Challenges of Quantification and Valuation of Biological Assets: Case of East African National Parks in Tanzania, Kenya, and Uganda

Prof Dr.Dr.Lucky Yona

The Business School, Eastern and Southern Africa Management Institute
P.O.Box 3030, Arusha, Tanzania

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Abstract

Biological assets in East African national parks, including wildlife, forests, and ecosystems, are critical for ecological balance, tourism revenue, and socio-economic development. However, quantifying and valuing these assets face significant challenges due to methodological inconsistencies, data gaps, and the complexity of non-market ecosystem services. This study examines nine protected areas in Tanzania, Kenya, and Uganda using a desk-based mixed-evidence synthesis: quantitative values are drawn from reported secondary statistics and illustrative calculations, while qualitative evidence is synthesized from policy documents and published case studies. The findings therefore present reported, derived, and proxy values rather than a fully harmonized natural-capital account. The review identifies unreliable baseline data, valuation bias, weak source traceability, and political barriers, and recommends staged implementation of the SEEA Ecosystem Accounting framework, transparent valuation protocols, sensitivity analysis, and direct community participation in future primary research.

Keywords: Biological Assets, Quantification, Valuation, National Parks

1. Introduction

Biological assets in East African national parks, encompassing wildlife, forests, rivers, and grasslands, represent a cornerstone of natural capital, supporting biodiversity, ecosystem services, and economic development. Unlike agricultural or livestock assets valued for direct financial returns, national park assets provide diverse benefits, including tourism revenue, carbon sequestration, watershed protection, and cultural heritage. In Tanzania, Kenya, and Uganda, national parks such as Serengeti, Ngorongoro Crater, Tarangire, Maasai Mara, Amboseli, Tsavo, Bwindi Impenetrable Forest, Queen Elizabeth, and Murchison Falls are global biodiversity hotspots that contribute significantly to national economies and global conservation goals. However, quantifying and valuing these assets is challenging because of their non-market nature, dynamic ecosystems, and data limitations.

Valuing biological assets is critical for integrating natural capital into policy and financial decision-making, aligning with global frameworks like the United Nations System of

Environmental-Economic Accounting (SEEA) and the Economics of Ecosystems and Biodiversity (TEEB) initiative. Accurate valuations inform conservation financing, such as carbon credits and biodiversity offsets, and support sustainable development. Yet challenges include methodological inconsistencies, a lack of reliable baseline data, and difficulty valuing non-use services like biodiversity existence value. These barriers hinder policymakers' ability to prioritize conservation investments and equitably distribute benefits to local communities.

This study is justified by the urgent need to address valuation challenges to enhance conservation and economic outcomes in East Africa. National parks in Tanzania, Kenya, and Uganda generate billions in economic value but face threats from poaching, habitat loss, and climate change. Existing valuation frameworks, such as International Accounting Standard (IAS) 41, are ill-suited for non-commercial assets, necessitating tailored approaches. By examining these challenges, the paper aims to provide actionable insights for policymakers, conservationists, and researchers to strengthen natural capital accounting and support the Sustainable Development Goals (SDGs), particularly SDG 15 (Life on Land).

These three countries were selected because their national parks are globally significant and exemplify diverse biological assets and economic contributions. Tanzania's Serengeti National Park hosts the iconic wildebeest migration, Ngorongoro Crater is a UNESCO World Heritage site known for its caldera ecosystem, and Tarangire National Park is renowned for its elephant populations and baobab trees, collectively generating substantial tourism revenue. Kenya's Maasai Mara is famous for its wildlife and migration continuation, Amboseli National Park for its elephant herds and Mount Kilimanjaro views, and Tsavo National Park for its vast landscapes and red elephants. Uganda's Bwindi Impenetrable Forest protects mountain gorillas, Queen Elizabeth National Park features diverse savannah wildlife, and Murchison Falls National Park includes the Nile River and waterfalls. These parks face similar valuation challenges, including data gaps and non-market valuation complexities, but differ in governance structures and conservation approaches, offering a comparative perspective. Additionally, their regional proximity and shared ecosystems (e.g., Serengeti-Maasai Mara) make them ideal for studying harmonized valuation strategies.

2. Literature Review

Valuing biological assets in national parks requires a clear separation between the object being measured and the benefit being valued. In this paper, a biological asset is a living plant or animal; an ecosystem asset is an ecosystem whose extent, condition, and functioning generate benefits; natural capital is the stocks of those assets and their capacity to generate flows; and ecosystem services are the flows of contributions from ecosystems to people. Total Economic Value (TEV) is a welfare-oriented framework that organizes use and non-use values; it is not a balance-sheet measure of the natural-capital stock. The distinction matters because a park's carbon stock, annual carbon-sequestration service, tourism receipts, and existence value are distinct objects and should not be added unless their boundaries and time bases are explicit. IAS 41 was designed primarily for managed agricultural biological assets and does not provide a complete accounting treatment for protected-area ecosystems. The SEEA Ecosystem Accounting framework is better suited to organizing ecosystem extent, condition, service supply and use, and, where data permit, monetary asset accounts (King et al., 2023; United Nations et al., 2021).

2.1. Theoretical Frameworks

The Total Economic Value (TEV) framework organizes direct-use, indirect-use, option, and non-use values. It is useful for structuring welfare analysis, but it does not imply that all components can be measured with the same data or summed without boundary checks. In contrast, the SEEA Ecosystem Accounting framework records ecosystem extent and condition, physical and monetary ecosystem-service flows, and monetary ecosystem assets based on the present value of expected future service flows. Recent protected-area applications in South Africa and Uganda show how the framework can consolidate disparate data while preserving distinctions between physical accounts, service flows, and monetary accounts (King et al., 2023). Recent East African evidence also shows that valuation studies remain heterogeneous: a systematic review of 222 studies found provisioning services dominated the evidence base, while market-price, travel-cost, remote-sensing, and participatory methods were used for different service types (Osewe et al., 2024). A Sub-Saharan African review of 154 studies similarly found a near-balanced use of participatory and non-participatory approaches and recommends integrating them rather than treating either as sufficient alone (Ali et al., 2025).

2.2. Empirical Studies

Empirical studies reveal significant challenges in quantifying and valuing biological assets. In Tanzania, Serengeti National Park’s wildebeest migration generates USD 250 million annually in tourism revenue, yet its carbon storage (100 million tons) is undervalued due to inconsistent carbon pricing (Tanzania Tourist Board, 2022; World Bank, 2021). Sinclair et al. (2015) note that wildlife censuses in the Serengeti are irregular, with data gaps affecting population estimates. Ngorongoro Crater’s ecosystem supports diverse wildlife valued at USD 200 million in tourism, but land-use conflicts reduce accuracy (Koskei and Glyptou, 2025). Tarangire’s elephant populations contribute USD 150 million in biodiversity value, according to the TAWIRI census (2024). In Kenya, Maasai Mara’s lion populations generate USD 1.2 million per lion in lifetime tourism value, yet poaching valuations are significantly lower, highlighting conservation trade-offs (Lindsey et al., 2021). Okello and Novelli (2014) found that Maasai Mara’s tourism revenue distribution often bypasses local communities, complicating valuation due to socio-political factors. Amboseli’s elephants and wetlands are valued at USD 80 million in ecosystem services, according to the WRTI census (2024-2025) (WRTI, 2024). Tsavo’s red elephants generate USD 120 million in tourism, but oil operations impact valuations (WRTI, 2025).

In Uganda, Bwindi Impenetrable Forest’s gorilla trekking generates USD 34 million annually, but non-use values, such as global WTP for gorilla conservation, are harder to quantify (Adams and Infield, 2003). Tumusiime and Vedeld (2015) report that contingent valuation studies in Bwindi yield inconsistent WTP estimates due to hypothetical bias. Similarly, carbon storage valuations in Bwindi rely on global carbon markets, which may not reflect local economic realities (UNEP, 2020). Queen Elizabeth National Park's savannah wildlife contributes USD 50 million to tourism, according to the Uganda Wildlife Authority (2023), but fire impacts reduce ecosystem values (WCS Uganda, 2024). Murchison Falls' Nile River and megaherbivores are valued at USD 70 million, with oil operations posing threats (Global Conservation, 2024). These studies underscore the challenge posed by unreliable baseline data, as wildlife censuses and ecosystem mapping are costly and infrequent across all three countries.

Methodological challenges are also evident. The Travel Cost Method (TCM) used in the Maasai Mara and Amboseli studies requires detailed visitor expenditure data, which are scarce (Ogutu et al., 2016). Contingent Valuation (CV) methods, applied in Bwindi and Queen Elizabeth, are prone to bias because respondents' WTP varies by context (Carson and Hanemann, 2005). Discounted Cash Flow (DCF) models, used for long-term valuations in Serengeti and Tsavo, are sensitive to discount rate assumptions, leading to valuation discrepancies (World Bank, 2018). Benefit Transfer methods, used in cross-country comparisons such as Ngorongoro and Murchison Falls, often fail to account for local ecological and cultural differences (Brander et al., 2012).

2.3. Regional Challenges

In East Africa, institutional and political barriers worsen valuation challenges. Adams and Infield (2003) highlight that community perceptions of unfair benefit-sharing in Bwindi and Queen Elizabeth cause conflicts that hinder conservation. In Tanzania, governance issues in Serengeti and Ngorongoro limit the adoption of SEEA frameworks; capacity gaps hinder data collection (Visseren-Hamakers et al., 2015; Koskei and Glyptou, 2025). Similarly, Kenya's Maasai Mara and Amboseli have fragmented land ownership, making ecosystem valuation more challenging (Homewood et al., 2012; WRTI, 2024). Climate change and poaching are also decreasing asset value, with annual losses in these parks estimated at 10-20% (Lindsey et al., 2021).

2.4. Gaps in Literature.

The literature continues to show three relevant gaps. First, protected-area studies in East Africa rarely present a complete chain from ecosystem extent and condition to physical service flow, monetary service flow, and asset value. Second, methods and reporting conventions remain heterogeneous, limiting comparison across parks; a recent Africa-wide review found that single-method valuations and inconsistent policy-impact reporting remain common (Akello et al., 2025). Third, local perspectives are often absent or represented indirectly. The current study addresses these gaps only at the level of a secondary synthesis. It does not claim to have conducted new household surveys, interviews, or participatory mapping; instead, it identifies where primary community research is needed and recommends that future accounts combine biophysical and monetary evidence with participatory and deliberative methods.

3. Methodology

This paper uses a desk-based mixed-evidence design rather than a conventional primary-data mixed-methods study. The quantitative strand extracts reported wildlife, tourism, carbon, ecosystem-service, and monetary figures from secondary sources and reproduces only those illustrative calculations for which the manuscript states the inputs. The qualitative strand synthesizes statements in published studies, policy documents, and institutional reports concerning governance, benefit-sharing, and community perceptions. No new respondents, interviews, focus groups, household surveys, or field measurements were undertaken. The design is therefore best understood as a secondary mixed-evidence review; the term “mixed methods” is retained only to indicate the integration of quantitative extraction with qualitative document analysis.

3.1. Desktop Study

The desktop review examined peer-reviewed articles, government and institutional reports, and international datasets cited in the manuscript, with an intended publication window of 2010–2025. Because the supplied study record does not report database names for every search, search strings, the number of records screened, or a formal inclusion/exclusion flow, the review should not be interpreted as a systematic review or meta-analysis. Each reported figure is treated as a secondary claim requiring source verification. Where the underlying report, dataset, observation year, geographic boundary, or unit is not available in the supplied materials, the figure is retained only as an attributed estimate and is not presented as independently measured by this study.

3.2. Triangulation of Literature

Triangulation was conducted by comparing three evidence types: peer-reviewed research for concepts and valuation limitations; institutional or government reports for reported park statistics; and international frameworks for accounting definitions and method selection. The comparison was documentary rather than statistical. Sources were coded for park, value category, measurement year, unit, valuation method, and evidence status (reported, derived, proxy, or method not specified). Disagreement between sources was not resolved by averaging; instead, it is reported as uncertainty and used to caution against direct cross-park ranking.

3.3 Quantitative Methods

Quantification used the following observable or reported indicators: wildlife abundance, tourism receipts or permit revenue, carbon stock, and descriptions of watershed, soil, wetland, fire-regulation, or riverine services. The unit of analysis is an annual flow in USD million unless otherwise stated. Wildlife counts and carbon stocks are physical quantities; tourism receipts are market transactions; and ecosystem-service and non-use amounts are monetary estimates. The data do not constitute a single synchronized accounting year, and the manuscript does not provide the park area, visitor counts, price year, exchange-rate convention, survey sample sizes, or raw spatial layers needed for complete reproduction. These omissions are reflected in the evidence-status column added below.

Valuation was organized as follows. Direct-use estimates were treated as market-price or reported-revenue measures, principally tourism receipts and permits. Where a travel-cost or contingent-valuation study was cited, its result was treated as a secondary estimate rather than

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recomputed because respondent-level data are unavailable. Indirect-use values were treated as service-flow proxies: carbon was calculated as physical carbon stock multiplied by an assumed carbon price; watershed, soil, wetland, fire-regulation, and riverine services were described as replacement-cost, avoided-cost, or benefit-transfer estimates only where the source or the manuscript stated that method. Non-use values were treated cautiously because a valid willingness-to-pay aggregate requires a defined population, elicitation instrument, sampling frame, payment vehicle, and treatment of hypothetical bias. Donations and grants are not equivalent to willingness-to-pay and are labeled as financial proxies. For any future DCF application, the present value should be calculated as $PV = \Sigma(F\_t/(1+r)\^t)$, with the flow, horizon, discount rate, growth assumption, and terminal treatment reported explicitly. The Table 1 figures are annualized category estimates, not DCF asset values.

3.4. Qualitative Methods

Qualitative analysis coded evidence on governance, benefit-sharing, community perceptions, and non-market values from secondary case studies and policy documents. The study did not collect local perspectives directly. Accordingly, statements about communities are representations reported by previous authors or institutions, not findings from new consultation. The synthesis records whether a source reports perceived benefits, costs, conflicts, or participation, but it cannot establish representativeness across all communities surrounding the nine parks. A future mixed-methods study should add stratified household surveys, interviews, focus groups, or participatory mapping with transparent consent and sampling procedures.

3.5. Limitations

The principal limitation is reliance on secondary data that may be outdated, incomplete, differently bounded, or based on incompatible valuation methods. The source record does not provide sufficient raw data to independently reproduce every cell in Table 1. In addition, several monetary estimates are described only by an institutional or author-year citation, without the underlying dataset, sample, spatial boundary, price year, or calculation. The revised tables therefore distinguish measured or reported figures from derived and proxy values. Sensitivity analysis is illustrative and one-way; it demonstrates the effect of major assumptions but does not replace park-specific uncertainty intervals or a full probabilistic model.

4. Research Findings

The secondary evidence indicates that the nine parks provide substantial tourism, regulating, cultural, and biodiversity-related benefits. However, the evidence is not a harmonized natural-capital account. Tourism figures are generally reported as market or revenue estimates; carbon and other regulating-service amounts are model-based or replacement-cost proxies; and non-use amounts are incompletely documented proxies. The comparative figures below should therefore be read as an organized synthesis of claims in the supplied sources, not as directly comparable measurements or as the monetary value of the parks’ biological stocks.

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For Serengeti, the manuscript reports approximately 1.5 million wildebeest and annual tourism revenue of USD 250 million. The carbon figure is an explicitly derived calculation: 100 million tonnes of carbon multiplied by an assumed price of USD 5 per tonne gives USD 500 million. The reported USD 150 million non-use amount is retained as an estimate, but the supplied manuscript does not disclose the contingent-valuation instrument, respondent population, sample size, aggregation rule, or how protest responses were treated. These limitations mean that the three components are not equally measured and should not be interpreted as a single audited total.

The carbon calculation is sensitive to the price assumption and to whether the stated 100 million tonnes refers to total stock, annual sequestration, or an accounting boundary that overlaps with other ecosystems. At USD 5, USD 10, and USD 20 per tonne, the same stock would imply USD 500 million, USD 1,000 million, and USD 2,000 million, respectively. These are gross price-sensitivity illustrations, not estimates of creditable offsets or net social cost of carbon, because permanence, additionality, leakage, transaction costs, and the distinction between stock and annual flow are not reported.

For Ngorongoro, the manuscript reports USD 200 million in annual tourism revenue, USD 300 million in ecosystem services, and USD 100 million in non-use or cultural-heritage value. The tourism amount is attributed to the cited secondary record, while the indirect and non-use amounts are not accompanied by a reproducible park-level dataset or a disclosed aggregation formula. They are therefore classified as proxies in the supplementary source table. The multiple-use setting also means that tourism, pastoralism, watershed functions, and cultural values require explicit spatial and beneficiary boundaries before they can be summed.

For Tarangire, the reported USD 150 million direct-use amount, USD 250 million ecosystem-service amount, and USD 80 million non-use amount are retained only as attributed estimates. The supplied manuscript does not identify the visitor series, price year, ecosystem-service biophysical quantity, unit cost, WTP study, or source for the non-use figure. The table consequently labels the first amount as a reported market-price estimate and the latter two as proxy or method-not-specified values rather than measured park accounts.

For Maasai Mara, the manuscript reports USD 100 million in direct tourism revenue, USD 200 million in watershed and carbon-related ecosystem services, and USD 80 million in non-use value. These amounts are not directly comparable with the Serengeti figures because the geographic boundaries, treatment of surrounding conservancies, visitor expenditures, carbon accounting, and valuation methods differ. The lion lifetime figure of USD 1.2 million is a species-level tourism-attraction estimate and should not be added to the reserve subtotal; it is a separate illustrative comparison with an illegal-trade value and does not measure total economic value.

The reported comparison between a lion’s lifetime tourism value and its illegal-trade value is useful only as an illustration of alternative economic incentives. It is not a park-level valuation,

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and the two figures measure different objects, time horizons, risks, and markets. The comparison should therefore not be interpreted as evidence that tourism revenue automatically accrues to local communities or as a direct estimate of avoided poaching losses.

For Amboseli, the manuscript reports USD 80 million in tourism, USD 150 million in ecosystem services, and USD 60 million in non-use value. The direct-use figure is attributed to the cited WRTI record, whereas the indirect and non-use figures lack a transparent park-level calculation in the supplied materials. They are retained as proxies and should not be compared with parks for which carbon stocks or visitor receipts were measured using different boundaries.

For Tsavo, the reported USD 120 million direct-use amount, USD 180 million indirect-use amount, and USD 70 million non-use amount are treated as secondary estimates. The manuscript does not provide the underlying tourism series, riverine-service model, carbon or hydrological quantity, or non-use elicitation design. The oil-operation discussion is a risk context, not a quantified deduction from the stated values.

For Bwindi, the manuscript reports approximately 400 mountain gorillas, a USD 700 permit price, USD 34 million in annual gorilla-tourism revenue, USD 50 million in ecosystem services, and USD 30 million in non-use value. The permit price is observed, but the USD 34 million figure also requires the number and composition of permit holders, which are not provided. The ecosystem-service and non-use amounts are estimates or financial proxies; international donations and grants should not be treated as a population-wide WTP measure without a stated valuation design.

For Queen Elizabeth, the reported USD 50 million tourism amount, USD 100 million fire-regulation amount, and USD 40 million non-use amount are retained as attributed estimates. The tourism figure is closer to a reported market-flow measure; the regulating and non-use figures are proxies because the supplied manuscript does not provide a fire-avoidance model, beneficiary population, WTP survey, or aggregation formula.

Table 1 organizes the reported category estimates, but its final column is labeled an arithmetic subtotal rather than “Total Economic Value.” The subtotal is the sum of three heterogeneous annual estimates and is not a strict TEV measure or a SEEA monetary asset account. The comparison is descriptive only: different years, spatial boundaries, source quality, physical indicators, and valuation methods prevent a defensible rank order. The source-and-method table and sensitivity analysis make those differences explicit.

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Table 1: Comparative Valuation of East African National Parks (USD Million/Year

Park	Direct use (USD million/year)	Indirect use (USD million/year)	Non-use (USD million/year)	Arithmetic subtotal (USD million/year)
Serengeti (Tanzania)	250	500	150	900
Ngorongoro Crater (Tanzania)	200	300	100	600
Tarangire (Tanzania)	150	250	80	480
Maasai Mara (Kenya)	100	200	80	380
Amboseli (Kenya)	80	150	60	290
Tsavo (Kenya)	120	180	70	370
Bwindi (Uganda)	34	50	30	114
Queen Elizabeth (Uganda)	50	100	40	190
Murchison Falls (Uganda)	70	120	50	240
All parks (arithmetic sum)	**1,054**	**1,850**	**660**	**3,564**

Source note: Table 1 is an author synthesis of secondary figures cited in the manuscript, not a single dataset.

Table 2 documents the source, valuation method, calculation status, and principal assumption for each park/category cell. Reported market flows are not equivalent to measured ecological stocks; indirect-use and non-use entries are largely derived or proxy estimates. The arithmetic subtotal is not a strict TEV or SEEA monetary asset value, and no claim is made that all components are mutually exclusive until double-counting checks are completed.

Reproducing the calculations from the supplied manuscript is limited. Serengeti's carbon estimate is 100 million tonnes × USD 5 per tonne = USD 500 million. The park subtotals equal direct use + indirect use + non-use; for example, Serengeti totals 250 + 500 + 150 = USD 900 million per year. The aggregate row totals USD 1,054 million direct use + USD 1,850 million indirect use + USD 660 million non-use = USD 3,564 million per year. For the remaining cells, the manuscript reports values but does not provide enough information to reproduce the underlying tourism, hydrological, replacement-cost, benefit-transfer, or WTP calculations. Those cells are therefore labeled as reported estimates, proxies, or method-not-specified values rather than being attributed generically to "Researcher 2025."

Table 2: Data source, valuation method, and evidence status for each park and value category

Park	Value category	Estimate (USD million/year)	Data source and basis	Valuation method	Evidence status
Serengeti (Tanzania)	Direct use	250	Tanzania Tourist Board (2022); TAWIRI (2024), as cited; tourism receipts/fees	Market-price / reported revenue; annual flow	Reported estimate; visitor series and boundary not supplied
Serengeti (Tanzania)	Indirect use	500	100 million tonnes carbon stock; World Bank (2021) price assumption	Carbon stock × USD 5/t	Derived illustration; stock/flow and creditability not established
Serengeti (Tanzania)	Non-use	150	Contingent-valuation claim in supplied manuscript	Stated-preference/WTP claim; aggregation not disclosed	Proxy/estimated; sample and population not supplied
Ngorongoro Crater (Tanzania)	Direct use	200	Tanzania Tourist Board (2022), as cited in manuscript	Market-price / reported tourism revenue	Reported estimate; secondary source details incomplete
Ngorongoro Crater (Tanzania)	Indirect use	300	Koskei and Glyptou (2025); Nairobi Convention (2023), as cited	Water-regulation/carbon proxy; exact method not disclosed	Proxy; spatial boundary and unit costs not supplied
Ngorongoro Crater (Tanzania)	Non-use	100	Cultural-heritage appreciation claim in supplied manuscript	Stated-preference or donation proxy; method not disclosed	Proxy; no WTP instrument or aggregation rule supplied
Tarangire (Tanzania)	Direct use	150	TAWIRI (2024), as cited in manuscript	Market-price / reported tourism estimate	Reported estimate; tourism dataset not supplied
Tarangire (Tanzania)	Indirect use	250	World Bank (2021), as cited; soil-conservation claim	Replacement/ avoided cost suggested by methods section; calculation not disclosed	Proxy; biophysical quantity and unit cost not supplied

Tarangire (Tanzania)	Non-use	80	No underlying source identified in supplied manuscript	Method not disclosed	Method-not-specified proxy
Maasai Mara (Kenya)	Direct use	100	KWS (2023) and WRTI (2024), as cited; tourism/entry fees	Market-price / reported revenue	Reported estimate; boundary and secondary effects excluded
Maasai Mara (Kenya)	Indirect use	200	Lindsey et al. (2021); WRTI (2024), as cited; watershed/carbon	Avoided/replacement cost or benefit transfer; exact method not disclosed	Proxy; partial watershed/carbon data
Maasai Mara (Kenya)	Non-use	80	Lindsey et al. (2021), as cited; global biodiversity appreciation	Donation/WTP proxy; aggregation not disclosed	Proxy; donations are not equivalent to WTP
Amboseli (Kenya)	Direct use	80	WRTI (2024–2025), as cited	Market-price / reported tourism estimate	Reported estimate; visitor series not supplied
Amboseli (Kenya)	Indirect use	150	WRTI (2024), as cited; wetland water provision	Replacement/ avoided cost; calculation not disclosed	Proxy; hydrological and unit-cost data not supplied
Amboseli (Kenya)	Non-use	60	Underlying source not identified in supplied manuscript	Method not disclosed	Method-not-specified proxy
Tsavo (Kenya)	Direct use	120	KWS (2023) and WRTI (2025), as cited	Market-price / reported tourism estimate	Reported estimate; source details incomplete
Tsavo (Kenya)	Indirect use	180	WRTI (2025), as cited; riverine habitats	Replacement/ avoided cost or benefit transfer; exact method not disclosed	Proxy; river-service model not supplied
Tsavo (Kenya)	Non-use	70	Underlying source not identified in supplied manuscript	Method not disclosed	Method-not-specified proxy
Bwindi (Uganda)	Direct use	34	UWA (2023); Gorilla Fund (2025), as cited; USD 700 permit price	Permit price × implicit permit	Partly derived/reported; not fully

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				volume; volume not supplied	reproducible
Bwindi (Uganda)	Indirect use	50	UNEP (2020), as cited; carbon and watershed	Replacement/ avoided cost or benefit transfer; exact method not disclosed	Proxy; carbon mapping described as incomplete
Bwindi (Uganda)	Non-use	30	Adams and Infield (2003), as cited; donations/grants	Financial-flow proxy for non-use; not a WTP aggregate	Proxy; donation data and population not supplied
Queen Elizabeth (Uganda)	Direct use	50	UWA (2023); WCS Uganda (2024), as cited	Market-price / reported tourism estimate	Reported estimate; underlying series not supplied
Queen Elizabeth (Uganda)	Indirect use	100	WCS Uganda (2024), as cited; fire regulation	Avoided-cost proxy; calculation not disclosed	Proxy; fire-risk model and beneficiary costs not supplied
Queen Elizabeth (Uganda)	Non-use	40	Underlying source not identified in supplied manuscript	Method not disclosed	Method-not-specified proxy
Murchison Falls (Uganda)	Direct use	70	Global Conservation (2024), as cited	Market-price / reported tourism estimate	Reported estimate; source is not a park accounts dataset
Murchison Falls (Uganda)	Indirect use	120	Global Conservation (2024), as cited; riverine habitats	Replacement/ avoided cost or benefit transfer; exact method not disclosed	Proxy; hydrological model not supplied
Murchison Falls (Uganda)	Non-use	50	Underlying source not identified in supplied manuscript	Method not disclosed	Method-not-specified proxy

Sensitivity results (Table 3) are interpreted one assumption at a time. They show that carbon-price selection, discounting, tourism uncertainty, and non-use aggregation can materially change the monetary result; they do not provide a statistically estimated confidence interval. A park-by-park sensitivity analysis requires the underlying visitor, carbon, hydrological, and WTP datasets.

Table 3: One-way sensitivity analysis of selected assumptions (USD million)

Assumption	Base calculation	Base case	Lower-value case	Higher-value case	Interpretation
Serengeti carbon price	100m tonnes × price/t	USD 5/t → 500	USD 5/t → 500	USD 10/t → 1,000; USD 20/t → 2,000	Price sensitivity only; does not establish offsets or social cost of carbon
Aggregate direct-use values	Sum of nine reported direct-use estimates	1,054	−20% → 843.2	+20% → 1,264.8	Illustrative revenue uncertainty; not a confidence interval
Aggregate non-use proxies	Sum of nine non-use estimates	660	−50% → 330	+50% → 990	Wide uncertainty reflects undisclosed WTP/proxy designs
Illustrative 20-year DCF of USD 250m annual flow	PV = Σ(Ft/(1+r)^t)	5% → 3,115.6	7% → 2,648.5	3% → 3,719.4	Illustration only; not included in Table 1 and no growth/terminal value assumed

5. Discussion

The organized secondary evidence points to substantial conservation and economic importance, but the headline aggregate of USD 3,564 million per year should not be interpreted as a directly measured total economic value. It is an arithmetic sum of reported tourism estimates, indirect-use proxies, and non-use estimates assembled from different sources and years. The strongest evidence comes from reported market flows; the weakest comes from park-level non-use and regulating-service amounts, whose underlying calculations are not disclosed.

The main empirical limitation is traceability. The manuscript lacks a common accounting year, synchronized park boundaries, raw visitor and revenue series, carbon-stock metadata, hydrological observations, and WTP sample information. As a result, uncertainty is not merely statistical; it also arises from different definitions of the asset, service, beneficiary, and time period. A defensible regional account should first compile physical extent and condition accounts, then service supply-and-use tables, and only then monetary estimates, following the protected-area applications of SEEA EA described by King et al. (2023).

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The valuation methods are not interchangeable. Market-price estimates record transactions; travel-cost methods estimate recreational welfare; contingent valuation estimates stated preferences; replacement or avoided cost estimates the cost of maintaining or replacing a service; benefit transfer imports values from another context; and DCF converts future flows into a present value. These methods answer different questions and can double count when the same tourism experience includes wildlife existence, cultural meaning, and ecosystem quality. The revised source table therefore reports the method and evidence status for each cell, while the sensitivity table shows the effect of selected assumptions.

The evidence on communities is also secondary. Published case studies and institutional reports describe benefit-sharing concerns, land-use conflict, and human–wildlife conflict, but the present study did not interview residents or measure their preferences. Community perspectives should therefore be presented as reported contextual evidence rather than as a representative regional finding. Future accounts should combine household or livelihood surveys with participatory mapping and deliberative valuation, disaggregate groups by gender, livelihood, tenure, and proximity, and report how local knowledge changes the definition of beneficiaries and the valuation boundary.

Poaching, climate change, fire, land-use change, and extractive activity are important threats, but the supplied evidence does not support applying a uniform percentage loss to all park values. The effect depends on species, ecosystem condition, exposure, visitor demand, and the valuation method. These threats should be modeled as changes in physical extent, condition, service flows, or future revenues rather than subtracted from the aggregate using a single regional rate.

The appropriate policy implication is a staged regional accounting system, not an immediate ranking of parks by total monetary value. The first stage should standardize boundaries, accounting years, physical indicators, and source metadata. The second should compile ecosystem extent and condition accounts and physical service supply-and-use tables. The third should add monetary service flows using method-specific guidance, publish uncertainty and sensitivity ranges, and test for double counting. A regional data-sharing mechanism could then support trans boundary ecosystems, such as the Serengeti–Maasai Mara system, without forcing incomparable estimates into a single ranking.

Conclusion and Recommendations

The nine parks provide important tourism, regulating, cultural, and biodiversity benefits, but the evidence assembled here does not justify presenting USD 3,564 million per year as a measured aggregate of biological assets or as a complete TEV. The figure is an arithmetic subtotal of heterogeneous secondary estimates. The principal contribution of this review is therefore diagnostic: it identifies the data, boundary, valuation, and governance conditions that must be resolved before a comparable natural-capital account can be produced.

Natural-capital accounting should follow the SEEA EA sequence of ecosystem extent, ecosystem condition, physical service flows, monetary service flows, and, where defensible, monetary asset

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accounts. Park authorities and national statistical agencies should publish a metadata sheet for each account: geographic boundary, reference year, physical unit, source dataset, valuation method, price year, discount rate, uncertainty range, and beneficiary. The protected-area applications documented for South Africa and Uganda provide a useful model for organizing these accounts (King et al., 2023).

Data collection should prioritize repeatable wildlife monitoring, visitor and expenditure records, carbon-stock inventories with ground validation, and spatial data on watershed, wetland, soil, fire, and riverine functions. Each monetary estimate should be accompanied by its physical numerator and valuation denominator—for example, tonnes of carbon and price per tonne, visitor-days and expenditure per visitor, or avoided treatment cost and service quantity. Future primary research should add transparent household and stakeholder sampling rather than inferring community preferences solely from secondary reports.

Carbon credits, biodiversity finance, and conservation funds may support park management, but monetary estimates should not be treated as financeable revenue without tests of additionality, permanence, leakage, legal ownership, transaction costs, safeguards, and benefit-sharing. The sensitivity results show why carbon-price scenarios should be reported separately from the value of carbon stock and from any expected project revenue. Similar safeguards are needed for biodiversity offsets and tourism-linked finance.

Community participation should be incorporated as a primary research component in the next phase. Secondary evidence can identify reported grievances and institutional arrangements, but it cannot establish whose values are represented or how benefits and costs are distributed. Future studies should use participatory mapping, focus groups, household surveys, and deliberative valuation with transparent sampling and disaggregated reporting. Revenue-sharing claims should be measured from actual institutional flows and linked to local priorities rather than inferred from park-level tourism totals.

Regional cooperation should focus on a common data dictionary, harmonized park boundaries and accounting years, interoperable wildlife and land-cover monitoring, and a shared source-and-method registry. Comparisons should be made only after checking whether the same service, beneficiary, price basis, time period, and valuation method are being compared. This approach would improve the usefulness of natural-capital information for conservation and development while avoiding false precision.

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