It is important to note that the benefits from a particular ecosystem service may be amenable to valuation using different procedures,[2] and opinions about which method is appropriate may vary according to the disciplinary background of the researcher.[3] As noted by the U.S. Environmental Protection Agency's Scientific Advisory Board in its recent report on valuation (EPA, 2009, page 16):

> *"Although there is not a one-to-one mapping between valuation methods and the concepts of value …, often different views about the appropriate role of alternative valuation methods stem from different views about the nature of value or the appropriate concept of value to apply in a given context. Researchers with different disciplinary backgrounds (e.g., economics, psychology, ecology, decision science) often adopt a particular concept of value and work primarily with and advocate a specific method or set of methods designed to measure that concept".*

Hence, while it would in some ways be easier if there was a standard and commonly accepted methodology for all valuation scenarios, the inherent flexibility of the methods allows for adaptation to multiple situations and creates a good deal of overlap. That different measures may produce different estimates of value for the same resource should not be taken as an indictment of valuation, but rather as a reflection of the notion that value is case-specific, context dependent and particular to individual assessments of well-being. When feasible and appropriate, valuations that employ different methods may be especially useful to researchers as checks on validity and to policy makers in terms to provide potential bounds on value.

2.5.1 Extractive uses and services tied to markets

More than one valuation approach is often necessary to understand the benefits derived from a particular good or service, in total or at the margin. Reef ecosystems, for example, provide an array of provisioning, regulating, supporting and cultural services, which in turn generate myriad values to people. Market-based approaches may be appropriate for valuation of provisioning services, goods obtained via extraction such as commercial fishery harvests, or in situations where estimates of avoided costs or replacement costs are needed. These values are likely to be

[2] Indeed, there is an extensive literature regarding the comparison and combination of stated and revealed preference approaches (see for example, Adamowicz, et al., 1994).
[3] The author of this report is not immune to such predispositions and represents an obvious case in point.

the easiest and cheapest to derive, but may have the least policy relevance for conservation decisions involving non-market goods and services. When using market-based approaches, analysts should be careful to differentiate between market value (obtained as the product of market prices and quantities), net economic benefits, which include estimates of producer and/or consumer surplus, and economic impacts. The potential for extra-market values (e.g. food security or cultural values) should also be considered when estimating the value of extractive uses.[4]

2.5.2 Marine ecosystem services

Valuation of benefits from supportive and regulating ecosystem services (e.g. benefits derived from coral reef ecosystems such as biodiversity provision, habitat, water quality and coastal protection) and are best obtained using the production function approach, which often requires an understanding of bio-physical relationships between natural resource inputs and measurable benefits that can subsequently be incorporated into other valuation methods (e.g. market price or HP). For valuation of goods and services associated with recreation and/or tourism, arguably the most important economic driver in the WCR, TCM is most often appropriate, but a CM application may be preferable in the case of composite goods or complex environmental changes. A combination of these two approaches can easily be accomplished using a single survey instrument, allowing for the hypothetical nature of stated preference CM estimates to be grounded in observable (revealed preference) TCM data. CM is also amenable to valuation of proposed policy changes or states of the world, and may be a useful tool for understanding tradeoffs that user groups are willing to make with regard to an array of environmental goods and services.

2.5.3 Non-use values

For estimation of non-use values to tourists and/or locals (e.g. willingness to pay to preserve endangered marine species, ecosystems or areas, or willingness to pay for preservation of the culture associated with artisanal fisheries), CVM and CM are the only methods available. When an understanding of total economic value is of interest, estimates of non-use values should not be ignored; these values have been shown to be substantial components of total economic value and important inputs for shaping policy (e.g. Loomis and González-Cabán, 1999; Carson and Mitchell, 1993; ETI, 2007). While each of these approaches has its merits, the general trend in the valuation literature over the past decade is toward the use of CM and away from CVM. Because CM allows for the valuation of several environmental characteristics with a single design, and it's flexibility in terms of being able to address use and non-use values, it may soon be recognized as the state of the art methodology for non-market valuation.

[4] For more details see Chiwaula and Witt, 2010.

Table 3: Appropriate valuation methods

Source of benefits	Ease of valuation	Appropriate methods
Food	High	MP, PF
Raw materials	High	MP, PF
Medicinal resources	High	CA, RC, PF
Recreation	High	TCM, CM, CVM
Aesthetics	High	HP, CVM, TCM, CM
Species/ecosystem protection	Medium	CM, CVM, TCM
Nutrient regulation	Medium	CA, CVM
Biological regulation	Medium	RC, PF
Storm/erosion regulation	Medium	RC, CA
History, culture, traditions	Medium	CVM, CM
Genetic resources	Low	MP, CA
Climate regulation	Low	CVM, CM
Science, knowledge, education	Low	CVM, CM

Adapted from Farber et al.(2006), Abaza and Rietbergen-McCracken (1998) and WRI (2009)

2.6 Valuation difficulties

2.6.1 Marginality

Value estimates should most often be derived in the context of marginal changes to resource quality or ecosystem services. Care must also be taken not to use values derived in the context of incremental changes to value the ecosystem as a whole. First, value estimates are derived assuming all other things are held constant. Second, unit values associated with goods or services (market or non-market) will naturally change as a function of scarcity and the scale of measure. For example, the value of a 5% improvement in coral quality will be considerably larger if the starting point for improvement is 5% cover than if it is 35% cover. Further, the value of a 5% change in coral cover on all reefs cannot be derived by simply scaling up the value of a 5% change on one reef by some measure of total physical area (Bockstael et al., 2000). Hence understanding the relationship between values and the scale of analysis is critical before attempting to assign values to entire ecosystems or natural populations.

Moreover, aggregated values may be more difficult to incorporate into appraisals of the costs and benefits of policy action or inaction (Turner et al. 2003). That is, attempts to estimate the total value of a given ecosystem are unlikely to be useful in most policy contexts. Conservation decisions are most often directed at incremental changes to resource quality or ecosystem service flows, rather than absolute “all-or-nothing” changes. As noted in Bockstael et al. (2000), to value a complete ecosystem, we must be able to compare the state of the world with the system in place to a prediction of what the world would be like without the ecosystem. For large scale ecosystems such as the CLME upon which the existence of all Caribbean humanity depends, the notion of willingness to pay or accept compensation for loss of the entire system simply cannot be quantified in a finite fashion (Steiner, et al. 2004). Valuation studies should therefore attempt to match desired value targets with resource quality changes or service flows that would feasibly result from policy actions or inaction.

2.6.2 Double-counting

When benefits from ecosystems are non-complementary, summing estimates of individual components of value may lead to over estimates of total economic value. Turner et al (2003) provides the example of a wetland that cannot provide both recreation values and effluent storage and treatment. It would be incorrect to add estimates of these two functions in deriving the total value of the wetland. Similarly, if estimating the value of improved reef quality, it may be incorrect to assume that incremental gains to recreationists and fishers can be achieved simultaneously.

2.6.3 Distribution of costs and benefits

Valuation studies often produce aggregate or average measures of benefits for a group or population. It is important to note that because costs and benefits of changes in resource quality are often unequally distributed across populations, what appears to be a net gain to society may place undue burden on particular user groups. For example, a policy that improves the net welfare of tourists at the expense of local communities is may not be viewed as beneficial even if the benefits far outweigh the costs. Such effects are of particular concern when the cost-bearing group is relatively poor. Analysts should attempt to understand the distribution of costs and benefits across segments of society so that strategies can be designed to avoid or ameliorate such unfavorable outcomes (Steiner, et al. 2004).

Costs and benefits are also often unevenly distributed over time. Natural resource improvements that we might pay for today (perhaps through preservation efforts) could give us a stream of benefits that lasts into the future. Likewise, a development project might generate immediate financial rewards, but have long-lasting cost implications in terms of environmental quality. Comparison of values over different time periods is not a straightforward problem, and requires that we understand that a dollar today is worth more than a dollar tomorrow. The process of discounting, while mathematically straightforward, requires the choice of a discount rate which reflects preferences for future dollars relative to current dollars. The choice of discount rate is critical, and may determine whether or not a given project has an acceptable benefit-cost ratio. Higher (lower) values of the discount rate will yield lower (higher) NPV of future values. For public policy decisions or decisions regarding the use of public assets, this rate should reflect society's opportunity cost of funds. Understanding that society is generally risk averse, especially with regard to expenditures of public funds, suggests the use of low discount rates. However, the relatively low incomes of nations in the CLME suggest the use of higher discount rates. It is recommended that NPV calculations be made using a reasonable range of discount rates so that decision makers understand the sensitivity of benefits estimates to the value of this parameter.

2.6.4 Nonlinearities in ecosystem service provision

Ecosystem service provision varies naturally over time and space and depends on the size and status of proximate species and habitats (Barbier et al. (2008, Koch et al. 2009). For example, the degree of coastal protection provided by a coral reef ecosystem depends on season, tide, the extent and composition of the reef, as well as the presence or absence of other near shore habitats such as mangroves and sea grass beds. Such nonlinearities in the provision of ecosystem services may result in inappropriate estimates of value if services are assumed to be linear or static. As such, attempts to value ecosystem services should incorporate such nonlinear relationships may be flawed, resulting in inappropriate management actions. Koch et al. (2009) provide important discussion of this issue and associated recommendations for valuation in the context of ecosystem-based management.

2.6.5 Aggregation

One of the main issues inherent in economic valuation studies is aggregating individual measures of value or willingness to pay (WTP) to the level of a population. In order to aggregate, several issues must be confronted, including defining the relevant population for the good (i.e. who has standing) and determining what measure of value to apply to that population (e.g. mean, median). The former issue is clouded by the notion of non-use values, which may apply to anyone, regardless of the physical location of the resource.

In the case of commercial and recreational uses of natural resources, it is common to apply and individual benefit measure to an estimate of the relevant population (e.g. number of fishers or divers). While straightforward, this type of summation ignores any changes in participation that occur due to marginal changes in the quality of the resource being valued. For example, if we estimate that a 10% increase in stock size will increase commercial fisher revenues by $1,000/year and there are 100 fishers, we might surmise that aggregate gains are $100,000, but this ignores the changes in effort that may be induced by the 10% increase in stock size. To account for changes in use, participation functions can be estimated (e.g. Bockstael et al. 1987, Parsons and Kealy 1995, Schuhmann and Easley 2000). Simple multiplication of individual estimates of value by population estimates also ignores the notion that individual values are likely a function of geographic distance from the resource. To account for such effects, the mathematical relationship between distance and value (so called "distance-decay functions") can be estimated (e.g. Hanley et al. 2003). Such functions may be useful in determining the relevant population (i.e. at what distance does value reach zero?) but may not be applicable to non-use values (Hanley et al. 2003).

It is also important to note that because individual values are highly dependent on the relative scarcity of the resource and availability of substitutes for the resource, distance decay relationships may not transfer to other areas or other resources within an area. In other words, because value is largely context-specific (Turner et al., 2003), our ability to aggregate site-specific resource values geographical and cultural space is limited, and many value estimates will not be amenable to reasonable aggregation from local to regional scales (Jin et al., 2003).

3 Review of marine valuation studies in the Caribbean

3.1 Overview

While the extant valuation literature contains analysis of most of the world's ecosystems and numerous species, applications to aquatic and marine goods and services have garnered the least attention (Turner et al., 2003). Because valuation studies are costly, time consuming and often require advanced econometric skills and training, valuation is more commonly applied in the setting of industrialized nations. Marine resource valuations in the Caribbean tend to focus on more obvious and measurable benefits, primarily those derived from coastal and reef ecosystems as associated with marine protected areas, recreation and tourism. Analyses of benefits from pelagic and continental shelf ecosystems are much less apparent in the literature. This is perhaps due to a perceived disconnect between these offshore systems and tourism, but is also no doubt in part due to the relative dearth of scientific knowledge of these systems and the consequent empirical difficulties associated with monetizing their benefits to people. Notable exceptions to this generalization include estimates of the commercial (market) value of pelagic and deepwater fisheries, which are readily available for most countries from FAO and/or national fisheries offices, and a select few studies that examine economic values associated with offshore bill fishing and whale watching. Information regarding the economics of mineral deposits and energy resources of continental shelf is available (e.g. U.N., 2004), but is not specific to the Caribbean Sea. Attempts to value benefits from the supportive and regulating ecosystem services of offshore ecosystems of the Caribbean Sea or benefits from genetic materials contained in these ecosystems appear to be nonexistent.

In terms of nations within the Caribbean Region, Jamaica has received the most attention by resource economists, and is essentially in a league of its own with regard to the both number and breadth of valuation studies. Much of this work was conducted as part of a World Bank project in the late 1990s, and includes estimates of the economic value of Montego Bay coral reefs associated with coastal protection, artisanal fisheries and tourism/recreation (Gustavson, 1998), non-use benefits of the Montego Bay Marine Park (Spash et al., 1998), and the NPV of biodiversity for marine bioprospecting (Ruitenbeek and Cartier, 1999). Bonaire and Curacao have each been the setting for numerous valuation studies, the majority of which are in the context of reefs within MPAs. Barbados, Belize, Costa Rica, St. Lucia and Tobago also have been the setting for multiple valuation studies, many of them in recent years. Counts of existing value estimates in each Caribbean nation are listed in Table 4, categorized by type of value.

3.2 Replicable applications in the Caribbean

While there have been a few attempts to value region-wide ecosystem services (e.g. Cesar et al., 2003 estimate the NPV of coral reefs from fisheries, coastal protection, tourism/recreation and biodiversity), WRI's *Coastal Capital* series and OAS's *Reefix* (based in part on a methodology developed by WRI) represent the only attempts to apply common methodologies to numerous countries in the Caribbean. Like numerous other valuation studies in the Caribbean, these efforts employ methodologies that do not require a great deal of primary data collection (e.g. benefits transfer) or are based on convenience sampling schemes. By relying on secondary data and relatively non-technical methods, these approaches provide a replicable and low-cost approach to valuation. Coupling these characteristics with precedence in the region, these approaches may be more amenable to acceptance and implementation by policy makers in the CLME. As noted in the UNDP status report on Biodiversity and Ecosystems in Latin America and the Caribbean (2010):

> "… *past studies have shown limited transferable messages that are all too often site specific and use numerous valuation methodologies. That no common framework for economic valuation exists hinders its uptake by political leaders*."

While this perspective has merit, it should also be recognized that these relatively low-cost methodologies are not conventional approaches to valuation in the sense of understanding willingness-to-pay or economic gains associated with incremental changes in resource quality or ecosystem service flows. This is not to say that they do not provide important contributions to the body of knowledge, or that attempts to incorporate the gross market value of natural resources into national accounts are unfounded. Indeed, lack of recognition by policy makers of the economic value of natural assets most certainly leads to inefficient resource allocations leaving society worse off (TEEB 2009). Estimation of market or near-market values can form a powerful case for conservation (Turner et al, 2003). WRI's *Coastal Capital* series has undoubtedly done more than any other single effort in this regard, by calling attention to previously unrecognized economic values associated with coastal and marine resources in the Caribbean.

Yet, the desire to adopt common and transferrable methodologies ignores many important aspects of valuation, including the need to address different components of economic value with different approaches, the inherently case-specific nature of natural resource value and the reality of marginal resource change. Moreover, that these methods necessarily rely on secondary market data virtually ensures that the significant components of value associated with non-market uses and passive uses will be omitted and potentially ignored. In short, while "one-size-fits-all" approaches to valuation are likely to send important signals regarding the economic importance of natural resources, they may grossly underestimate true economic value and may not be sufficient for informing policy.

There is ample evidence to support these notions. Numerous studies have successfully applied multiple methodologies to a single ecosystem (e.g. Cabrera, et al. 1998 for mangroves in Mexico; Cartier and Ruitenbeek, 1999 and Gustavson, 1998 for reefs in Montego Bay, Jamaica; ETI, 2007 for reefs in Puerto Rico; van Beukering et al. 2009 for reefs in Bermuda). These large multi-disciplinary efforts generate a holistic perspective on the value of ecosystems, as well as the relative sizes of different elements of economic value. Pointing to the case-specific nature of value, tourism and recreation values appear to exceed those associated with coastal protection by up to a factor of 10 in Jamaica (Cartier and Ruitenbeek, 1999), but may be 100 times greater in Puerto Rico (ETI, 2007), while Cesar et al. (2003), report average coastal protection and tourism values that are approximately equivalent for the region as a whole.

3.3 Tourism and marine resources

Tourism in the Caribbean accounts for significant shares of national GDPs, employment and foreign exchange (Tsounta, 2008, Griffith, 2009). Total visitor arrivals have exceeded 42 million in recent years (CTO, 2009).[5] Annual visitor spending is estimated to be in excess of US $27 billion (Griffith, 2009). Based on tourism's total share of GDP, capital investment and total exports, the Caribbean is the most tourism-dependent region in the world (WTTC, 2011). Numerous studies analyze the complex relationships between tourism, natural resources and Caribbean economies (e.g. Holder, 1988; Beekhuis, 1981; Burke and Maidens, 2004). One could realistically argue that nearly all economic analyses and valuation studies in the Caribbean are at least indirectly related to tourism. Because tourism is the principle economic driver in the

[55] Comprised of roughly half stay-over and half cruise ship arrivals.

Caribbean, it is not surprising that many valuation studies relying on primary data use tourists as their subjects.

Carr and Heyman (2009) provide an excellent overview as well as a wealth of detailed information on numerous economic statistics associated with tourism, commercial fisheries and marine resources (including reefs and MPAs) in Antigua and Barbuda, Jamaica and the Caribbean region as a whole. Based on a comparison of tourism and commercial fisheries in Antigua and Barbuda relative to Jamaica and other OECS member states, the authors conclude that Antigua and Barbuda is at a crossroads, in danger of overexploiting its marine resources, particularly its fish and coral reef communities.

Dixon et al. (2001) provide an overview of issues related to sustainable use of environmental resources in the context of tourism, as well as opportunities and threats related to tourism and the environment. The authors suggest incentive-based policy actions to generate revenues that can be directed toward conservation, including capturing economic rents via user fees, taxation and investment incentives. Specifically, the authors recommend modest taxation of tourists via a combination of indirect taxes on goods used primarily by tourists (e.g. hotel services) and entry or departure levies. They note that such taxes may be the most effective means of rent capture provided that tourists understand that the purpose of these levies is for environmental preservation. Protected areas entrance fees are also recommended as a source of income generation and means of limiting potential over-use.

Uyarra (2002) and Uyarra et al. (2005) use principle components analysis to establish empirical links between environmental quality and tourism. Based on a convenience sample of 316 tourists in Bonaire and 338 tourists in Barbados, the authors illustrate the correlation between 16 environmental attributes and holiday destination choice. The willingness of tourists to return to the destination at the same price and lower levels of environmental quality is also measured, and used to suggest the potential impact of climate change on tourism. The authors find that all environmental attributes had positive effects on holiday enjoyment. Tourists in Bonaire have stronger preferences for marine environmental quality while tourists in Barbados tend to have stronger preferences for terrestrial environmental features, especially those relating to beaches. Notable exceptions include the higher importance of clear water and the presence of sea turtles in Barbados relative to Bonaire. A large majority (80%) of respondents indicated that they would be unwilling to return for the same price if the quality of their preferred environmental features were diminished (via coral bleaching in Bonaire and diminished beach size in Barbados), suggesting potentially devastating economic impacts from climate change.[6]

3.4 Valuations of the Coral Reef Ecosystem

In the area of coastal and marine resource valuation, beaches and reefs have received the most attention to date. Brander et al. (2006) provides an overview and meta-analysis employing 166 worldwide reef valuation studies. Chong et al. (2003) conduct a meta analysis using 25 reef valuation studies covering 20 sites and 7 countries in the Caribbean. Cesar (2000) and Gustavson et al. (2000) provide collections of articles on coral reef valuation. Conservation International.

[6] Details of the survey methodology are not provided by the authors, and it is questionable whether this work is based on a random sample and therefore representative of the true tourist populations. It appears that scuba divers and tourists from the U.K. are over-represented in Barbados sub-sample.

(2008) provides a global compilation of values for reefs (as well as mangroves and seagrasses) including several summaries from the Caribbean.

Vergara et al. (2008) investigate the economic consequences of coral mortality in the Caribbean due to the effects of warm seas and severe high-temperature episodes that may result from climate change. Specifically, assuming a temperature increase of 2ºC as a response to a doubling of CO2 concentration over pre-industrial levels, the authors estimate economic losses associated with 50 percent coral reef mortality by 2040 and 90 percent mortality by 2060. Presumably using benefits transfer from estimated global reef values, direct economic losses associated with fisheries, tourism activity, coastal protection and pharmaceutical uses are estimated to be between US$4.83 billion and US$6.6 billion for 50% coral mortality and between US$ 8.68 billion and 11.98 billion for 90% mortality. Notably, the loss of pharmaceutical values comprises between 55 and 75% of estimated losses, suggesting that reef value estimates were likely transferred from less tourism-dependent areas.

3.4.1 Estimates of multiple values from reef ecosystems

As noted in section 2.6.1 above, value estimates should be derived in the context of marginal changes to resource quality or ecosystem services, and are derived holding other factors and changes constant. Valuations typically only produce estimates for the most readily observed or measured components of human well being. Hence, estimates of the "total economic value" of ecosystems are likely mislabeled. Declarations of total value may carry considerable weight and garner attention, hence their use, while rare, is understandable. Several studies use multiple methodologies to estimate different components of total economic value.

Cesar et al. (2003) employs benefits transfer from applications of the market price method and cost-avoidance methods (principally Burke et al., 2002) to estimate annual net benefit streams and net present value (NPV) of coral reefs from fisheries, coastal protection, tourism/recreation and biodiversity in the Caribbean Region. The 19,000 km^2 of reefs in the Caribbean are reported to provide annual net benefits of US$391 million from fisheries, US$720 million from coastal protection, US$663 million from tourism/recreation and US$79 million from biodiversity value, for total annual net benefits of US$1.85 billion, or a net present value of US$49.5 billion (assuming a 3% discount rate).[7]

Ruitenbeek and Cartier (1999), Gustavson (1998) and Gustavson (2002) report estimates of the net present value (NPV) of coral reefs to tourism and recreation, fisheries, coastal protection and bioprospecting in Montego Bay, Jamaica. Readily identified direct local uses of the reef (near shore fisheries and tourism services) were valued using PF approaches, whereby costs of operations are subtracted from total revenues taken in through the use of the coral reefs.[8] NPV estimates for tourism in Montego Bay ranged from US$210 million to US$630 million, depending on the discount rate. The estimated NPV for nearshore fisheries is US$1.31 million. The value of coral reefs for coastal protection was estimated by calculating the total value of land and property that is at risk to loss from erosion along the shoreline within the bounds of the

[7] The reliability of these estimates is difficult to ascertain due to the lack of detail provided in this report or in the referenced report.

[8] Similar to a market price approach, the resulting values can be considered an estimate of producer surplus or net gains to service providers.

Montego Bay Marine Park. The NPV of this land and property that is assumed to be protected by the reef is estimated to be US$65 million. Summing these three values yields a total NPV of US$381 million, or US$8.93 million per hectare of reef. Non-use values associated with the reef are also estimated using CVM.[9] The estimated NPV of the (reef) biodiversity in Montego Bay Marine Park is estimated to be US$13.6 million to tourists and US$6.0 million to residents. Pharmaceutical bioprospecting values are estimated using cost estimates, discovery hit rates and end-use values. Marine bioprospecting values are estimated to be approximately $70 million for the Montego Bay reefs, of which roughly $7 million in value could be captured via royalty regimes or rental agreements.[10]

Using similar methods (as part of WRIs Coastal Capital Project) Burke et al. (2008a) and Cooper et al. (2009) use the MP approach to estimate the economic value of reef ecosystems to fisheries and tourism in Tobago, St. Lucia and Belize[11] and apply the CA approach to value coastal protection services. In Tobago, where the authors assume that 40% of tourist visits are attributable to coral reefs, 2006 direct economic impacts from visitor spending are estimated to be US$43.5 million, with an additional indirect impacts of US$58 to US$86 million. In St. Lucia, 25% of tourist visits are assumed to be attributed to reefs, generating US$91.6 million in direct effects and US$68 to US$102 million in indirect effects. The total direct economic impact from reef and mangrove associated tourism in Belize is estimated to be between approximately US$150 and US$196 million.[12] Revenues from reef (and mangrove) associated fisheries production and processing are estimated to be between US$14.2 and US$15.9 million per year in Belize, with over 70% attributable to exports. Economic impacts from reef fisheries in Tobago and St. Lucia are considerably smaller, estimated to be between US$ 0.7 and US$1.1 million for Tobago and between US$0.4 and US$0.7 million for St. Lucia. The estimated value of coastal real estate that is protected by coral reefs is estimated to be between US$18 and US$33 million in Tobago, between US$28 and US$50 million in St. Lucia, and between US$120 and US$180 million in Belize.

Carleton and Lawrence (2005) attempt to estimate values associated with environmental resource services in the Turks and Caicos, including the amount of tourist spending attributable to coral reefs, fisheries values from reefs and the value of coral reefs for coastal erosion protection. The value of coral reef biodiversity is estimated at $4.7 million per year, reef contribution to coastal protection is valued at $16.9 million per year, and reef fisheries are valued at $3.7 million per year. Diving on coral reefs is estimated to be worth $8.3 million per year and other forms of tourism supported by reefs are estimated to be worth at least $9.8 million per year in Gross Value Added and consumer surplus. Total value of reefs in TCI is estimated to be $47.3 million per year, of which $17.7 million per year contributes directly into GDP.[13]

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[9] The CVM survey was administered to a sample of 1058 locals and tourists. Average WTP for coral reef improvement is reported as US$3.24 per person.

[10] Bioprospecting values are estimated to be $7,775 per species, or $2600 per sample with a typical success rate of 1 in 30,000 samples. These estimates translate to $530,000/ha or $225,000/% coral abundance.

[11] In Belize the valuation pertains to both reef and mangrove ecosystems.

[12] The authors suggest that additional indirect impacts may be between US$26 and US$69 million per year.

[13] While this work contains some excellent discussion regarding the importance of valuation, the valuation methodologies employed differ markedly from accepted practices and appear to be based largely on unjustified

3.4.2 Reef recreation and tourism not specific to MPAs

While the majority of valuation studies related to reef-based recreation in the Caribbean pertain to recreation that takes place in marine protected areas, there are a few general reef recreation studies that warrant mention. A review by Brader et al (2006) shows that the world average value of coral reef recreation is $US 184.00 per visit (2000 dollars), with considerable variation across location, method and goods and services. Coral reef recreation in the Caribbean has the highest mean value of all areas analyzed, roughly US$400.00 per visit. Of all recreational activities analyzed, diving and snorkeling produce the highest values, typically over $200 depending on location. Although this level of detail is not provided by the authors, we can infer that diving and snorkeling on reefs in the Caribbean will produce significantly large economic values.

3.4.2.1 Scuba diving and snorkeling

Divers' willingness to pay for coral quality improvements in Barbados estimated by Schuhmann et al (2011) range from US$29 to US$195, depending on the baseline level of quality. These values are strikingly similar to those found by Parsons and Thur (2008) who valued changes in coral cover in Bonaire National Marine Park. In this latter study, potential aggregate annual losses from degradations in coral quality are estimated to range from US$0.8 million to $5.2 million per year, which is equivalent to a NPV of losses ranging from US$27 million to $173.4 million. Casey et al. (2010) estimated the willingness of tourists in the Riviera Maya to contribute to a coral trust. Schuhmann et al (2011) show that divers' willingness to pay for high levels of coral cover may be as high as US$195 per two-tank dive.

Beharry-Borg and Scarpa (2010) use a CM approach to estimate Willingness to pay for coastal and marine attributes by snorkelers and non-snorkelers in Tobago. Attributes in the CM include water quality (chance of ear infection), water clarity (vertical visibility), coastline development, marine protected areas, fish abundance, coral cover, beach litter (plastics) and number of boats. Snorkelers are willing to pay up to $35.00 per trip for high fish diversity (up to 60 fishes), up to $50.00 for high coral cover (45%), $40.00 for vertical visibility up to 10 meters, and 22.00 for water quality that allows for a low chance of an ear infection. A high degree of heterogeneity in preferences and WTP is discovered across and between groups.

3.4.2.2 General reef-based tourism and recreation

van Beukering et al. (2009) and Bermuda Department of Conservation Services (2009) estimate the value of reef-associated tourism in Bermuda using data from a reef-associated tourist operator survey (revenue data) and a tourist exit survey which assessed the importance of coral reefs to the visitation experience. Tourist exit survey data supported an application of the TCM to estimate consumer surplus, while reef operator data was used to estimate producer surplus via the net factor income method.[14] CVM was also applied to estimate WTP for coral reef conservation. Separate results are provided for cruise ship and air tourism. In addition to value

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assumptions. For example, coastal protection values are estimated using assumptions about erosion rates and potential damage from hurricanes, absent any details regarding the underlying scientific basis. Land and property values are derived from real estate magazines, presumably from listing prices which likely overstate true market values. Reef values associated with tourism are estimated using assumptions about the percentage of visitors' motivations that are attributable to reefs, which is based on the number of reef pictures in advertisements.

[14] Similar to the market price method, costs of production were subtracted from reported tourism expenditures.

estimates, the authors report that 68% of all tourists reported being willing to pay additional funds to support activities that preserve Bermuda's coral reefs. The average cruise ship tourist is willing to pay US$28 per visit to Bermuda and the average airplane tourist is willing to pay US$19. The total annual value of coral reef-associated tourism is estimated to be US$ 409 million.

Recreational and cultural values associated with reefs are also estimated by van Beukering et al. (2009) via an application of the CM approach to a survey of 400 Bermudian households. Attributes in the CM were recreational fishing, coral diversity/fish diversity, recreational activities (scuba diving/snorkeling), and water quality (described as coral diversity, fish diversity, water clarity, and swimming restrictions). The payment vehicle was an environmental levy. Households had an average WTP of US$ 42 per month for minimizing marine pollution (translated as the ability to swim anytime and anywhere) and US$ 32 per month per household for maintaining coral reef quality (i.e. coral and fish diversity). Average WTP for water clarity (maintained by a healthy coral reef system) was US$27 per month.[15]

Using market values of recreation and tourism activities combined with estimates of consumer surplus derived from an application of TCM, ETI (2007) reports that the value of reefs associated with recreation and tourism in Puerto Rico is US$ 939,776,410, the majority of which is attributable to beach uses and water sports (not including snorkeling). This value is slightly more than half of the total value ascribed to Puerto Rico reefs in this study. Hargreaves-Allen (2010b) provides an estimate of the economic impact of nature-based tourism on the economy of the Bahamas. Although not specifically attributed to coral reefs, we can assume that a significant percentage of the estimated $44 million in annual economic impacts are linked to reef-based recreation.[16]

3.4.2.3 Species-specific values associated with reef-based recreation

Diver preferences and willingness to pay for reef-related attributes such as fish abundance, coral cover and encounters with specific species have been examined at select locations in the Caribbean. Rudd (2001) finds that the presence of spiny lobster (*Panulirus argus*) have a significant impact on the market share of various dive charter packages in the Turks and Caicos, indicating that spiny lobsters have non-extractive use value. Rudd and Tupper (2002) and Rudd, et al (2001) find evidence that the market share of dive profiles in the Turks and Caicos increases with Nassau grouper size and abundance. Viewing reef sharks and sea turtles also had a large impact on market share in the simulations. Schuhmann et al. (2011) use a CM approach to show that divers may be willing to pay up to US$145 for encounters with marine turtles in Barbados. These results indicate the ability to charge higher prices or MPA admission fees if divers are taken to sites with a higher probability of encountering these species. Rudd et al. (2001) suggest that increased revenues from such fees may be enough to cover both the explicit costs of MPA expansion and the opportunity costs to local fishers associated with lost fishing opportunities.

[15] Less than half of respondents indicated that they would be willing to pay an environmental levy, suggesting a high degree of protest zeros or non-response in this study.

[16] Although the valuation methodology is not clearly articulated, we can assume that tourism values are derived using something akin to the market price method, and as such may be more reliable than the benefits transfer estimates detailed in the report.

3.5 Marine protected areas

The majority of valuation studies associated with MPAs in the Caribbean involve estimates of WTP associated with entrance fees and/or improvements in the quality of non-extractive uses such as recreational scuba diving. Exceptions include general valuations and financial analyses. An interesting issue with regard to the valuation of marine parks and protected areas is whether it is the existence of a park itself that is valued or if is simply the quality of the attributes and the features of the natural environment that are directly related to the management of the park.[17]

3.5.1 General valuations of MPAs

Beharry-Borg and Scarpa (2010) administered a CM survey to a sample of 284 locals and tourists in Tobago to understand WTP for amenities related to coastal waters. One of the attributes included in the CM was the presence of marine protected areas. Estimated WTP for snorkelers was up to $34.00 for the presence of an MPA. Interestingly, the authors found is little difference in WTP for the presence of an MPA based on whether or not the MPA allows fishing. In contrast, in an examination of MPAs in BVI, Saba, Guadeloupe and St. Lucia, van't Hof (1998) finds that respondents generally did not indicate that the existence of the MPA itself was an important factor in their decisions to visit the area or engage in various recreational activities in the area, and that if the same level of environmental quality could be achieved without a formal MPA, visitation and activity would likely be unaffected.

Spash (2000) and Spash et al. (2000) use CVM in Jamaica and Curacao to estimate WTP for marine biodiversity as provided by marine protected areas. Respondents were questioned about willingness to pay into a trust fund to create a marine park that would improve biodiversity by 25% within the boundaries of the park.[18] The baseline, "no management" scenario was described as a 15% reduction in biodiversity. In addition to the CVM scenario, locals and tourists were asked to indicate the sources of value derived from marine parks, expressing categories as a percentage of total value. The highest percentages of value were from swimming (roughly 33% on average), and from diving and snorkeling (12.5%). As might be expected, tourists tended to associated higher value from recreational and aesthetic uses (swimming, sunbathing), while locals tended to assign higher values to extractive uses (fishing and seafood). Surprisingly, 28% of Jamaican locals assigned no benefits to the existence of a marine park. Average willingness to pay was roughly US$25 per person per year, and was relatively constant across the two nations and between locals and tourists. Willingness to pay was found to be significantly related to age, education, knowledge of marine biodiversity, the number of sources of value derived from marine parks, and a sense of duty to protect marine life and habitats.[19]

[17] Pendleton (1995) notes that the economic benefit from marine protection areas should be measured as the avoided losses in reef value that would be incurred in the absence of the park, net of costs for protection.

[18] The WTP question was open-ended and respondents were told that payment would be annual for a period of five years.

[19] A potential issue with this application is that many respondents were unfamiliar with the term biodiversity (62% and 63.5% in Jamaica and Curacao respectively), hence may not have appreciated what they were valuing. This information bias may render the results from unaware respondents less reliable. Further, each sample contained a significant portion of zero bids and protest bids (27% and 32% in Jamaica and Curacao respectively), which biases downward the WTP values. Aggregation and benefits transfer were not attempted for a variety of reasons detailed in

Raboteur and Rodes (2006) use CVM to estimate WTP for the creation of a nature reserve at Pigeon Malendure to protect coral reefs. Based on a sample of 100 individuals average household annual willingness to pay ranges from approximately US$10.50 to US$15.00 per household (2003).

3.5.2 Financial analysis of MPAs

Geoghegan (1998) summarizes experiences with protected area financing in the Caribbean, and provides guidelines for designing revenue generation strategies and a framework for selection of appropriate funding mechanisms. Success stories are detailed in four case studies, which include some estimates of revenues and costs. Case studies include: Nelson's Dockyard National Park, Antigua, the British Virgin Islands Reef Conservation Fee, Saba Marine Park, Netherlands Antilles, and Pigeon Island National Historic Park, St. Lucia.

Examples of financial analysis of specific parks include Woodfield (1997), who estimates net income from Wreck of the Rhone Marine Park (WRMP) in the British Virgin Islands (BVI) using the market price method. National parks in the BVI require the use of mooring buoys, which have an associated "conservation fee" that is used by the National Parks Trust for reef protection schemes. This fee is the primary source of revenue for the WRMP. Costs associated with the program include payroll, boat maintenance and fuel. Total income is estimated to be $179,478.26. Cost estimates of $82,768.59 yield an overall net income of $96,709.67 associated with the mooring buoy program. Buchan et al. (unknown date) examines income, expenses, net income and income/expense ratios by income source for Saba Marine Park, NA, [20] which is estimated to produce US$1.9 million in income for the national economy, over 86% of which comes from dive tourism. Economic impacts of the park are estimated to sum to approximately US$3.05 million, representing roughly 21% of the national economy.[21]

3.5.3 WTP and Recreation in Marine Protected Areas

Numerous valuation studies have been conducted in the Caribbean to analyze diver and/or snorkeler preferences and willingness to pay for different aspects of marine quality in the context of access to marine parks and protected areas. The general conclusion of these studies is that vacationing recreationists have significant WTP for access to marine protected areas provided that threshold levels of quality and limits on use are maintained. Green and Donnelly (2003) investigate the use of MPAs by scuba diving operators in the Wider Caribbean and Pacific coast of Central America and show that the capacity to generate revenue from Scuba diving user fees in this region has not yet been fully exploited. Despite the fact that half of all dives throughout the region take place within a MPA (approximately 7.5 million dives), only a minority of MPAs charge fees for use. Depondta and Green (2006) also note this phenomenon, indicating that only 25% of Caribbean MPAs containing coral reefs charge divers entry or user fees. Perhaps of equal

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the manuscript. This study employed appropriate CVM techniques though estimates of value are likely downwardly biased.

[20] Estimates of income/expense ratios show values greater than one for all tourism uses of the park except yacht tourists. Local uses of the park appear to have negative net incomes, and as such cost the park money.

[21] A multiplier of 1.6 is used to estimate economic impacts of the park. If other tourism is attributed to the park, the park may be responsible for nearly half of the islands total GDP.

importance, those few MPAs that do charge fees, charge amounts well below estimated diver willingness to pay.

Terk and Knowlton (2008) provide a more recent review of user fees for diving in Caribbean MPAs, including a detailed list of MPAs in the Caribbean containing coral reefs. The presence of fees, management type and fee structure are noted for each park. The authors find that diving fees are levied in only 16 of the 38 countries and territories and in only 34 of the 194 identified MPAs, despite the fact that 82% of the MPAs in the region protect coral reefs.[22]

WTP values and consumer surplus estimates are often compared to park maintenance expenditures to make a case for tourist taxes as a source of park funding. It is apparent from the literature that most Caribbean MPAs are underfunded and that user fees can serve as a source of revenue, often more than sufficient to cover operating expenses and park management. For example, Thur (2010) estimates divers' willingness to pay for access to the Bonaire National Marine Park (BNMP) using CVM. Results suggest that divers are willing to pay significantly more than the existing $10 annual user fee for access to the park. DaCosta (2010) uses CVM to estimate WTP to enter Buccoo Reef Marine Park, Tobago and finds that even nominal fees of US$3-$4 would result in revenues that are significantly larger than current budget allotments for park maintenance, suggesting that user-based financing is highly plausible. Woodfield (1997) finds that quality of the environment and marine life was indicated as "very important" for 86% of visitors to the Wreck of the Rhone Marine Park, and that 59% of visitors indicated a willingness to spend 10-50% more on overall expenses associated with the park. van't Hof (1998) reports similar results for other Caribbean parks including the Saba Marine Park in the Netherlands Antilles, the Reserve Ilets Pigeon in Guadeloupe, the Soufriere Marine Management Area in St. Lucia, and the Virgin Island National Park in the U.S. Virgin Islands. Weilgus et al (2010) use CVM to estimate diver WTP to enter the La Caleta National Marine Park in the Dominican Republic, and derive revenue-maximizing charges of US$52.70 for local divers and US$58.80 for international visitors. Assuming that operating costs per diver would be approximately US$10, the authors suggest that by curtailing fishing and focusing on tourism fishers could earn up to 90% of their fishing income from dive-related tourism alone.[23]

Several studies note that despite significant WTP, fees that maximize revenues may be suboptimal because high fees may result in adverse impacts on national economies through decreased tourism (e.g. Edwards, 2008, Thur, 2010, Planter and Piña, 2006, Dharmaratne, et al., 2000).[24] Importantly, these studies also show that nominal increases in fees can produce substantial increases in revenues without significantly decreasing overall tourism demand.[25]

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[22] Where they exist, the average dive fee paid per day for a two tank dive was found to be $US 1.15±3.49, roughly equivalent to the average price of a cup of coffee ($US 2.68±1.37).

[23] This is assuming a significant recovery of fish stocks which induces higher visitation rates by divers.

[24] Planter and Piña (2006) use a CVM (payment card) survey to estimate respondents' maximum willingness to pay to enter the Natural Protected Areas in Mexico's Caribbean coast. More than 85% of the tourists would still visit the protected area if a US$2 fee were imposed. Estimated socially optimal fees are found to be lower than revenue maximizing fees, generating more consumer surplus through higher visitation rates.

[25] For example, the $20 fee for the BNMP dive tag which was acceptable to 94% of divers would generate over US$500,000 in revenues, which is nearly double the budget of the park. As a result of this study, the price of a dive tag was increased to $25 in 2005.

MPAs in Jamaica have received considerable attention in the literature, with principal focus on the Montego Bay Marine Park (MBMP). Work by Ruitenbeek and Cartier (1999), Gustavson (1998) and Gustavson (2002) were detailed earlier in this report. Walling (1996) and Dharmaratne, et al. (2000) use CVM to estimate WTP by first time visitors and repeat visitors for use and non-use values associated with management of MBMP.[26] Spash (2000) and Spash et al. (2000)[27] use CVM to estimate WTP into a trust fund that would improve biodiversity by 25% within the boundaries of the park relative to a baseline, "no management" scenario of a 15% reduction in biodiversity. Average willingness to pay was roughly US$25 per person per year, with little difference in WTP between locals and tourists.[28]

Bunce and Gustavson (1998) and Bunce et al. (1999) detail the results of a rapid socioeconomic assessment of coral reef user groups (fishers, water sports operators and hoteliers) in the MBMP to ascertain social, cultural, and economic backgrounds, the type and nature of reef-related activities, perceptions of reef management and economic dependence on the reef.[29] The size of each user group, fishing effort by gear, and fisher average income (weekly and annual by gear) are also estimated.[30]

Reid-Grant and Bhat (2009) provide a review of funding sources for marine protected areas as well as empirical analysis of potential funding sources for the MBMP. WTP for park maintenance and conservation by stakeholders is analyzed using a combination of the TCM applied to cruise ship and air travel visitors, and CVM-like surveys administered to hoteliers and tourism-based businesses. Notably, despite considerable consumer surplus[31], nearly half of tourists indicated that they would not be willing to donate to the park. Further, while three out of five hoteliers stated willingness to donate to conservation efforts or were willing to provide a means of collecting donations from hotel guests, none were willing to donate funds directly to the MBMP. Only 38% of tourism-based business owners indicated willingness to donate to conservation of the MBMP despite the fact that a majority indicated that the park was important

[26] WTP by first time visitors is approximately double that of repeat visitors at roughly US$20 and US$10 respectively. Non-use values are found to be comparably small at less than US$2.

[27] This study was also carried out in Curaçao for a hypothetical marine park.

[28] Willingness to pay was found to be significantly related to age, education, knowledge of marine biodiversity, the number of sources of value derived from marine parks, and a sense of duty to protect marine life and habitats. A potential issue with this application is that many respondents were unfamiliar with the term biodiversity (62% and 63.5% in Jamaica and Curacao respectively), hence may not have appreciated what they were valuing. This information bias may render the results from unaware respondents less reliable. Further, each sample contained a significant portion of zero bids and protest bids (27% and 32% in Jamaica and Curacao respectively), which may result in downward bias of the WTP values. Aggregation and benefits transfer were not attempted for a variety of reasons detailed in the manuscript.

[29] Sample sizes are relatively small, at 35, 11 and 6 respectively. Interview data were supplemented with data from focus groups with fishers and water sports operators, as well as phone interviews with hoteliers, and field observation of user groups.

[30] Hotels account for the largest percentage of employment, with approximately 16,000 directly or indirectly related jobs. Fishing is estimated to directly employ roughly 380 individuals providing an annual net income between US$3,000 and US$4,500.[30] Watersports operations in Montego Bay account for approximately 200 jobs.

[31] Net gains above trip costs are estimated to be US$586 or US$739 per person per trip. Estimates of the total annual consumer surplus are US$189 and US$993 million for cruise travelers and air travelers respectively.

to their business and recognized that anthropogenic activities had an adverse effect on the quality of the park.

Numerous valuation studies have examined the Bonaire Marine Park (BMP), one of the few Caribbean MPAs that is entirely financed by user fees. By all accounts, valuation studies played an important role in establishing appropriate user fees which lead to self-financing of the BMP. Indeed, at the time of the first valuation study (reported in Dixon et al., 1993, and Dixon et al., 1995 and Dixon et al., 2000), there was no entrance fee for the BMP.[32] These initial studies provide estimates of the contribution of BMP to the local economy, the costs of maintenance of BMP[33] and divers' WTP entrance park fees, and estimate threshold levels of dives beyond which damage to reefs would likely be induced. Prior to the implementation of any fees, mean reservation prices (maximum WTP) for entrance into the park estimated via CVM were US$27.40 per visitor year,[34] and 92 percent of visitors agreed that a proposed $10 user fee was reasonable.

Pendleton (1995) uses a simple TCM based on estimates of the number of visitors to Bonaire and annual visits to the marine park and calculates total annual consumer surplus to all visitors to the Bonaire Marine Park as approximately US$19.2 million. Based on net profits that accrue to locally-owned reef-related businesses and taxes levied on foreign-owned reef-related businesses, net annual benefits associated with dive-related tourism are estimated to range between US$7.92 million and US$8.8 million.[35] Uyarra (2002) and Uyarra et al. (2005) report that a large majority of divers (97%) and non-divers (84%) in Bonaire show acceptance of the existing $10 entrance fee to BNMP, suggesting WTP values may exceed the fee, creating consumer surplus for most visitors.

Parsons and Thur (2008), use a CM survey administered to a sample of 211 U.S. Scuba divers who had visited Bonaire in 2001 to estimate the economic loss to SCUBA divers associated with hypothetical declines in the quality of the coral reef ecosystem in Bonaire National Marine Park. Declines in quality from the status quo level to three inferior levels[36] are estimated to range from US$45 to US$192. Aggregation suggests that the discounted present value associated with losses in quality ranges from US$42 million to US$179 million.

Thur (2010) estimates willingness to pay for access to the BNMP by recreational scuba divers using a CVM (payment card) survey administered to the same sample of 211 U.S. Scuba divers.

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[32] An annual fee of US$10/diver had been proposed but not yet implemented.

[33] Total gross revenues associated with dive tourism are estimated to be US$23.2 million in 1991, which is comprised of US$10.4 million from hotels, US$4.8 million to dive operations, US$4.7 million from other expenditures including non-hotel restaurants, souvenirs and car rentals, and US$3.3 million for air transport. Government revenues from room, casino and departure taxes levied on visiting divers totaled an additional US$340,000. Initial direct costs associated with the establishment and rehabilitation of the park were estimated to be approximately US$ 518,000. Annual operating costs were estimated to be US$150,000.

[34] This value is derived excluding zero responses. When zeros were included in the sample, mean WTP was US$24.1.

[35] Data are from 1991.

[36] The status quo level of quality in Bonaire is defined as 35% coral cover, 300 fish, 45 corals and 100 foot visibility. Lower levels are "good quality" (30% coral cover, 225 fish , 40 corals and 75 foot visibility), "medium-quality" (20% coral cover, 125 fish, 25 corals and 50 foot visibility), and "poor quality" (5% coral cover, 50 fish , 10 corals and 20 foot visibility. The experimental design was constructed such that only declines in the overall quality of the coral reef system could be valued, under the assumption that changes in reef quality would affect all attributes.

Results suggest that divers are willing to pay significantly more than the existing $10 annual user fee for access to the park, with 94% of respondents were willing to pay at least $20, over 75% willing to pay at least $30, and more than 50% willing to pay at least $50. Annual mean WTP is estimated to be $61.[37] As a result of this study, the price of a Bonaire dive tag was increased to $25 in 2005. Finally, Uyarra et al. (2010) use CVM to estimate tourists' WTP for entrance into Bonaire Marine Park. A large majority of tourists (90%) indicated satisfaction with park conditions and current fees, yet only 46% of divers and 40% of non-divers were willing to pay higher fees. Total WTP by divers is found to be significantly lower in 2008 than in 1991 (based on Dixon, 1993), and WTP by U.S. divers is found to be significantly lower than in 2002 (based on Thur, 2010). These results suggest that steady increases in entrance fees to BNMP have effectively captured a good portion of the total consumer surplus from visits to the park.

Waterman (2009) uses CM to estimate the economic value of hypothetical environmental management changes in Folkestone Marine Reserve (FMR), Barbados. Attributes included in the CM design are sewage treatment, water sports zoning and facilities/information.[38] Tourists are found to have slightly higher willingness to pay for additional sewage treatment and facilities than residents. Notably, the author discovers negative willingness to pay by residents for additional water sports zoning in FMR, suggesting that such measures would detract from utility.[39] Ranking of the WTP estimates suggests that of the three non-price attributes, additional water treatment is valued the highest and facilities/information is valued the lowest. In addition to the willingness to pay estimates, this study collected data that could prove useful in directing future valuation work, including respondents attitudes toward beach entrance fees, frequency of visits to the FMR, perceptions of the importance of beaches to the economy of Barbados, perceived magnitude of tourism-induced impacts on socioeconomic and environmental conditions in Barbados, and attitudes toward economic growth.

While not providing explicit values, Pendleton (1994) estimates trip demand for dive sites in the Sandy Bay/West End Marine Reserve in Roatan Honduras. Results of this study suggest that reef quality has a positive and significant effect on demand, and that divers are willing to trade off travel time for increased coral cover and reef topography.

3.6 Coral reef ecosystem service values

Estimates of the value of Caribbean reefs associated with ecosystem services are largely associated with the protection of coastal property or the contribution to coastal property values. Coastal protection values are principally derived using the cost-avoidance (CA) approach. Attempts at estimating the value associated with reef biodiversity also exist in the literature, but are typically based on stated preference methods (e.g. Spash, 2000; Cartier and Ruitenbeek,

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[37] This is described as a conservative estimate by the author based on higher WTP estimates produced by other elicitation formats and the potential for strategic bias by park users.

[38] Each attribute is expressed in terms of three qualitative levels (two higher quality deviations from the status quo). The payment vehicle was described as an environmental (conservation) levy, paid on an annual basis for residents and a per trip basis for tourists. Separate models are estimated for residents and tourists. Sample sizes are 212 and 163 respectively. The author does not report the sampling scheme or the dates of the sample period.

[39] Tourists were willing to pay $61 for moderate increases in zoning but only $26 for total exclusion of watercraft in the FMR.

1999), which suggests some form of use or non-use value rather than ecosystem service value. The PF approach could be employed in this manner, and would require an understanding and explicit modeling of the contributions of marine biodiversity to human well-being.

Regarding the value of coastal protection, included in the analysis by Cesar et al. (2003) are estimates of the NPV of coral reefs in the Caribbean region, based on benefits transfer from cost-avoidance applications reported in Burke et al. (2002). The 19,000 km^2 of reefs in the Caribbean are reported to provide annual net benefits of US$720 million from coastal protection. This study also reports annual biodiversity values of US$79 million, though it is not clear how these estimates were derived.

ETI (2007) also includes benefits transfer estimates from applications of the cost-avoidance methods (Burke and Maidens, 2004), to estimate the value of reefs for coastal protection in Puerto Rico. This study reports an economic value of coral reefs associated with coastal protection is US$9,969,258.

Cartier and Ruitenbeek (1999) use the using the total value of land and property at risk to loss from erosion along the shoreline within the bounds of the Montego Bay Marine Park to estimate the coastal protection value of coral reefs in Montego Bay.[40] The NPV of this land and property that is assumed to be protected by the reef is estimated to be US$65 million. Also in Jamaica, Cesar, et al. (2000) estimate values associated with coastal protection in the Portland Bight Marine Protected Area using benefits transfer. As their data are gathered from outside the Caribbean region, these estimates are likely unreliable and are not reported here. Carleton and Lawrence (2005) use the CA approach to value coral reefs for coastal protection in Turks and Caicos. Coral reefs contribution to coastal protection is valued at $16.9 million per year.

van Beukering et al. (2009) and Bermuda Department of Conservation Services (2009) undertake a large study using a variety of valuation methods to derive the value of coral reef ecosystems in Bermuda. Included in this analysis is an estimate of coastal protection using the CA approach. By estimating potential damages and associated economic losses from a storm event, with and without the presence of a reef, the authors conclude that reefs provide US$266 million in annual service values associated with storm protection. This study also estimates the contribution of reefs to the amenity value of real estate using a hedonic price model applied to data from 593 house and condominium sales.[41] Total amenity value is calculated as the difference between the total price of the houses sold in the dataset and the extrapolated calculation of the house prices in a “deterioration” scenario whereby beaches disappear. Recreational and cultural values associated with reefs are also estimated using a CM survey of Bermudian households. One of the attributes in the CM was water clarity maintained by a healthy coral reef system. Households had an average WTP of US$27 per month for this attribute.

WRI’s *Coastal Capital* series (Burke, et al., 2008a, Burke, et al., 2008b, Cooper, et al., 2008, Cooper, et al., 2009, Wielgus, et al., 2010, Waite, et al., 2011, Kushner, et al., 2011) includes values of coral reefs associated with shoreline protection in St. Lucia, Trinidad and Tobago, Jamaica and Belize and the Dominican Republic, estimated via the CA method. The value of

[40] This appears to fit the description of the CA approach, though the authors it as a PF application.

[41] The authors assume that distance from house to beach is an accurate measure of the coral reef amenity.

coastal property protected by reefs is estimated with a multistep process. First, using GIS, lands vulnerable to wave and storm damage and protected by reefs are estimated. Shoreline stability is then calculated using an array of physical factors and storm probability, and a percentage of that stability that is attributed to reefs is assigned by recalibrating stability in the absence of reefs. Avoided damages are calculated using the value of coastal real estate that is both vulnerable to damage and protected by reefs, multiplied by the probability of a 25-year storm event. Coral reefs in Belize provide an estimated US$120–$180 million in avoided damages per year. Using the HP approach, the costs of beach erosion to hotels in the Dominican Republic are estimated to be between US$52 and US$100 million of the next ten years (Weilgus et al., 2010). Coral reef decline may hasten the speed of such erosion and economic losses.

The approach by Kushner et al. (2011) in Jamaica is to first model the contribution of reef degradation to wave height via the loss of rugosity-induced friction, and subsequently predict the change in beach erosion that would result. The value of potential changes in beach width is then estimated using results from an earlier CM application by Edwards (2009). Average per visitor loss in consumer welfare due to beach erosion attributable to coral degradation is estimated to be US$26.10 per vacation. The total loss of consumer surplus from current rates of reef degradation is estimated to be in excess of US$19 million over 10 years, and up to US$33 million if erosion rates increase due to further reef degradation.

Hargreaves-Allen (2010b) employs benefits transfer to estimate the TEV of ecosystems, species and landscapes on Andros Island, based on ecosystem service values derived from the literature. Of the approximately $260 million in total economic value, only 7% is attributed to coral reefs. In terms of specific services, the author estimates that 25% comes from carbon storage, 19% is from the extraction of raw materials and 11% is attributable to biodiversity. Because the methodology is not clearly articulated in the report, it is difficult to assess the validity of these results. As discussed above, benefits transfer estimates can be unreliable if value estimates are transferred to dissimilar study sites. Because such a small percentage of TEV is assigned to reefs, it seems unlikely that value estimates were derived from the Caribbean region.

3.7 Reef fishery valuation studies

3.7.1 Financial analysis of reef fisheries

Economic analysis of reef fisheries in the Caribbean is largely comprised of market based estimates of costs, earnings and profit. These are typically associated with broader investigations into the socio-economic characteristics of particular fisheries and often combine fisher survey data with primary or secondary landings data. Examples include Sary (2001) and Sary et al. (2003) who report economic data associated with a small scale reef fishery in Jamaica, Jiménez and Sadovy (1997) who analyze the effect of mesh size on the commercial value of the catch in a Puerto Rico trap fishery, Parker and Franklin (unpublished) who analyze catch, effort, landings and profitability in the Barbados trap fishery, Gill et al (2007) who profile catch, prices, costs and revenues for small-scale fishers in the Grenadine Islands, and Franklin (2007) who details several socio-economic factors of the lobster fishery of the BVI including costs, revenues, assets, catch and effort. Linkages between fishers and other economic sectors are commonly discussed in these reports, as are the social and economic importance of the fishery to stakeholders.

Weilgus et al. (2010) use landings and price data to estimate the economic importance of reef and mangrove-dependent fisheries in the Dominican Republic. Gross income from reef-dependent fisheries has decreased significantly from over $41 million in the early 1990s to under $17 million in the early 2000s. With 99% of landing sold domestically, these fisheries serve as an important source of protein and contribute to food security. Waite et al. (2011) examine the economic contribution of reef-related fisheries in Jamaica using gross revenues from reported sales. The authors find that Jamaica's reef-related fisheries were worth an estimated US$34.3 million per year from 2001 to 2005, or roughly 0.3 percent of national GDP, and that fisheries are a significant source of employment and food security.

3.7.2 Economic value and economic impacts

Other studies provide broader estimates of the economic value of fisheries, including Carleton and Lawrence (2005), who estimate that reefs fisheries in Turks and Caicos create $3.7 million per year in gross value added[42] and Schuhmann et al (2011) who provide estimates of economic impacts associated with spending and income in the Barbados trap fishery. The market value of coral reef-associated fisheries harvests in Bermuda was estimated by van Beukering et al. (2009) using existing data from the Marine Resources Section and face to face interviews with six fishermen.[43] Reported average catch rates for reef associated species were multiplied by market prices to approximate value. Fishing costs were estimated to comprise between 40 and 80% of the gross value of total catch. The annual value of coral reef ecosystems as related to Reef-associated fisheries is estimated to be US$4.9 million.

As part of a broader reef valuation effort, Gustavson (2002) uses primary data from fishers and the market price approach to estimate the value of Montego Bay coral reefs associated with nearshore artisanal fisheries. These values constitute estimates of net gains to fishers (producer surplus or rents), as economic costs of provision are deducted from revenues. The NPV of Montego Bay nearshore artisanal fisheries (trap, net, hand line and spear) is estimated to range from-US$1.66 million to US$7.49 million. ETI (2007) reports values associated with commercial and artisanal reef fisheries in Puerto Rico based on fisheries market data. Values associated with reef fisheries are small (US$407,415) relative to an estimated TEV of coral reefs and associated environments in Eastern Puerto Rico of over US$1.852 billion.

In a multi-country analysis, Agar et al. (2005) describe socio-economic characteristics of the U.S. Caribbean trap fishery (Puerto Rico and US Virgin Islands St. Croix, St. Thomas and St. John). Based on a sample of 100 randomly selected trap fishers, estimates of annual catch, effort, revenue, capital investments and costs are generated. Fishers are shown to be highly dependent on the fishery for income. While financial profits are modestly positive[44], economic profits are shown to be largely negative, indicating that fishers are not covering the opportunity cost of their time and capital. This suggests overcapitalization in the fishery and a potentially tenuous future for fishers, and may also indicate that fishers derive non-monetary rewards from fishing.

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[42] This value is less than 8 percent of the estimated TEV of reefs in Turks and Caicos.

[43] Results are also reported in Bermuda Department of Conservation Services (2009). The small sample size used in this study suggests estimates should be interpreted with caution.

[44] Annual fisher profits range from $4,760 to $32,467 in Puerto Rico, and from $3,744 to $13,652 in St. Thomas and St. John. In St. Croix, annual profits range between $9,229 and $15,781.

Cesar et al., 2003 uses benefits transfer based largely on interval estimates of value reported in Burke et al. (2002), to estimate NPV of coral reefs from fisheries in the Caribbean Region. The 19,000 km^2 of reefs in the Caribbean are reported to provide annual net benefits of US$391 million from fisheries.[45]

3.7.3 Other reef values

Edwards (2008) uses CVM to estimate tourists' willingness to pay an environmental tax to fund ocean and coastal management activities. Results are used to predict how decreased coral reef quality might affect visitation decisions and the coastal tourism industry.[46] The authors note that attempts to capture the entire consumer surplus from visitors would cause visitation to decline significantly, but a modest visitor tax of $1 per person would cause only a minor decline in visitation generate revenues far in excess of existing government allocations for coastal zone management.

3.7.4 Other applications of economics to reef resources in the CLME

Beyond monetary valuation, several analyses warrant mention in terms of their contribution to understanding economics associated with the reef ecosystem in the CLME. Market data are used in conjunction with bioeconomic modeling[47] by Ley-Cooper and Chávez (2009) and Chávez (2007) to assess the Caribbean lobster stock, value of total Caribbean lobster harvest, annual profit and total employment. The authors also estimate the rate of harvest and effort that would maximize economic and biological yield in the Chinchorro Bank (northwestern Caribbean coast of Mexico) commercial lobster fishery and recommend policy changes to restore stock and harvest rates to historical levels. Employment levels at different yields (e.g. MSY, MEY) are also evaluated. Caribbean lobster harvest is estimated to have a market value of US$ 286 million, generating annual profits of US$ 169 million, providing employment for 32,000 fishers. Simulation results for the Chinchorro Bank fishery suggest that a 29% reduction in fishing mortality could produce MSY, but only at the risk of socio-economic crisis in the fishery. Moderate annual reductions in fishing effort and mortality (0.025% per year), potentially attained by shortening first fishing season by one week, could restore stocks to levels seen 30 years prior, and produce substantial economic gains (doubling of profits and fishers).

[45] The reliability of these estimates is difficult to ascertain due to the lack of detail provided in this report or in the referenced report.

[46] The CVM survey was administered to a sample of 481 tourists in 2007. Mean WTP is US$130.07 for a general tourism is tax and $165.15 for an environmental tax, which translate to US$16.16 and $20.52 per person per day. Using estimates of the impact of these taxes on visitation, the authors suggest that an environmental tax of $1 per person would cause a 0.1% decline in the visitation rate and would generate revenues of $1.7 million, which is roughly 88% of the ‘‘best case’’ cost estimate for natural resource protection provided by coastal zone managers. A $2 per person tax would decrease visitation by 0.2% and generate revenues of $3.4 million. The authors note that attempts to capture the entire consumer surplus from visitors would cause visitation to decline by 52.4%.

[47] Lobster catch data from the Chinchorro Bank, Mexico from 1982-2006 to calibrate models used in simulations (e.g. mortality, catch-at-age and recruitment functions). Costs of fishing, effort (number of trips) and market values of catch are used to estimate benefits from fishing. Benefit-cost ratios for different levels of catch are combined with estimates of fishing mortality to estimate fishing capacity (number of boats and number of fishers). Exploitation scenarios (MSY, MEY) are simulated in order to estimate resulting profits, employment and benefit-cost ratios. 2004 FAO catch statistics from all 25 countries harvesting spiny lobster are used to generate aggregate values.

The potential for economic incentives to aid in reef conservation has also been analyzed in the Caribbean. Niesten and Gjertsen (2010) summarize case studies employing three different approaches to providing marine resource users with economic incentives for conservation: buyouts, conservation agreements, and alternative livelihoods. Caribbean cases include the gill net and trammel net buyout in St. Croix, USVI, and alternative livelihoods training at the Port Honduras Marine Reserve, Belize and St. Croix East End Marine Park, USVI. de Groot and Bush (2010) provide an overview of the concept of Entrepreneurial Marine Protected Areas (EMPAs), which achieve protection via private sector, market-driven coalitions and governance. Two cases in Curacao are used to detail the conditions under which conservation via private sector governance can benefit both private and public interests. These conditions include legitimate and durable authority, incentives for enforcement and investment in natural assets, public sector oversight and consumer education.

Highlighting the importance of understanding and predicting fisher behavior for policy development, Salas et al. (2004) estimate a random utility model of daily species target selection for three fishing communities on Mexico's Yucatan coast. This work recognizes the impact of daily decisions by fishers regarding (e.g. whether or not to go fishing, what species to target or whether to engage in non-fishing employment or gear repair) on the efficacy of management actions. Using data on expected CPUE and revenues from spiny lobster and octopus, opportunity costs of fishing for alternative targets, the opportunity cost of fishing (i.e. forgone non-fishing earnings), travel costs and weather conditions, fisher responses to changes in species prices and CPUE are simulated. Target choice and target switching are found to be based on resource availability and revenues from prior trips. Decreases in species-specific revenues are found to induce switching to alternative targets.[48] Increases in the availability of species are found to direct effort away from other species.[49] Fisher behavior is heterogeneous across communities, indicating that results from modeling efforts such as this one may not apply directly to other areas.

Ruitenbeek et al. (1999) use fuzzy logic modeling and economic optimization to estimate changes in Montego Bay coral reef quality that would occur from interventions such as solid waste collection, tree planting, sediment traps and construction of a large scale waste treatment facility. Results demonstrate that optimal intervention strategies may be dependent on coral quality targets; hence simply promoting low-cost management interventions may be suboptimal. Household solid waste collection, installation of an outfall, and the use of a river sediment trap are shown to be relatively cost-effective for the case of Montego Bay.[50]

4 Values associated with the pelagic ecosystem

[48] These effects are found to be non- linear. Price changes appear to have threshold effects on target decisions depending on the relative costs and skills required for targeting different species.

[49] Notably, an increase in the availability of octopus may affect the probability of targeting lobster for two reasons: increased probability of octopus catch and decreased likelihood of lobster catch due to predation by octopus.

[50] These interventions could achieve coral quality improvement greater than 10% at a cost (NPV) of US$12 million. Costs of marginal improvement in coral quality are shown to be non-linear: achieving the maximum potential improvement of 20% would entail present value costs of US$153 million.

Relative to studies focused on nearshore reef ecosystems in the CLME, valuation of ecosystem goods and services in the pelagic zone have received little attention. The valuation work that does exist focuses largely on provisioning services associated with commercial fisheries. Exceptions include analyses of cultural services associated with tourism and recreation, especially sport fisheries and other recreational opportunities such as whale watching. Despite the importance of the Caribbean pelagic ecosystem in terms of supporting and regulating services such as habitat provision, egg/larvae transport, atmospheric gas exchange and carbon transfer through the food web, there has been no attempt at valuing such services.

4.1 Pelagic fishery values and analyses

Estimates of market values associated with commercial fisheries in the Caribbean are available from a variety of sources including FAO, CIA, Earthtrends, and the Caribbean Regional Fisheries Mechanism. These data include primary and secondary fisheries employment, the value of fisheries output, imports and exports, and the contribution of fisheries to GDP.[51] National fisheries offices likely have more reliable and recent data, though no attempt was made to compile such data for this report.

4.1.1 Commercial fishery value estimates

As is the case with nearshore fisheries, many economic analyses of pelagic fisheries in the CLME focus on estimates of costs, revenues and profits. The Caribbean Regional Fisheries Mechanism (CRFM) *Fishery Reports* contain numerous examples of commercial fisheries valuations. Other examples include Potts et al. (2003), who report market values (capital, operating costs, revenues, ROI) and employment in the Trinidad and Tobago pelagic flying fish fishery; Grant (2006) who reports commercial longline fishery values in Grenada (costs, expenditures, income), as well as fisher and community dependence. Schuhmann et al. (2010) report costs, net profit and return on investment in the Barbados longline fishery. Hargreaves-Allen (2010b) uses a combination of the MP method and benefits transfer to estimate the TEV of ecosystems, species and landscapes on Andros Island, Bahamas. Estimates of the impact of these ecosystems on the economy of the Bahamas, are provided, including $70 million in gross revenues attributable to commercial fishing.

4.1.2 Economic linkages and pelagic fisheries

A few notable studies have taken economic analysis beyond the relatively straightforward investigation of costs, earning and return on investment. For example, Mahon et al. (2007) estimate the value added as fishery landings in Barbados move up the supply chain to the final consumer.[52] Jaunky (2011) examines the causal relationship between the growth of fish exports and economic growth in SIDS using data from 23 SIDS over the period 1989–2002.[53] Results

[51] These data are incomplete for many countries. As of fall 2011, data on primary and secondary fisheries employment was found for 14 Caribbean nations. Values of fisheries output, imports and exports were available for 13 nations. The contribution of fisheries to GDP was available for 14 nations.

[52] Estimated value added is approximately US$19, which amounts to roughly 2.6 times the landed value of the fishery.

[53] Caribbean states included in the analysis are Antigua and Barbuda, The Bahamas, Barbados, Dominican Republic, Grenada, Haiti, Jamaica, St. Kitts and Nevis, St. Vincent and the Grenadines, and Trinidad and Tobago.

show strong evidence of long-run bi-directional causality between the growth of fish exports and economic growth, indicating that fish exports may be a means for SIDS to sustain economic growth. By extension, proper fisheries management may be a critical component of economic growth for SIDS.

Nguyen and Jolly (2010) use FAO data to estimate aggregate seafood import demand functions for Caribbean countries. Seafood import demand is shown to be price elastic. Seafood imports increase with domestic income and tourist arrivals, and decrease with domestic fishery production. The effect of import tariffs and production support policies on seafood imports and domestic fishery production are analyzed, and both policies are shown to increase producer surplus. Production expansion policies appear to increase consumer surplus, while import tariffs may reduce consumer surplus.[54]

4.1.3 Fisheries multipliers

A potentially important resource understanding the economic impact of fisheries harvests is provided as part of an economic impact analysis for world marine capture fisheries conducted by Dyck and Sumalia (2010).[55] The authors provide estimates of economic impacts and household income effects for each country in the world, which could serve as valuable inputs for economic impact analysis of fisheries in the CLME. These values represent the total amount of output supported by fishery landings and total household income generated through indirect and induced effects due to output from the fishing industry.[56]

4.2 Sport fishing values

Recreational fishing for pelagic species is a source of significant economic value in many Caribbean nations, and a potentially untapped source of foreign exchange in many others. For example, Hargreaves-Allen (2010b) reports over $10 million in gross revenues from guided recreational fishing trips in Andros Island, Bahamas. Ditton and Clark (1994)[57] use the market price method and CVM to derive estimates of spending by participants in recreational billfish tournaments, consumer surplus from participation in billfishing and economic impacts of non-resident expenditures related to billfishing in Puerto Rico.[58] Total annual expenditures sum to

[54] The effects of expansion policies on fishery effort, stock and the overcapitalization are not analyzed.

[55] Their estimates of direct, indirect, and induced impacts suggest that the true economic impact of global marine capture fisheries is nearly three times as large as the value of ex-vessel landings. In other words, each dollar of output in the fisheries sector supports about three dollars of output through linkages with other sectors in the global economy.

[56] Calculation of economic impact multipliers and household income multipliers is straightforward using these estimates. For example, in Table A4 of Dyck and Sumalia, the landed value of fisheries in Barbados is reported as US$3.12 million, which generates economic impacts of US$3.79 million and an income effect of US$0.89 million. This implies that the output multiplier for fishery landings in Barbados is 1.215 (3.79 / 3.12) and the household income multiplier is 0.285 (0.89 / 3.12). That is, each dollar of fishery landings generates an additional 1.215 dollars in total output and 0.285 dollars in household incomes.

[57] Also reported in Clark et al. (1994).

[58] Data were collected using a mail survey of 433 participants in billfishing tournaments held in Puerto Rico between 1991 and 1992. The sample included 347 Puerto Rico residents and 86 non-residents. Average per trip spending on trips targeting billfish (net of tournament fees) was estimated to be $711 and $3,945 by resident and

$26 million, with an additional $18 million in estimated consumer surplus.[59] The economic impact of recreational billfishing in Puerto Rico is estimated to be $4.75 million, with 200 jobs attributable to billfishing. The authors conclude that the recreational billfish fishery is worth many times commercial value of catch. The magnitude of the consumer surplus estimates suggests that additional revenues could be earned and transferred to local economies. Support for this notion is provided by Gillet, et al. (2007) who use cost and revenue estimates for recreational fishing in the BVI to suggest that the economic impact from the development of a recreational fishing industry could exceed US$ 4 million.

4.3 Other recreation in the pelagic ecosystem

Whale watching is an important source of revenues in the Caribbean region, and takes place in as many as 14 nations including TCI, USVI, BVI, Puerto Rico, Martinique, Grenada, and SVG, Antigua, Saint Lucia, Nevis, St. Barthélemy, and Guadeloupe, according to Vail (2005). Estimates for the Caribbean Region derived from Hoyt (2001) and Hoyt and Hvenegaard (2002) suggest that nearly 89,000 people went whale, dolphin or porpoise watching in the Wider Caribbean in 1999, generating revenues in excess of US$11 million. Total expenditure on whale watching in the Dominican Republic alone was US$5.2 million in 1999, and was nearly US$3 million and US$1 million in the Bahamas and Dominica respectively. More recent values from Alie (2008) suggest that up to 568,000 individuals engaged in Caribbean whale watching in 2006, generating nearly US$23 million in revenues. The industry has clearly shown significant growth, with the number of whale watchers increasing by over 100% per year in the early 1990s and over 40% per year on average from 1994 to 1998 (Hoyt and Hvenegaard, 2002). Norman and Catlin (2007) report a value of whale shark tourism in Belize of US$1.32 million.[60] Cline (2008) reports that recreational diving with sharks in the Bahamas has generated roughly US$800 million in gross revenues since 1987, and is estimated to have generated US$78 million in revenue in 2007 alone.

5 Values associated with the continental shelf ecosystem

Of the three ecosystems detailed in this report, the continental shelf ecosystem has received the least attention by economists, perhaps because this system is far removed from economic drivers and economic activity. Exceptions include analyses of commercial fisheries associated with the shelf. For example, Hutchinson (2008) who uses a PF approach to analyze the importance of effort and the price of shrimp (finfish) bycatch in determining the level of shrimp landings in the Trinidad and Tobago shrimp trawl fishery. Effort (trip days) is found to have a significant effect

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non-resident anglers respectively (the latter including the costs of transportation to Puerto Rico). Spending per billfish caught is estimated to be $1,963 and $2,132 by resident and non-resident anglers. Annual consumer surplus for non-resident anglers is estimated to be $11,135.

[59] The consumer surplus estimate was derived using data from a CVM question that asked if respondents were willing to pay into a management fund that would finance law enforcement and research efforts in support of billfish.

[60] Estimation method and data sources are not given.

on landings of shrimp and finfish bycatch. Contrary to expectations and anecdotal evidence, the relative price of the two species was not found to be a significant determinant of landings.

Agar et al. (2005) describe socio-economic characteristics of the U.S. Caribbean trap fishery (Puerto Rico and US Virgin Islands St. Croix, St. Thomas and St. John). Based on a sample of 100 randomly selected trap fishers, estimates of annual catch, effort, revenue, capital investments and costs are generated. Fishers are found to be highly dependent on the fishery for income. Annual fisher profits range from $4,760 to $32,467 in Puerto Rico, and from $3,744 to $13,652 in St. Thomas and St. John. In St. Croix, annual profits range between $9,229 and $15,781. Economic profits are shown to be largely negative, indicating that fishers are not covering the opportunity cost of their time and capital. This suggests overcapitalization in the fishery and a potentially tenuous future for fishers, and also indicates that fishers derive non-monetary rewards from fishing.

6 Summary of Wider Caribbean Marine Resource Values

6.1 Overview of analyses

To date, economic valuations in the CLME have focused on only a limited number of benefits derived from marine ecosystem goods and services, primarily benefits that are relatively easy to measure and convey to the public, such as recreation opportunities in nearshore protected areas, and benefits that are ascribed to easily measured market indicators such those derived from real estate and capture fisheries. Values associated with reefs have received far more attention than those associated with the pelagic or shelf ecosystems, no doubt due to the ease of access to associated user groups by researchers and the relatively straightforward linkages between changes in resource quality and well-being. Table 4 presents a summary of value knowledge by country for the reef and pelagic ecosystems. Table 5 presents a brief summary of the extent of knowledge of Caribbean marine resource values as well as reference notes. Coupled with the gaps identified in section 6.2 and the outline for modeling efforts detailed in section 7, hese tables provide a framework for guiding future valuation efforts in terms of targeting areas and ecosystem services where value estimates are relatively scarce.

WTP and consumer surplus estimates from tourism and recreation are consistently significant across countries, regardless of the valuation method employed. Even in areas where reef quality is diminished, tourists appear to derive considerable net benefits from marine-based recreation in the Caribbean. Reefs in marine protected areas have received particular attention, especially with regard to recreation and tourism. Protected areas in Jamaica and Bonaire have been the subject of multiple valuations over many years. It is clear from the literature that user fees as a basis for financing conservation are underutilized in the region, especially in the case of MPAs, with Bonaire as a possible exception. Financial analyses of MPAs in the Caribbean suggest that these areas provide favorable benefit-cost ratios, and that public investments in reef protection are worthwhile.

After recreation and tourism, the value of reefs for coastal protection and fisheries have garnered the most value estimates, largely as a result of WRI’s Coastal Capital series in Belize, the Dominican Republic, Jamaica, St. Lucia and Tobago. Value of reefs for shoreline protection appears significant especially in tourist areas. If taken in isolation, shoreline protection values create a potential quandary for conservation policy as the value and density of coastal development are correlated with increases in the value of reefs. The value of reefs for fishing is largely limited to market analyses of capture fisheries. It is clear that small scale fishers in the Caribbean are highly dependent on reefs for livelihoods, and that reef-dependent fisheries may be characterized as marginally profitable at best.

Valuation work in the pelagic and continental shelf ecosystems has been modest in comparison to that in the near shore zones, and is limited to market values of capture fisheries and off-shore recreation opportunities. Where the economics of bill fishing and whale watching have been examined, evidence points to viable market opportunities and economic contributions.

6.2 Gaps, unknowns and possibilities for future valuation work

Despite the great deal of valuation work in the WCR[61] and the increasing attention that economic valuation is garnering by policy makers, it is clear that there a great deal remains unknown. Economic values associated with the pelagic and continental shelf ecosystems in the CLME remain largely unspecified. As shown in Table 3, valuation work is absent for numerous nations and territories in the CLME, most notably those in Central America.

Despite a plethora of market data and evidence of overfishing in the WCR (CARSEA, 2007), the economic impacts of overfishing remain largely unexplored. These include effects on national economies, employment, food security and tourism. Likewise, the economic practicality of fisheries subsidies in terms of the relative values of contemporaneous support for livelihoods and future economic costs of overfishing remains unknown.

As overfishing could potentially lead to the loss of historic fishing heritage as well as an important source of protection against exogenous economic shocks, understanding the values from the cultural and security benefits of small scale fisheries seems deserving of attention. Other cultural service values that remain largely unknown include the value of WCR marine ecosystems to research and education and the amenity value of reefs to coastal property owners. This latter value, estimable via the HP method, would appear to be an important partner to studies that estimate the value of reefs for coastal protection.

Supporting and regulating services provided by the marine ecosystems of the WCR that have been recognized as important in the context of natural sciences, have not be linked to valuation. Examples include the contribution of Caribbean reefs and other coastal ecosystems to fisheries production, climate regulation and habitat provision. With appropriate modeling, the PF method could be applied to improve our understanding of these values.

Finally, despite the continued improvement of appropriate methodologies, estimates of non-use values for WCR marine ecosystem goods and services are in short supply.

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[61] Approximately 200 individual value estimates were reviewed for this study.

Table 4: Knowledge of values by country for reef and pelagic ecosystems in the CLME

Reef Ecosystem												Pelagic Ecosystem		
	MPA													
	Benefits	Costs	Recreation/Tourism	Artisanal Fisheries	Biodiversity	Shoreline protection	Research & Education	Non-use values	Raw materials	Bioprospecting	Costs climate change	Commercial Fisheries	Recreational Fisheries	Other recreation
Entire WCR	-	-	-	-	-	-					-	+		+
Anguilla											-			
Antigua- Barbuda														
Aruba														
Bahamas			-	-	-	-	-		-			-	-	-
Barbados	-		+	+								+		
Belize	-			+		-	+					-		-
Bermuda			-			-								
BVI	+	-	+	-									-	
Cayman I.														
Costa Rica	+	-	-										-	
Cuba														
Curaçao	-				-					-				
Dominica					-									-
Dom. Rep.	-		-	-		-								
Grenada					-							-		
Guadeloupe	+		+											
Guatemala												-		
Haiti														
Honduras	-			-										
Jamaica	‡	-	‡	‡	‡	+				+		+		
Martinique														
Mexico			-	-				-						
Montserrat														
Nicaragua														
Panama														
Puerto Rico			-	+	-	-		-					+	
St. Kitts/Nev.														
Saint Lucia	+		+	-	-	-								
SVG					-									
St. Maarten														
Trinidad/Tobago	-		+	-		-						+	-	
TCI	-		+	+	+	-								
USVI	-	-		-										
Bonaire	‡	+	‡											
Martinique														
Saba	+		+											
St. Eustatius														

Table 5: Framework for valuation of living marine resources of the Wider Caribbean Region

Benefits		State of value knowledge					
		Reef Ecosystem		Pelagic Ecosystem		Shelf Ecosystem	
Provisioning Services	Food	‡	□ Knowledge largely limited to market analyses of capture fisheries. □ High degree of dependence of small scale fishers on reefs for livelihoods consistently shown in the literature. □ Artisanal fisheries appear to be marginally profitable at best; most fishers have alternative sources of income. □With appropriate modeling, PF method could be applied in terms of reef contribution to fisheries production.	‡	□ Capture fisheries market values only. □ Fleet profitability appears dependent on access to export markets. □ Examination of the economic costs and benefits of government subsidies seems warranted.	+	□Capture fisheries market values only
	Medicine	+	□ Bioprospecting (option) values appear significant but may not be realized in domestic markets due to governance arrangements. □ Additional valuation work in this area seems warranted.				
Regulating Services	Climate regulation	-	□ Current state of knowledge regarding the economic consequences of reef loss in the context of climate change is inadequate.				
	Hazard protection	‡	□ Value of reefs for shoreline protection appears significant especially in tourist areas; density and value of coastal development increases value of reefs for protection.				
Cultural Services	Recreation	‡	□ WTP/consumer surplus consistently significant, even in areas where reef quality is diminished. □ User fees as a basis for financing conservation are underutilized in the region.	+	□ Market values of sport fishing for pelagic appear significant. □ Whale/dolphin watching demand growing throughout CLME.		
	Research/ Education	-	□ Inadequate understanding of the value of reef ecosystem for education and research purposes.				
	Aesthetics	-	□ Estimation of the amenity value of reefs to coastal property owners via HP method underutilized.				
	Culture	-	□ Estimates of cultural values associated with artisanal uses of the reef to locals and tourists absent from the literature.				
	Non-use values	-	□ Inadequate understanding of the value of reef ecosystem for altruistic/bequest/option value reasons.				

‡ high state of knowledge of resource values
+ moderate state of knowledge of resource values
- low state of knowledge of resource values

7 Recommendations for valuation in the CLME

Paralleling recent recommendations by the EPA's Scientific Advisory Board (EPA, 2009), and the National Research Council (NRC, 2011) it is recommended that economic valuations in the CLME adopt an ecosystem services approach that is expanded in scope and method relative to the current piecemeal approach to valuation and that valuation efforts and supporting data collection be integrated across disciplines, institutions and nations. This integrated, multidisciplinary approach to valuation should begin with the identification of ecological processes that affect human well-being, and include the construction of ecological or bio-physical models that transform natural or anthropogenic changes in ecosystem services into measurable indicators of benefits that are amenable to valuation. Ideally, such models should include quantitative depictions of:

1. How policy actions (or inaction) will affect ecosystem structure and function,
2. How these effects on the ecosystem translate into changes in ecosystem services,
3. How these changes in ecosystem services affect measurable benefits to people (EPA, 2009; NRC 2011).

Examples of measurable benefits indicators include changes in fisheries output, incidence of health effects, beach width, likelihood of storm damage, encounters with species, tourism visits and probability of pharmaceutical discovery. Once quantified, such measures can be readily incorporated into appropriate valuation exercises.

Hence, a framework for valuation in the CLME is as follows:

- Areas of policy importance should be cross-referenced with gaps and unknowns identified in tables 4 and 5 above;
- A series of conceptual models outlining the pathways by which the associated ecosystems provide services and measurable benefits should then be generated (EPA, 2009);
- These general conceptual models can then form the basis for estimation of ecological production functions (NRC, 2011) that facilitate the quantification of cause-and-effect pathways, and ultimately, valuation efforts;
- The production functions should be calibrated to conditions at sites of policy interest by incorporating important natural or anthropogenic changes affecting the ecosystem and transferring these changes through the models to identify benefits changes to specific user groups.

Figure 2 below shows a general representation of a conceptual model that can be tailored to specific changes in ecosystem conditions and ecosystem services. The gaps between the arrows are where the opportunities for modeling occur. Ideally, a set of models can be developed that are amenable to straightforward and accurate calibration to site-specific conditions across the CLME. A suite of conceptual models targeted toward areas of policy interest where valuation information is currently insufficient forms a practical framework for future valuation.

An effort akin to the latter components of such a modeling effort is provided by Kushner et al. (2011) for reef degradation in Jamaica. Starting with the second arrow in Figure 3 below, predicted losses in reef quality are used to estimate changes in the regulating service of wave height reduction. The link between these steps is a model of wave height specified as a function of rugosity-induced friction. Wave height changes are then used to predict the change in beach erosion using a second modeling effort. The value of potential changes in beach width is then estimated using a CM application. A pollution effects model specifying the relationship between storm water runoff and reef quality changes would serve as a useful foundation for this pathway. Such a model would allow policy makers to directly connect changes in pollution and economic value.

Figure 2: Ecosystem-to-benefits pathway general form

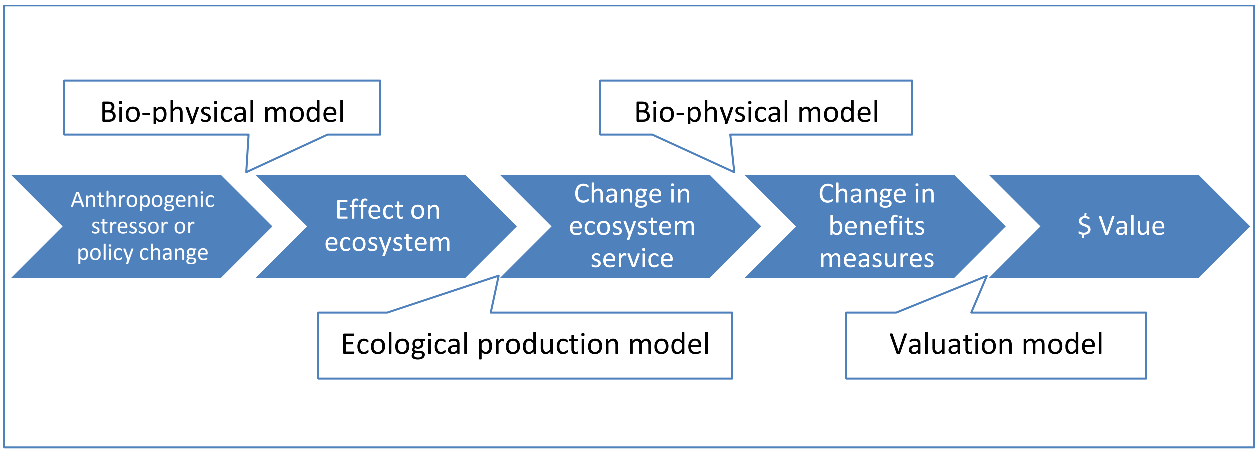
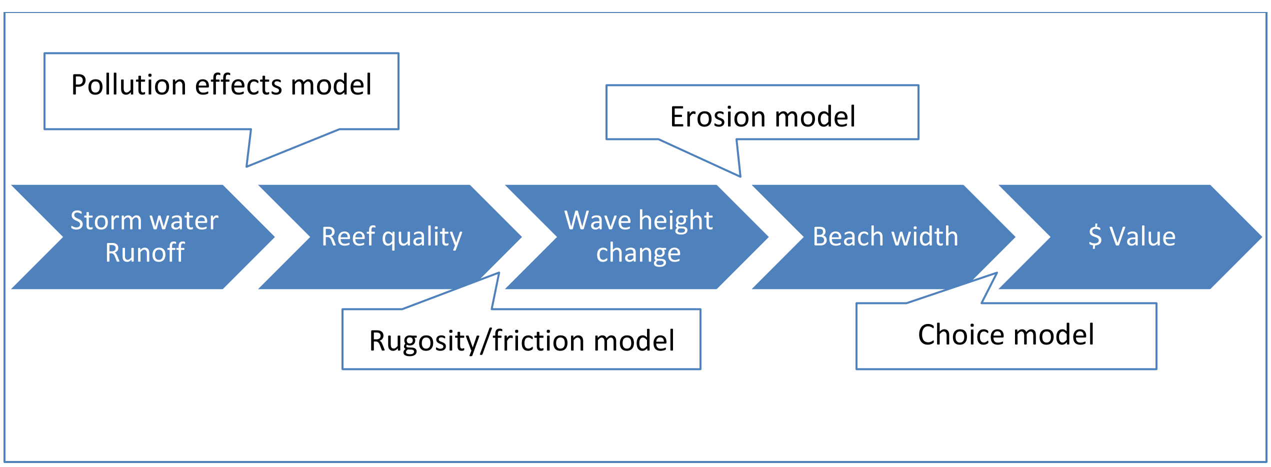


Figure 3: Ecosystem-to-benefits pathway example



Education of the general public regarding the importance of ecological change is an important co-requisite to efforts aimed at expanding the scope of valuation of marine ecosystems in the WCR. In order to make appraisals of acceptable tradeoffs, the public must be accurately informed of the human, social and ecological consequences at issue (EPA, 2009). When valuations involve the use of survey instruments, interviews or focus groups, education efforts should be directly incorporated. However, as such efforts most certainly involve sampling from a larger population, additional efforts should be made to inform the general public of the potential for changes in ecosystem services and the associated consequences in terms of costs and benefits to households and society at large. Importantly, these education efforts should present a scientific appraisal of what stands to be gained or lost under the business-as-usual (BAU) paradigm relative to that which can be accomplished with improved governance and resource management.

It is also recommended that these outreach efforts be used as an opportunity to identify benefit measures of particular concern to different user groups, so that subsequent valuation studies can be directed the indicators of most concern. Ranking or rating exercises that allow respondents to indicate the relative importance of potential effects of policy or BAU will allow researchers to minimize the complexity of valuation exercises and may lend insight into potential value differences across user groups.

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