Assessing Ecosystem Services in Mangroves: Insights from São Tomé Island (Central Africa)

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Mangroves are some of the most productive coastal systems on the planet and provide valuable ecosystem services (ES). They are especially important in threatened ecosystems and developing countries, where they are likely to have direct impacts on local communities. An approach based on ES allows assessing ecosystems across the domains of ecology, sociology and economy. This study focused on the evaluation of ES in mangroves and started by creating a comprehensive global list of mangrove ES based on the Millennium Ecosystem Assessment. These services were then quantified using the best available indicators for mangrove systems. The mangroves of Diogo Nunes, São João dos Angolares and Malanza, located in the São Tomé Island, were used to illustrate the challenges in applying ES indicators in this type of ecosystems. The obtained results confirmed that mangroves can provide important and diverse services. However, the high variability among mangrove systems affects their ability to deliver ES, requiring caution for the extrapolation across regions. This assessment emphasizes how the ES framework can be used as a tool to develop management plans that integrate conservation goals and human wellbeing.

Keywords: environmental conservation, ecosystem services indicator, quantification of ecosystem services, transitional systems, gulf of guinea

INTRODUCTION

The concept of ecosystem services (ES) appeared in the 1960s, intending to link ecological and economic research (Martin-Ortega et al., 2015). Since then, the concept of ES has been greatly expanded and in the Millennium Ecosystem Assessment (MEA) it was defined as 'the functions and products of ecosystems that benefit humans, or yield welfare to society' (MEA, 2005). The definition of ES remains elusive, often varying according to a stakeholder or specific context (Fisher et al., 2009), even though understanding the connection between human society and ecosystems is crucial to integrate the domains of ecology, economy and sociology (MEA, 2005). Later in 2009, the Common International Classification of Ecosystem Services (CICES) emerged and defined ES as the 'contributions that ecosystems make to human well-being' (Haines-Young and Potschin, 2018), adapting the MEA methodology to obtain a more detailed hierarchical method to classify ES (Czúcz et al., 2018). More recently, the Intergovernmental Platform on Biodiversity and Ecosystem Services

(IPBES) system was proposed. It differs substantially from previous ES assessment systems for being anchored on the concept of Nature's Contributions to People, defining ES as 'all the positive contributions, losses or detriments, that people obtain from nature' (Brauman et al., 2019).

The assessment of ES involves identification, mapping and quantification, the latter of which can be measured in three domains: biophysical, social, and economic (Haines-Young et al., 2018). In this study, only the identification and quantification of ES were considered. Together, these two steps provide stakeholders with tools to raise awareness and to manage the landscape effectively (Vihervaara et al., 2017). Mapping ES involves methodologies from all the domains previously mentioned and provides a spatial representation of the capacity of a system to deliver ES (Vihervaara et al., 2018). During the last 10 years, major changes and advances have been made in ES mapping (Englund et al., 2017), and several countries have been incorporating ES assessment methodologies in decision-making. In the European Union efforts are being made to develop integrated methodologies for ES mapping, valuation, accounting and assessment (e.g. ESMERALDA Project; www.esmeralda-project.eu) and to promote the use of ES in decision making (Burkhard et al., 2018) and implementation of the EU Biodiversity Directive. Nonetheless, many ES are difficult to identify, especially in under-studied systems, and quantification relies on indicators, which are often non-existent, inadequate or hard to measure (Müller and Burkhard, 2012). The economic valuation provides a monetary justification for the allocation of financial resources toward ecosystem preservation (Gómez-Baggethun et al., 2010) and it is based on the measure of the economic value of ES (Brander et al., 2018). However, they require strong safeguards since many ES hard to convert into a marketable value are undervalued (Castro et al., 2014).

The MEA has become the classical system of ES classification, recognizing four categories: 1) provisioning (e.g. food, fiber, and other resources); 2) regulating (e.g. climate regulation, protection against soil erosion, flood protection, water purification), 3) cultural (e.g. recreation, spiritual values, aesthetics, education and research); and 4) supporting (e.g. habitat diversity and nutrient cycling). The relevance of each of these categories is strongly context-dependent. For instance, in developing countries provisioning services have a more direct association with poverty alleviation and food security, and their impact is often felt almost instantly by human populations (MEA, 2005). The supporting, regulating and cultural services tend to be overlooked since their impacts on human well-being are less direct and they are harder to measure (Alcamo et al., 2003b). However, this does not mean that they are less relevant to human wellbeing (TEEB, 2010).

The *tragedy of the commons* is often evident in marine fisheries of developing countries, due to the difficulties in determining and enforcing property rights, while populations are often over-reliant on fisheries that depend on ecosystem integrity (Alcamo et al., 2003a; Ostrom and Ostrom, 2015). Our unawareness of ecosystem functioning and ES delivery also undermines our ability to manage resources (Alcamo et al., 2003b), making it crucial to find objective means of quantification. Furthermore, in many developing countries, the voices of impoverished local communities and conservation interests are ignored by political and economic interests, contributing to an undervaluation of ES (Samarakoon, 2004). Sustainable ecosystem management is key to preserve the long-term delivery of ES, but requires practices that promote ecological functioning (Agbenyega et al., 2009).

Mangrove forests are considered some of the most productive systems on Earth (Walters et al., 2008) and provide important ES, often related to the daily activities of rural communities (Spalding et al., 2010). These intertidal forests are known for their capacity to provide coastal protection against natural hazards, such as storm waves, and erosion (Badola and Hussain, 2005). Wetland areas, like mangroves, are also known to store carbon, which is an appealing contribution to climate change mitigation (Donato et al., 2011). They are very important nursery areas for a large variety of fish and invertebrates, providing refuge and food for many of these species during the first development stages (Mumby et al., 2004). However, they are frequently under strong anthropogenic pressure (Spalding et al., 2010) and are among the most threatened marine ecosystems (Duke et al., 2007). Over the last 20 years, 35% of the global area of mangrove forests was lost (Valiela et al., 2006). One of the most common drivers of biodiversity loss is habitat transformation at the expense of land conversion to agriculture, although this is a type of ES trade-off less documented in coastal ecosystems (MEA, 2005). The failure to implement adequate policies and the persistence of ill-defined property rights are some of the underlying causes of this loss (Sathirithai, 1998), making it essential to recognize the value of these ecosystems whilst developing efficient evidence-based conservation strategies. Also, there is a demand to understand the path of distribution of services costs and benefits, moreover to perceive the impact of trade-offs between ES to avoid corruption (MEA, 2005). The Ecosystem Service Framework (ESF) is a benefit-oriented approach and a valuable tool to engage managers and regulators, since it focuses on social and economic benefits, setting the basis for policy changes (Alcamo et al., 2003b). Recent studies have proposed methodologies to assess mangroves ES, namely through mapping (e.g. Kuenzer and Tuan, 2013), economic valuation (e.g. Barbier et al., 2011) and applied social evaluations (e.g. Satyanarayana et al., 2012).

This study aims to highlight the ecological and socio-economic importance of mangroves by evaluating the ES they deliver at a global, regional and local scale, using São Tomé Island (Central Africa) as a case study. On a global scale, mangrove ES will be identified and compared to other estuarine and terrestrial ecosystems. At the regional scale, mangrove ES in Africa will be quantified using previously identified adequate indicators and values available in the literature, and again comparing with regional estuarine and terrestrial ecosystems. Finally, São Tomé mangrove ES will be quantified using indicators based on field assessments or expert-based knowledge. A comparison across scales will showcase the challenges of ES assessment, which are known to be highly variable across regions and at multiple geographical scales. Even though the main focus of the study will be the regional and local assessments, it is essential to develop a global list of ES provided by mangroves. This is the first attempt to use the ESF in São Tomé mangroves and provides important clues to promote biodiversity conservation and the sustainable use of resources.

METHODOLOGY

Mangrove ES were identified (*Identification*) at a global scale, and quantified (*Quantification*) at a regional scale for tropical Africa, based on an extensive literature review. Subsequently, three São Tomé mangroves were chosen to evaluate ES, using the best available information (*Assessment of Ecosystem Services in São Tomé Mangroves*).

Assessment of Mangrove Ecosystem Services

Identification

An existing general list of ES (Layke et al., 2012) was adapted, focusing on mangroves ES. The list was revised to include all ES that were found in the literature, by performing a search for the keywords "ecosystem services" and "mangroves", on Google Scholar and Web of Knowledge, between January and June of 2018.

Mangroves can be classified as terrestrial, aquatic, or both (Friess et al., 2016). Therefore, the relative importance of mangrove ES was assessed by identifying and comparing ES delivery in terrestrial, estuarine (excluding mangroves) and mangrove ecosystems. To do so, the keywords "terrestrial" or "estuarine" and "ecosystem services" were used. This study followed the MEA ES classification scheme (MEA, 2005). Although other classification methodologies were considered, such as CICES and IPBES, the final decision was in favor of MEA, due to its well established and recognized methods (Caputo et al., 2019) and the thorough list of specific indicators provided by MEA assessments.

Quantification

Quantification requires the use of ES indicators. Several general lists of ES indicators have been published, even though there are no operational practices or guidelines to develop or select ES indicators (Broszeit et al., 2017). This study was based on an existing list of ES indicators (Layke et al., 2012), which was improved by adding and replacing indicators following information found in the literature (**Table 1**). Indicators were selected based on a confidence level assessment, using a scoring system based on two elements: 1) the ability to convey information: intuitive; sensitive; accepted; and 2) data availability: gathered at sufficient temporal and spatial scales; processed and available; normalized and disaggregated. Each element had three underlying criteria, classified from one (low) to three (high) and the value of each element was obtained as the arithmetic mean of the criteria scores (see Layke et al. (2012) for further details). The indicator with the highest score was selected (i.e. when the sum of each element value resulted in a low or medium score, the decision fell on the selection of another indicator to replace it).

Indicators can be measured directly, for instance when a state or process is quantified during field observations, or indirectly, for instance when based on proxy indicators, expert-based knowledge, or when the data requires interpretation or adjustments (Vihervaara et al., 2017). Most indicators were selected based on data availability. The most common adjustments were the addition of a temporal dimension to

express ES flow (Vihervaara et al., 2017) and the conversion to International System units. Some cases required special adjustments, such as ES *biomass fuel*. This ES is most commonly assessed based on the consumption of fuel *per capita* but an estimate of fuel consumption of the overall population in the vicinity of the study area was used in this study, to provide a value representative of the mangrove system.

Several scientific research papers and reports were consulted to quantify each indicator at the regional level (Tropical Africa), separately for each of the three ecosystems considered (mangroves, estuaries and land). The search was performed on the web-search engines previously mentioned (*Identification*), it was established that the limit was the first 30 publications of the results, since it was intended to obtain as much information as possible without reaching the point of data repetition and that all the information acquired was specific to the selected indicators and study region.

Assessment of Ecosystem Services in São Tomé Mangroves

The assessment approach developed in *Assessment of Mangrove Ecosystem Services*, was then applied in the context of São Tomé Island. Firstly, each mangrove was mapped, using GPS locations and satellite images (Google Images, 2018. São Tomé. Digital Globe) on QuantumGIS (QGIS 2.18.13). The satellite image analysis was essential to identify areas with mangrove trees and watercourses designated as the "mangrove area". Then, to better comprehend the surrounding areas of the study site as well as to characterize the type of stakeholders present, the main land-use types (Burkhard et al., 2009) around each mangrove were mapped. This step is essential for well-developed decision-making. Since there was no standardized value for the definition of buffer area, we opted for the lower value found in the literature, 100 m, because of the small scale of the case study (Macintosh and Ashton, 2002; Atkinson et al., 2016).

Then, the improved global list of ES for mangrove systems (**Table 2**) was used to identify ES in São Tomé Island, using site-specific literature, complemented by expert-knowledge and a field assessment conducted in August 2017. The ES quantification was considered only for services with suitable data, which were *wild foods*, *water regulation*, and *nursery area* services. The indicators used were obtained from *Quantification* and the estimates were preferably based on field assessments. The *wild foods* service quantification was based exclusively on literature available for the study area (Pisoni et al., 2015), where it was possible to quantify the number of species used as food source. While *water regulation* and *nursery area* services were calculated based on the field assessments. The first was calculated by measuring the concentration of nitrogen in the water, while the second was calculated using different fishing techniques to quantify the proportion of juveniles in the local populations. The assessment only considered the mangrove area defined in the mapping.

The ES assessment took place in the mangroves of Diogo Nunes, Angolares and Malanza (**Figure 1**), in São Tomé Island (0°25′N - 0°01′S, 6°28′E - 6°45′E). These systems were chosen to represent the diversity of mangroves on the island, considering spatial distribution, mangrove size and anthropogenic pressure.

TABLE 1 | Indicators for ecosystem services quantification based on the reference article (Layke et al., 2012). Data availability (none *, little **, plenty ***) and necessary modifications. Services with no indicators provided in the reference article are represented as × and services absent in the reference article are shaded in gray.
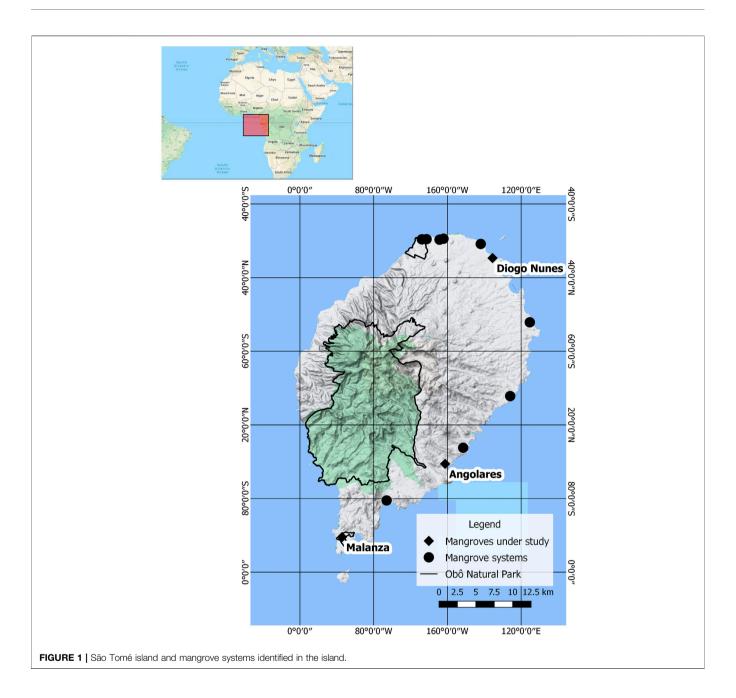
	Ecosystem Services	Indicators based on reference article	Data availability	Selected indicator
Provisioning	Capture fisheries	Value of marine production	*	Yearly rate of seafood extraction (kg km^{-2} year^{-1}; Hattam et al., 2013)
	Crops cultivation	Crops production	*	Land prices near the ecosystem (US$ km^{-2}; Liquete et al., 2016)
	Aquaculture	Aquaculture production	***	Yearly aquaculture production (kg year^{-1})
	Wild foods	Number of wild species used as food	***	-
	Timber	Forest biomass production	***	Yearly forest biomass production (kg km^{-2} year^{-1})
	Fibers and ornamental resources	Value of forest products	***	Yearly value of forest products (US$ km^{-2} year^{-1})
	Biomass fuel	Fuelwood consumption	***	Yearly consumption of fuelwood (kg km^{-2} year^{-1})
	Genetic resources	Value of genetic resources	**	Yearly value of genetic resources (US$ km^{-2} year^{-1})
	Medicines and pharmaceuticals	Value of pharmaceutical products developed in natural systems	**	Yearly value of medical resources (US$ km^{-2} year^{-1}; de Groot et al., 2012)
	Water for non-drinking purposes			Yearly freshwater runoff (m^3 year^{-1}; Egoh et al., 2012)
Regulating	Air quality regulation	Flux of atmospheric gases	***	Yearly flux of atmospheric gases (g km^{-2} year^{-1})
	Global climate regulation	Capacity of Carbon sequestration	***	Yearly rate of carbon sequestration (kg km^{-2} year^{-1})
	Regional climate regulation	Evapotranspiration	***	Evapotranspiration rate (cm day^{-1})
	Water regulation	Soil water infiltration	*	Nitrogen concentration (mg N L^{-1}; EEA, 2018)
	Coastal Erosion regulation	Landslide frequency	*	Percentage of wave attenuated (for 100 m of ecosystem length; Atkinson et al., 2016)
	Groundwater recharge			Groundwater recharge rate (mm km^{-2} year^{-1}; Burkhard et al., 2009)
	Wastewater treatment	Amount of waste processed by ecosystems	*	Adaptation of the indicator **Nutrient retention** (Egoh et al., 2012) to **Percentage of nutrients absorbed during wastewater discharge** (%)
	Disease regulation	Disease vectors predator populations	**	-
	Soil quality regulation	×		Index of soil quality (BISQ; Feld et al., 2010)
	Pests regulation	×		Presence/absence/frequency of pests (Hattam et al., 2013)
	Pollination	×		Costs of bees (US$ pollination period^{-1}; Egoh et al., 2012)
	Natural hazards regulation	Mortality losses from natural disasters	***	Mortality losses during natural hazard with or without mangroves
	Nutrient cycle	Value of nutrient cycle for terrestrial ecosystems	*	Adaptation of the indicator **Turnover rate** (Burkhard et al., 2009) to **Yearly rate of nitrogen storage** (N; kg km^{-2} year^{-1})
Cultural	Aesthetic/ethical values	Number of nature/rural visitors	**	Yearly number of visitors for sightseeing (visitors year^{-1})
	Recreation and ecotourism	Visitors to natural areas	***	Yearly number of visitors for recreation (visitors year^{-1})
	Spiritual and religious values	×		-
	Cultural heritage	×		Number of households which consider an area or aspects of an area as cultural heritage (Böhnke-Henrichs et al., 2013)
	Scientific/education	×		-
Supporting	Primary production	NPP	***	-
	Nutrient flow			Yearly nitrogen flow (N; kg km^{-2} year^{-1} - Burkhard et al., 2009)
	Water cycling	×		Transpiration by total evapotranspiration (Burkhard et al., 2009)
	Habitat heterogeneity			Habitat diversity index (Burkhard et al., 2009)
	Nursery area			Number of species with juveniles by the total amount of species (Vasconcelos et al., 2011)

TABLE 2 | Ecosystem services identified globally in mangrove, estuary and terrestrial systems. Black circles and white circles represent the presence and absence, respectively, of each ecosystem service (adapted from Layke et al., 2012 to mangrove ecosystems).

	Ecosystem Services	Mangrove	Estuary	Terrestrial
Provisioning	Capture fisheries	●	●	○
	Crops cultivation	●	●	●
	Aquaculture	●	●	●
	Wild foods	●	●	●
	Timber	●	●	●
	Fibers and ornamental resources	●	○	●
	Biomass fuel	●	○	●
	Genetic resources	●	●	●
	Medicines and pharmaceuticals	●	●	●
	Water for non-drinking purposes	●	●	○
Regulating	Air quality regulation	●	●	●
	Global climate regulation	●	●	●
	Regional climate regulation	●	●	●
	Water regulation	●	●	○
	Coastal erosion regulation	●	●	○
	Groundwater recharge	●	●	●
	Waste treatment	●	●	○
	Diseases regulation	●	●	●
	Soil quality regulation	●	●	●
	Pests regulation	●	●	●
	Pollination	●	●	●
	Natural hazards regulation	●	●	●
	Nutrient cycle	●	●	●
Cultural	Aesthetic/ethical values	●	●	●
	Recreation and ecotourism	●	●	●
	Spiritual and religious values	●	●	●
	Cultural heritage	●	●	●
	Scientific/education	●	●	●
Supporting	Primary production	●	●	●
	Nutrient flow	●	●	●
	Water cycling	●	●	●
	Habitat heterogeneity	●	●	●
	Nursery area	●	●	○

Black mangrove *Avicennia germinans* is present in all study mangroves, and true mangroves *Rhizophora* sp. are only absent from Diogo Nunes. The Diogo Nunes mangrove, on the northeast coast, is the smallest study system (0.01 km^2, Afonso, 2019). It is an intertidal mangrove system with low vegetation coverage, surrounded by agricultural fields (47.5% of the study site, **Figure 2A**; Afonso, 2019) and a community of 392 people (INE São Tomé e Príncipe, 2014). Located on the east coast, the Angolares mangrove has 0.13 km^2 (**Figure 2B**; Afonso, 2019). It is formed by two branches that are only connected to the sea during periods of high runoff or spring tides. The vicinities are occupied mostly by agroforests (59.4%)

(Afonso, 2019). The nearest community, São João dos Angolares, has 2037 inhabitants (INE São Tomé e Príncipe, 2014). The Angolares and Diogo Nunes watersheds have both been seriously modified by human activities, and are mostly covered by agroforest (73% and 70%, respectively - based on Soares, 2016). Malanza, on the southern coast of the island, is the largest mangrove in the country (Brito et al., 2017), covering 1.52 km^2 (Afonso, 2019). It is dominated by mangrove and agroforests (53.6% and 36.7%, respectively, **Figure 2C**; Afonso, 2019). This is an open system, but its connection to the sea is heavily constricted by a bridge, which affects water, sediment and ecological dynamics (Félix et al., 2017). This bridge connects the two nearby communities of Porto Alegre

FIGURE 1 | São Tomé island and mangrove systems identified in the island.

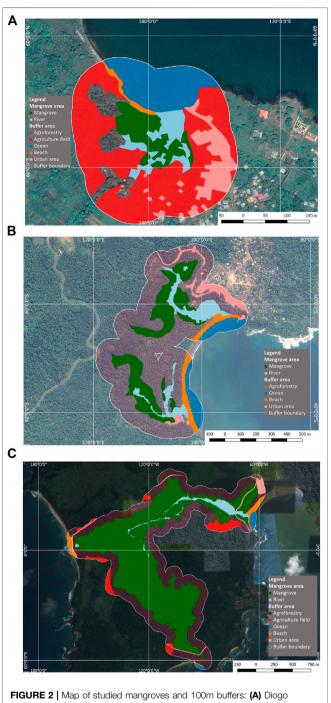and Vila Malanza, which have respectively 795 and 550 inhabitants (INE São Tomé e Príncipe, 2014).

RESULTS

Global Identification of Mangrove Ecosystem Services

A total of 33 ES were identified in mangroves globally (**Figure 3**). Most of these were regulating (13) or provisioning services (10), while cultural and supporting services were less represented (5 each). The original ES list (Layke et al., 2012) was extended to

include water for non-drinking purposes, groundwater recharge, nutrient flow, habitat heterogeneity, and nursery area. Some ES, such as livestock, freshwater, and soil formation were excluded since they were not indicated for mangroves. Livestock and freshwater services were never mentioned for mangroves in the literature, while the role of mangroves for soil formation remains a topic of debate (Lee et al., 2014). *Nutrient cycling* is sometimes considered a supporting service (Burkhard et al., 2009), here it was classified as a regulating ES, while *nutrient flow* was classified as a supporting ES.

Only 31 and 27 of the 33 mangrove ES listed were assigned to other estuarine systems and terrestrial ecosystems, respectively

FIGURE 2 | Map of studied mangroves and 100m buffers: **(A)** Diogo Nunes, **(B)** Angolares, and **(C)** Malanza (image sources: Google Images, 2018; Digital Globe).

(**Table 2**). No ES was exclusive to mangroves, even though none of the systems used for comparison delivered as many ES as mangroves. Most differences between mangroves and estuaries were related to provisioning services since estuaries do not provide forest products, such as *fibers and ornamental resources* and *biomass fuel*. Regulating services were less

represented in terrestrial systems since many of these are associated with water (**Table 2**).

Regional Quantification of Mangrove Ecosystem Services

Mangrove ES were quantified using existing indicators (**Table 1**) and data from Africa, obtained between 1964 and 2019 (**Table 3**, **Figure 4**). A thorough literature review provided values for most indicators (~43%), especially those relating to provisioning and supporting services. Regarding provisioning services indicators, only the one associated with *wild foods* was used with minor adaptations, the rest had to be adjusted to include a temporal/spatial scale. All indicators of regulating services were adjusted, except for those that were not used due to a low confidence level. Only two indicators were found for cultural services, of which only *recreation and ecotourism* was quantified. Concerning supporting services, the *primary production* indicator was used without modifications, but all others were adapted. Only 52% of the mangrove ES were quantified (excluding indicators with low or medium classifications) (**Table 3**), including 60% of both provisioning and supporting services (**Figures 4A,C**), 31% of regulating services (**Figure 4B**), and 20% of cultural services (**Figure 4C**).
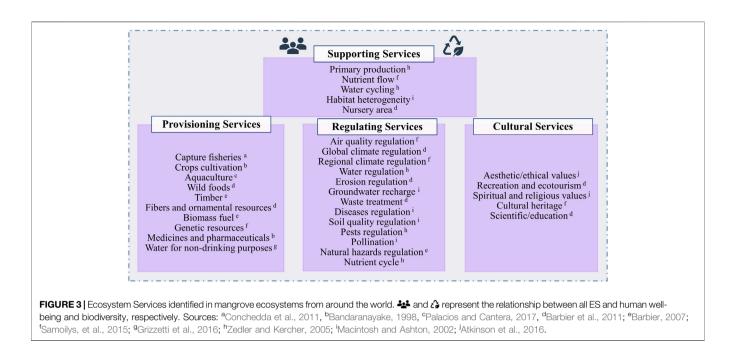
The quantification of ES varied between ecosystems (**Table 3**). Seven ES indicators presented the most benefits in mangroves, namely capture fisheries, global climate regulation, regional climate regulation, water regulation, groundwater recharge, habitat heterogeneity and nursery area (**Table 3**). In this study, most benefits do not necessarily correspond to the quantification of the highest value, as it depends on the indicator being used. For instance, in this study, it corresponds to a high value in *aquaculture* but to a low value in *air quality regulation*. Only terrestrial systems presented more benefits than mangroves in some ES, more specifically four, that includes three provisioning services and one cultural service.

São Tomé Mangroves Ecosystem Services
Identification
Five provisioning, four regulating, two cultural and one supporting service were identified in São Tomé mangroves based on a literature review (**Table 4**). This survey was complemented by fieldwork assessments, which provided information to identify 15 additional ES. Literature information was less representative for regulating and supporting services and all ES listed for São Tomé in the literature were identified during field assessments.

Quantification
Of the 27 ES identified in São Tomé mangroves, only wild foods, water regulation, and nursery area were quantified locally (**Table 5**). Wild foods, which consisted mostly of seafood, had higher values than those found in the literature, while water regulation had lower values. The quantification of the supporting service *nursery area* was slightly lower in São Tomé that in other mangroves in tropical Africa.

FIGURE 3 | Ecosystem Services identified in mangrove ecosystems from around the world. 👥 and ♻ represent the relationship between all ES and human well-being and biodiversity, respectively. Sources: [a]Conchedda et al., 2011; [b]Bandaranayake, 1998; [c]Palacios and Cantera, 2017; [d]Barbier et al., 2011; [e]Barbier, 2007; [f]Samoilys, et al., 2015; [g]Grizzetti et al., 2016; [h]Zedler and Kercher, 2005; [i]Macintosh and Ashton, 2002; [j]Atkinson et al., 2016.

DISCUSSION

The current study emphasizes the importance of mangroves as providers of a high variety of ES to local populations, particularly important in developing countries. ES provided by the mangrove biome were identified, quantified for tropical Africa and, for the first time, assessed for São Tomé Island. This exercise highlighted the challenges in obtaining local data and reliable indicators to quantify mangrove ES in developing countries. This study also provides an example of the relevance of ES approaches to support the implementation of conservation policies.

Mangrove as a Source of Ecosystem Services

Previous studies identifying global mangrove ES listed only 17 ES (Barbier et al., 2011; Vo et al., 2012; Drakou et al., 2017), while this study identified 33. This difference is most likely due to the fact that previous studies focused on selected ES, for instance, provisioning and regulating services (Liquete et al., 2013), while the current study assessed all mangroves ES, providing a comprehensive list of services and indicators.

The differences between the number of ES identified in the literature 12) and those identified, in the present study, for São Tomé (an additional 15) seem to be mainly due to two related factors: 1) spatial scale and 2) information availability. All mangroves in São Tomé are extremely small, with an area ranging from 0.01 km² (Diogo Nunes) to 2 km² (Malanza). The effect of scale in mangrove ES delivery is related to the minimum area required for the development of particular activities or ecological processes. For instance, ES delivered as *aquaculture* are highly dependent on the available production area. Although the regular pond size for shrimp production in Ecuador can be as big as 0.5 km² (Hamilton, 2011), most mangroves in São Tomé are

too small to support economically sustainable aquaculture (Martín-López et al., 2019). On the other hand, the resolution and extent of the ES assessments may also affect the obtained results and ES estimates differ substantially between the fine and coarse resolution analyses (Grêt-Regamey et al., 2014). Our results suggest that finer resolution assessments conducted at the community level (i.e. mangrove specific information) capture ES spatially explicit information that would be lost at a coarser resolution. That is also related with the second factor concerning the difficulty in assessing some ES without in-depth analyses (*Challenges in Assessing Ecosystem Services*). Very few studies focused on ES identification in mangroves for the tropical African region (e.g. Owuor et al., 2019b). The additional ES identified for STP mangroves highlighted the importance of conducting local surveys to create comprehensive ES inventories (Afonso, 2019).

Mangroves across the globe provide a diverse set of ES. Differences were found in terms of the services provided by estuarine and terrestrial ecosystems, with the lowest number of ES identified in terrestrial systems, which was expected, as other aquatic environments share more similarities with mangrove ecosystems. When comparing the results of ES quantification, mangroves presented more benefits in seven out of the 14 quantified ES. This may be represented by the highest (e.g. net primary production indicator) or the lowest (e.g. nitrogen concentration on water indicator) quantities of a certain service suggesting that mangroves could have an overall positive impact on human societies, namely when compared to estuarine and terrestrial ecosystems. Regarding São Tomé mangroves, two out of three ES quantified at this local scale presented more benefits than those identified for tropical Africa, namely *wild foods,* and *water regulation*, underpinning the relative importance of small mangroves (Curnick et al., 2019).

The potential and the effective delivery of mangrove ES (ES flow) is another relevant question to be considered since it can be

TABLE 3 | Ecosystem Services identified and quantified (mean values and range) in mangrove, estuary and land systems from Tropical Africa (adapted from Layke et al., 2012). The number of estimates used to calculate the mean is indicated in parentheses.

| Ecosystem services | Indicators | Mangrove | | Ecosystem of reference | | | |
| | | | | Estuary | | Terrestrial | |
		Mean value	Range	Mean value	Range	Mean value	Range
Provisioning							
Capture fisheries	Yearly rate of seafood extraction (kg km^{-2} year^{-1})	4.19x10^5 (6) [1]	Min: 352.53 Max: 1.09x10^6	4.27x10^3 (5) [2]	Min: 300 Max: 1.02x10^4		
Crops cultivation	Land prices						
Aquaculture	Yearly aquaculture production (kg year^{-1})	2.00x10^7 (1) [3]	-	8.38x10^5 (1) [4]	-	2.09x10^7 (3) [5]	Min: 2.32x10^6 Max: 5.74x10^7
Wild foods	Number of wild species used as food	13.5* (2) [6]	Min: 10 Max: 17			122 (3) [7]	Min: 35 Max: 272
Timber	Yearly forest biomass production (kg km^{-2} year^{-1})	2.49x10^7 (1) [8]	-				
Fibers and ornamental resources	Yearly value of forest products (US$ km^{-2} year^{-1})					294.88 (6) [9]	Min: 4.17x10^{-3} Max: 1.06x10^3
Biomass fuel	Yearly consumption of fuelwood (kg km^{-2} year^{-1})	1.67 (1) [10]	-			3.82x10^4 (5) [11]	Min: 9.11 Max: 1.14x10^5
Genetic resources	Yearly value of genetic resources						
Medicines and pharmaceuticals	Yearly value of medical resources						
Water for non-drinking proposes	Yearly freshwater runoff (m^3 year^{-1})	4.7x10^9 (1) [12]	-				
Regulating							
Air quality regulation	Yearly flux in atmospheric CH$_4$ (g km^{-2} year^{-1})					2.00x10^{-3} (1) [13]	-
Global climate regulation	Yearly rate of carbon sequestration (kg km^{-2} year^{-1})	5.36x10^7 (3) [14]	Min: 79.3 Max: 1.6x10^8	4.8x10^5 (1) [15]	-	5.52x10^4 (2) [16]	Min: 2.80x10^4 Max: 8.24x10^4
Regional climate regulation	Evapotranspiration rate (cm day^{-1})	0.59 (2) [17]	Min: 0.47 Max: 0.70	0.1 (1) [18]	-	0.23 (3) [19]	Min: 0.19 Max: 0.28
Water regulation	Nitrogen concentration (mg L^{-1})	0.3 (1) [20]	-	0.39 (1) [21]	-		
Coastal erosion regulation	Percentage of wave attenuated						
Groundwater recharge	Groundwater recharge rate (mm km^{-2} year^{-1})	429.57 (1) [22]	-			74.17 (3) [23]	Min: 0.25 Max: 2.03x10^3
Wastewater treatment	Percentage of nutrients absorbed during wastewater discharge						
Disease regulation	Disease vectors predators						
Soil quality regulation	Index of soil quality						
Pest regulation	Presence of pests						
Pollination	Cost of bees						
Natural hazards regulation	Mortality losses during cyclone						
Nutrient cycle	Yearly rate of nitrogen storage (kg km^{-2} year^{-1})					1.69x10^5 (1) [24]	-
Cultural							
Aesthetic/ethical values	Yearly no. visitors for sightseeing						
Recreation and ecotourism	Yearly no. visitors for recreation	2 619.50 (6) [25]	Min: 84 Max: 1x10^4	2 736 (1) [26]	-	22 659 (6) [27]	Min: 10 Max: 9.58x10^4
Spiritual and religious values							
Cultural heritage	No. households considering area heritage						
Scientific/ education							
Supporting							
Primary production	NPP (kg km^{-2} year^{-1})	1.23x10^6 (1) [28]	-	3.3x10^5 (2) [29]	Min: 2.40x10^5 Max: 4.20x10^5	1.07x10^{16} (15) [30]	Min: 1.05x10^6 Max: 5.37x10^{16}
Nutrient flow	Yearly nitrogen flow (kg km^{-2} year^{-1})					6.67x10^3 (1) [31]	
Water cycling	Transpiration/total evapotranspiration						
Habitat heterogeneity	Habitat diversity index	0.46 (4) [32]	Min: 0.12 Max: 0.84	0.12 (4) [32]	Min: 0.04 Max: 0.16	0.38 (4) [32]	Min: 0.06 Max: 1.14^7
Nursery area	No. species with juveniles by total of species	0.68 (3) [33]	Min:0.56 Max: 0.77	0.54 (1) [34]	-		

*only data available quantified big groups, not species. In green indicators with no data available; orange indicators whit low or medium classification; blue ecosystem services without indicators; gray ecosystem services not relevant in the reference ecosystem. [1] (Blaber, 2002; Feka and Manzano, 2008; UN-REDD, 2014), [2] (Baran, 2000; Blaber, 2002; Ecoutin et al., 2010; Lamptey and Ofori-Danson, 2014), [3] (Feka and Ajonina, 2011), [4] (Lalèyè, 2000), [5] (FAO fisheries and aquaculture department, 2017a; FAO fisheries and aquaculture department, 2017b; FAO fisheries and aquaculture department, 2017c), [6] (Mmom and Arokoyu, 2010; Satyanarayana et al., 2012), [7] (Asibey, 1974; Falconer and Koppell, 1990; Jamnadass et al., 2015) [8] (Komiyama et al., 2008), [9] (Heubach et al., 2011; Heubes et al., 2012; INE são tomé e príncipe, 2014; Ndoye and Tieguhong, 2004; Nkem et al., 2010; Schaafsma, 2012) [10] (Feka and Ajonina, 2011), [11] (Brocard et al., 1998; de Faria et al., 2014; Kersten et al., 1998), [12] (Payet and Obura, 2004), [13] (Joint Research Center and NCEA, 2016), [14] (Ajonina et al., 2014b; Njana et al., 2018; UN-REDD, 2014), [15] (Mitsch et al., 2010), [16] (Lal, 2005; Sanogo et al., 2016), [17] (Khiteka and Kitheka, 1998; Roger et al., 2011), [18] (Li et al., 2005), [19] (Anayah et al., 2013; Chapas and Rees, 1964; Nizinski et al., 1994), [20] (Kamau, 1998), [21] (Bah et al., 2019; Healey et al., 1988), [22] (Roger et al., 2011), [23] (Anayah et al., 2013; Edmunds, 1990; Takounjou et al., 2011), [24] (Vitousek and Sanford, 1986), [25] (Ajonina et al., 2014a; Mallon et al., 2015; UN-REDD, 2014; UNEP, 2011), [26] (Mallon et al., 2015), [27] (Agyeman et al., 2019; Mallon et al., 2015) [28] (Alongi and Mukhopadhyay, 2015), [29] (Knoppers, 1994), [30] (Clark et al., 2001; Folega et al., 2015; Lo Seen Chong et al., 1993), [31] (Adepetu and Corey, 1977), [32] (Joint Research Center, 2018), [33] (Adite et al., 2013; Gajdzik et al., 2014; Kimani et al., 1996; Wright, 1986) [34] (Louca et al., 2009)

associated with aspects such as conservation status. *Recreation and ecotourism* is an important ES provided by the largest and best preserved mangrove of the island of São Tomé (Malanza), representing an important source of income to different stakeholders at the community level (Afonso, 2019). ES provided by mangroves are likely to vary from site to site. For example, mangroves in Thailand are well known for providing coastal protection, with an estimated economic value of nearly $6.4 US m^{-1} year^{-1} (Sathirithai, 1998). In São Tomé, most mangroves are in inner basins, not directly exposed to the coastal dynamics, and occupy a very small percentage of the coastline. Thus, mangrove ES flow will be strongly context dependent.

Challenges in Assessing Ecosystem Services

The selection of an adequate classification system is an important step for the assessment of ES, and at the same time a challenge, since the quality of classification systems is inherently subjective (Caputo et al., 2019). The MEA classification system (MEA, 2005) was selected because it is a widely cited and well-known approach. It is the most used in global and regional assessments, and it is widely used by the scientific community, facilitating comparisons between studies. However, it has some limitations, namely those related to the simplification of extremely complex interactions. Recent studies ponder the use of only 'final services' to avoid considering processes as services because the value of end-products already includes the processes and components of the ecosystem needed for its production (Haines-Young et al., 2018). Therefore, the description of ES must integrate multiple concepts, such as ecosystem structure and composition, to facilitate the conceptualization of ES (Wallace, 2007) and the assessment of benefits to people (Raudsepp-Hearne et al., 2010). Like most ES classification systems, MEA does not consider the particularities of the marine systems (Liquete et al., 2013). This may add an additional layer of difficulty because ES lists often need to be adapted regionally.
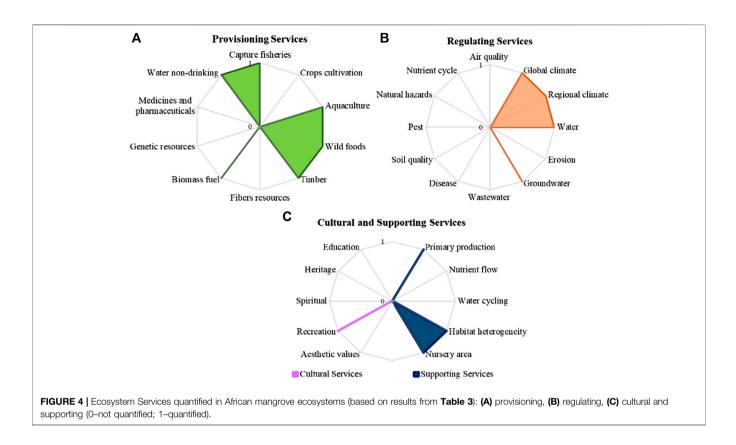
Natural ecosystems provide many services that can be associated with ecological functions, and at least potentially, with an important revenue stream, even if this might not be recognized by the local community. Major challenges in assessing ES are related to the identification of regulating and supporting services. Their identification is seldom straightforward and, generally, they do not provide direct products. For instance, *air quality regulation*, delivered by many ecosystems, is a well-known service among the scientific community and its functional relevance, in terms of pollutants removal, has been clearly proven, especially in areas of high urbanization and increased population (MEA, 2005). The assessment of ES allows the recognition of some services that are difficult to identify, for example evaluating air pollutant concentrations in areas with or without specific ecosystems. Furthermore, it can change the ecosystem's valuation since it pinpoints different qualities relevant to effective ecosystem functioning that are essential to humans.

The quantification of ES is a complex process and a challenge already recognized in the scientific literature (Martinez-Harms et al., 2015; Newton et al., 2018). Preferentially, the indicators should represent a realistic value of ES flow, rather than service capacity, which considers the total value of the service that cannot be regularly used in its fullness. Several difficulties are associated with the selection of the most complete and appropriate indicators. In general, many adaptations are needed to capture the indicator flow. In many cases, only simple adaptations are needed, such as adjusting spatial and temporal units are needed since the temporal and spatial inadequacy were the most listed unresolved issues in the literature (Rivero and Villasante, 2016). Although the aim was to biophysical quantify every ES, it was not possible since some indicators were only expressed in economic value, implying the need to continue to find indicators that can capture the biophysical value of the service, before measuring the economic value to society. Many hundreds of indicators are available, as a consequence of the development of global and regional biodiversity targets (Vihervaara et al., 2017). Biophysical assessments can identify sources of benefit that can be helpful for local communities, thus a biophysical assessment should always be complemented with socio-economic information.

No data was available to quantify most mangrove and estuarine systems ES in tropical Africa (13 out of 27 and 16 out 25 ES, respectively), while terrestrial ES are more commonly quantified (9 out of 22 ES). São Tomé mangroves are reported in many global lists, but they are generally excluded from analyses due to lack of data (Hamilton and Casey, 2016). As already identified in this study, limitations of data availability are an obvious issue for tropical African coastal ecosystems (Adekola et al., 2015; Rivero and Villasante, 2016), hampering our ability to quantify ES in this region. Furthermore, most of the indicators were developed to assess status and trends in biodiversity and ecosystem integrity, but not directly to evaluate ES (Feld et al., 2010). Indicators cannot quantify the ES reliably if they are unsuitable to evaluate the quality and quantity of ES benefits (Layke et al., 2012). Establishing reliable and useful indicators is key to evolve in the quantification of ES since the lack of available data can lead to a biased overvaluation of better-known services, ecosystems, and regions, and that no data is assumed as no benefit. The *disease regulation* service is an example of a less studied service (Liquete et al., 2016), and thus, it has fewer available indicators. Additionally, many cultural services indicators are considered generic or underdeveloped (Hattam et al., 2013). This strongly limits the quantification of the number of people benefiting from the service (Hein et al., 2006).

Improving Mangrove Management and Empowerment of Local Communities in Developing Countries

Mangroves are highly threatened ecosystems (Gilman et al., 2008), especially due to urban expansion in coastal areas (Nfotabong-Atheull et al., 2013). São Tomé is a developing country where 32,3% of the population is living below the international poverty line ($1.9 in purchasing power parity term - Conceição et al., 2019), and population density is high, pushing people toward unsustainable

FIGURE 4 | Ecosystem Services quantified in African mangrove ecosystems (based on results from Table 3): (A) provisioning, (B) regulating, (C) cultural and supporting (0–not quantified; 1–quantified).

use of resources (Samarakoon, 2004). To make matters worse, ecosystem deterioration can reduce the delivery of ES in the long-term, further increasing the risk of poverty traps (Uchida et al., 2019). This is particularly important when people are not aware of all the threats that may affect mangroves, which is the case of São Tomé (Afonso, 2019). Developing countries typically have inadequate institutional "safety nets", forcing communities to choose between satisfying their short-term needs or long-term sustainability (Dawson et al., 2010; Uchida et al., 2019). This combination of factors creates a positive feedback loop, that can trap human societies and ecosystems in a downward spiral of ever-worsening conditions. The importance of protecting mangrove areas has been discussed with the inhabitants of communities in the vicinity of mangroves included in the Obô Natural Park (Pisoni et al., 2015). However, most locals were less aware of its boundaries and claimed to be reliant on resources taken from the park.

The ESF can be a tool to counteract this loop since it can be used to satisfy human needs and environmental sustainability (Poppy et al., 2014). By identifying and quantifying existing ES, as done herein for the mangroves of São Tomé, the ESF is also a useful and innovative tool to guide management decisions and to weigh alternatives, when compared to more traditional management tools (Martinez-Harms et al., 2015). Furthermore, it has the potential to contribute to conservation goals, developing informed decision-making, adding value to protected areas, and creating opportunities to sustainably manage ecosystems. However, some caution is needed, since incomplete assessments can undervalue endemic or threatened species with functional roles that are harder to evaluate or merely understudied (Ingram et al., 2012).

The ESF facilitates the assessment of ecosystems at different scales and contributes to the evolving knowledge of mangrove ES, especially in small mangroves where local surrounding communities demonstrate difficulties in recognizing the importance of maintaining natural ecosystems. This study provides an important contribution to identifying specific conservation measures in these ecosystems. Evaluating and comparing ES provided by mangroves at different scales also contributes to standardizing the process of ES identification and quantification for easier comparison between mangroves at similar scales. In places where assessment and conservation measures are already being implemented for mangrove use, is it now recognized that a balance between local needs and sustainable use of resources is key to achieve mangrove conservation (Satyanarayana et al., 2013). This is the case of Sri Lanka. Most importantly, despite the many challenges for effective implementation, it can help bring awareness about our reliance on ecosystems and the fragility of ecosystem functioning, facilitating wider societal engagement in environmental conservation. ESF is even more important at locations where ES are of most immediate benefit to local (and often poor) people and might be strongly affected by global changes such as climate change, sea-level rise and the consequent effects of ocean acidification, warming, salinity, and hypoxia on fisheries.

TABLE 4 | Ecosystem services identified in mangroves of São Tomé. Source of information: literature review (adapted from Pisoni et al., 2015) and field work. Black circles and white circles represent, respectively, the presence and absence of each ecosystem service.

	Ecosystem Services	São Tomé Mangroves	
		Literature review	Field work assessment
Provisioning	Capture fisheries	●	●
	Crops cultivation	○	○
	Aquaculture	○	○
	Wild foods	●	●
	Timber	●	●
	Fibers and ornamental resources	○	○
	Biomass fuel	●	●
	Genetic resources	○	○
	Medicines and pharmaceuticals	●	●
	Water for non-drinking purposes	○	●
Regulating	Air quality	○	●
	Global climate regulation	●	●
	Regional climate regulation	○	●
	Water regulation	○	●
	Erosion regulation	●	●
	Groundwater recharge	○	●
	Wastewater treatment	●	●
	Disease regulation	○	●
	Soil quality	○	●
	Pest regulation	○	●
	Pollination	○	●
	Natural hazards regulation	●	●
	Nutrient cycle	○	●
Cultural	Aesthetic/ethical values	●	●
	Recreation and ecotourism	●	●
	Spiritual values	○	○
	Cultural heritage	○	○
	Scientific	○	●
Supporting	Primary Production	○	●
	Nutrient flow	○	●
	Water cycling	○	●
	Habitat heterogeneity	○	●
	Nursery area	●	●

TABLE 5 | Ecosystem Services quantified in São Tomé Mangroves and values for comparison with mangroves in the tropical African region (indicators source: **Table 1**).

Ecosystem Service	Indicator	Mangroves	
		São Tomé	Tropical African region
Provisioning Wild foods	Number of wild species used as food	27.00*	13.50
Regulating Water regulation	Nitrogen concentration (mg N L^{-1})	Min: 0.01	0.30
		Max: 0.02	
Supporting Nursery area	Number of species with juveniles by total of species	Min: 0.44 Max: 0.79	0.68 Min: 0.56 Max: 0.77

* Value from all mangroves in São Tomé (Pisoni *et al.*, 2015)

*value from all mangroves in são tomé (Pisoni et al., 2015)

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/supplementary files.

AUTHOR CONTRIBUTIONS

Conceptualization and design of study, AB, FA, PF, and RL; Data Collection, AB, FA, FR, JH, PC, PF, and RL; Data analysis and interpretation, AB, FA, PF, and RL; Writing and preparation of original draft, FA; Funding acquisition, PF. All authors contributed to manuscript revision, read and approved the submitted version.

FUNDING

ACKNOWLEDGMENTS

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