10 THE ECONOMIC VALUE OF ELEPHANTS

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INTRODUCTION

ELEPHANTS PLAY a huge role within any landscape where they occur. They are habitat engineers. As charismatic species they awaken emotions among people like few others. As keystone species, they contribute significantly to the integrity of ecosystems and must be very carefully managed. From an economic perspective, they are also value generators. In this broad context, we first consider the range of relevant economic values, using the Total Economic Value approach in a generic sense, and then apply this framework to identify the specific factors that determine the economic value of elephants in South Africa. Thereafter we summarise both regional (southern African) and international studies that consider the economic value of elephants. We conclude with an assessment of the state of knowledge on elephants' contribution to the economic value of elephant-containing ecosystems and the economy as a whole.

This assessment borrows heavily from studies concerning the economic value of elephants carried out in Botswana, Namibia, and Zimbabwe, since similar studies in South Africa could not be located. To date, published studies in South Africa focused either on the cost of the individual elephant management options – which is a subject treated in the relevant management chapters of this book – or else investigations of the value of tourism. The specific contribution of elephants to the value of tourism was not isolated in these studies.

BACKGROUND ON ECONOMIC VALUE

Adam Smith, the 'father of modern economics', distinguishes between two types of economic values: exchange values and use values. He clarifies as follows (quoted from reprint in Smith, 1997, 131):

> The word VALUE … has two different meanings, and sometimes express the utility of some particular object, and sometimes the power of purchasing

> other goods which the possession of that object conveys. The one may be called 'value in use'; the other, 'value in exchange'. The things which have the greatest value in use have frequently little or no value in exchange; and, on the contrary, those which have the greatest value in exchange have frequently little or no value in use.

He explains the distinction between exchange and use value by referring to the well-known water-diamond paradox. Nothing is more useful than water, yet it has almost no exchange value. In contrast, diamonds have relatively little *real* use, but have extremely high exchange values. Exchange values are easy to observe. They are the market values of a product, good, or service. Use values, however, are not observed. If care is not taken one could easily ignore these use values when making decisions. The economic valuation of ecosystem goods and services is an attempt to mitigate the impact of either the absence of markets or the wrong signals markets send by estimating the value of natural capital in terms of what these resources contribute to society. Some are opposed to the quantification of the value of natural resources (McCauley, 2006), but most of these antagonists are ignorant about the way economists distinguish between the environment's use value and exchange value. Ecological economists are fully aware of the fact that it might not always be possible, or even necessary or desirable, to estimate the use value of a resource – especially when dealing with so-called *critical* natural capital (Ekins *et al.,* 2003; Farley & Gaddis, 2007; Blignaut *et al.,* 2007). Yet, by estimating the values that are deemed appropriate, economists acknowledge the fact that environmental values exist and that they contribute meaningfully and significantly to social welfare.

Figure 1 provides a breakdown of the suite of environmental values by first distinguishing between the primary and secondary value of the environment. Primary values – values without an economic purpose – are also called intrinsic values and reflect the non-demand values of ecosystems. In some instances, primary values could also be considered as the value of life itself.

Economists do not place a monetary value on these, but often take cognisance of them in a qualitative sense. Ecosystems' secondary values, also called the Total Economic Value (TEV) of ecosystems, comprise direct, indirect, option, existence, and bequest values. See box 1 for a discussion as to the different components of TEV.

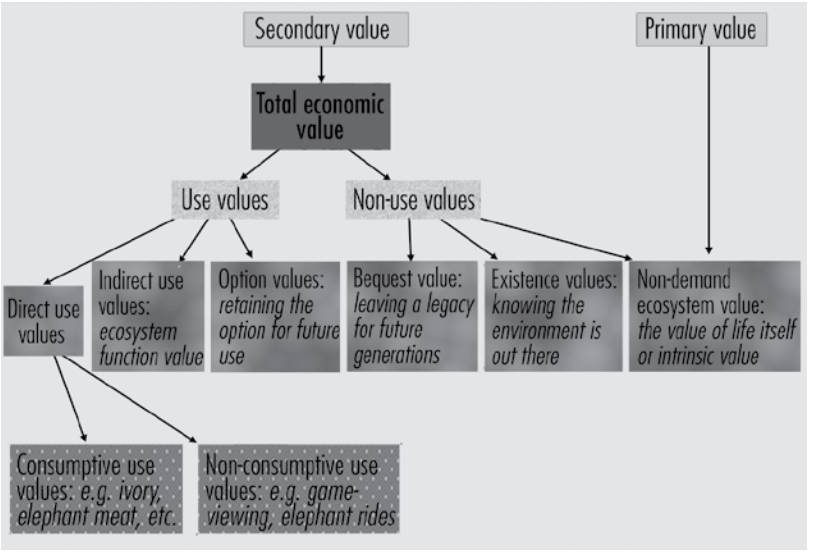

Figure 1: Values for the environment (adapted from Turner *et al.*, 1994)

In the next section we discuss this suite of values with specific reference to elephants.

FACTORS DETERMINING THE TOTAL ECONOMIC VALUE (TEV) OF ELEPHANTS

The TEV of elephants cannot be calculated by summing up all the animal's use and non-use values. There is conflict, even 'rivalry', among some of the categories. For example, the direct consumptive use of an elephant for its ivory excludes the possibility to enjoy any non-consumptive or non-use value from that individual animal. The direct consumptive use of the individual, however, does not – at least theoretically – exclude any non-consumptive or non-use value of the population as a whole. In some cases the direct consumptive use of a resource could have a negative impact on non-use values, depending on how people act and react to such direct use. This is due to the fact that non-use values are driven by perceptions and heavily influenced by specific contexts, which can change over time and in response to events. Neither are these values easily transferable from one setting to another.

The impact of elephants on their surroundings can also lead to a decline in the TEV of the return on the ecosystem in general. If not managed properly, elephants can lead to environmental degradation. Such degradation could

Box 1: Total Economic Value (TEV): A description

Direct use values are often exchange values since markets can exist for them. The estimation thereof is conceptually straightforward, but not necessarily easy. The fact that markets do (or can) exist does not imply that they are functioning well. Market imperfections such as legislations, trade-bans, and spatial and temporal differences between resources, can distort such a market and hence the market outcome. Direct use values can be sub-categorised as:

- consumptive use values (e.g., elephant meat, ivory, trophy hunting)
- non-consumptive use values (e.g., game-viewing, elephant rides, etc.).

Indirect use values correspond closely to the value of ecosystem functions (e.g., watershed protection, carbon sequestration, nutrient recycling). In the past these values tended to be use values but this is changing, with the advent of the carbon and water markets, and they are increasingly becoming exchange values. Biodiversity markets, however, are not well developed yet and the role an individual species, such as an elephant, plays within an ecosystem is also not isolated within this market. This is not to imply that this cannot change in future. Much discussion is under way to develop a biodiversity market of which both South Africa and all of southern Africa could be beneficiaries. Indirect use values are, however, not just positive. Individual species, such as an invasive alien plant, can have a negative impact on the social and economic value, and the ecological functioning of an ecosystem in general, and likewise the over-population of an endemic species such as an elephant can be globally negative.

Option value is an expression of an individual's preference not to make use of a resource today because he/she prefers to retain the option to use the resource in future and, therefore, is willing to pay for today's conservation to retain the option for any possible future use.

Bequest value is a measure of an individual's willingness-to-pay to ensure that an environmental resource is preserved for the benefit of his/her descendants. Bequest values are non-use values for the current generation, but a potential future use or non-use value for their descendants.

> Existence value measures the willingness to pay for the preservation of the environment that is not related to either current or optional use, thereby being the only true 'non-use' value. Existence values are based on the concept of the environment [or an individual species] being there. In some cases, bequest values are treated as part of existence values as it is often difficult to differentiate between the two on an empirical level.

lead to a loss in ecosystem function (indirect use value), which not only implies a loss in ecosystem productivity and resilience, but also the need for ecosystem restoration. The damage to field crops by elephants that escape from conservation areas and the ensuing challenges between humans and elephants are a direct cost to the affected human community. But this cost is not reflected in, for example, the value an international tourist derives from viewing elephants in the protected area where the damage-causing individual lives. This implies that space and context matter when considering economic valuation. Additionally, partial analyses may skew perception of the TEV. For example, should a study only focus on one aspect of the total economic value, say its non-consumptive use value, but not consider any other value – such as the loss of plausible consumptive use values or its nuisance value – this can lead to partial or even wrong conclusions. It is best to consider the suite of values as a package and, from an economic vantage point, optimise the suite of them rather than any one individual component. This implies the need for systems thinking and adaptive management, well informed by good data.

Lastly, two entrenched problems, inherent to all forms of economic valuation, are the issues of time and income difference. Studies have to make adequate provision for both the time preference of money – which usually depreciates over time – and the change in value of ecosystems goods and services – which often increases over time, should they become more scarce due to habitat loss. As for income differences, often communities adjacent to conservation areas are poor, while visitors to the park are affluent. These two constituencies tend to value and evaluate a resource such as elephants quite differently because of their different perspectives, and their different relationships with, or uses of, elephants. One has to consider and seek to either optimise the value of the system as a whole or to manage it sustainably and not just that of an individual value.

Most of the economic valuation studies of elephants done in the past focused on direct consumptive use value. Since 1989, when the African elephant

Box 2: Non-consumptive use values of elephants

Direct (non-consumptive) use: Within the tourism industry, elephants are important drawcards or attractions. The benefits of elephants within the ecosystem from a tourism perspective include direct income to households through employment, ownership, or equity in tourism-linked businesses, as well as foreign exchange earnings for the government, and government income through taxation of individual earnings, sales taxes and corporate taxes. It is, however, costly and a management-intensive exercise to host elephants. Elephant tourism options include either low numbers/high paying options (no self-drive; overnight lodges) or high numbers/low budget options (self-drive and camping or self-catering lodges). Elephant-related tourism expenditure is therefore a good indicator of people's willingness to pay for them.

Indirect use: Elephants are a keystone species in any biome where they occur and they play an important biological role in ecosystem functioning, ensuring the survival and continued evolution of many species. These values are generally not measured and can go two ways. One could value the indirect value of elephants either as an umbrella species, and therefore incorporating a range of other values in their value as well, or, individually by considering its role in the ecosystem. This could be positive, as an important habitat engineer, or negative, as a megaherbivore whose actions can lead to ecosystem degradation requiring restoration and intensive management. This is especially the case when population densities become too high.

Non-use values: There is an ongoing global concern for the continued existence of elephants. This concern is expressed mainly in the form of donations focusing on the protection of the elephant. In Kenya, for example, the elephant conservation industry is largely dependent on this form of money transfer for its continued survival. How sustainable and efficient it is, however, can and is being questioned (Norton-Griffith, 2007). Wildlife policies create the enabling environment for wildlife conservation, also for elephants, which, if designed appropriately, will be conducive to both conservation and the development of economic opportunities through markets. Market mechanisms can be developed to harness the non-use values of elephants in conjunction with their direct and indirect use values. (Based on Geach, 1997.)

was listed in Appendix I of the CITES list of endangered species (becoming effective in 1990), the direct consumptive use of elephants has been reduced dramatically and is effectively zero at present. Over time, however, it is likely to recover some of its importance thanks to the ongoing debate within CITES, especially between China, Japan and the other Far Eastern countries, on the one hand, and Europe and the United States on the other. The Far Eastern countries view the CITES trade ban as unnecessary and would like to see it annulled. By and large, the countries in southern Africa also support the removal of the trade ban, but for completely different reasons. They are concerned with the impact of their large and increasing populations of elephants on their habitat (see Chapter 3). Together, these countries form a lobby canvassing for the lifting of the ban, either in full or in part. Relaxation of the ban will lead to a new series of economic drivers influencing elephant conservation management. Such a change would also affect other, non-consumptive use factors, which determine the TEV of elephants, as is listed in box 2.

LITERATURE OVERVIEW

Southern Africa

Several studies estimating the economic value of elephants have been undertaken in Botswana, Namibia, and Zimbabwe. Nearly all of this work focused on direct use values associated with the elephant. Policy in all three countries is aimed at promoting generation of income and employment from wildlife, and research has thus been focused primarily on the value of elephant utilisation.

Prior to the Appendix I CITES listing of the African elephant, Child & Child (1986) and Child & White (1988) documented the financial values associated with elephant culling, which was being undertaken at that time in Zimbabwe to control the growing numbers of elephants in national parks. They showed that the culling programme, operated by a special unit within government, was profitable. Sales of ivory and dry, salted hides exceeded the costs of low-budget culling of matriarchal herds in the national parks. In addition, low-quality dried meat was provided cheaply to neighbouring communities in an attempt to engender local support for elephant conservation by offsetting the need for poaching for bush meat. The numbers culled varied between 800 and 1 500 per annum.

In 1989 the Botswana Department of Wildlife and National Parks undertook an analysis of the options for utilisation of its large and rapidly

growing elephant population. At that time, the only use of elephants was non-consumptive, as part of the general wildlife viewing experience. Hunting was banned and culling had not been introduced. The Appendix II listing for elephants at the time would have allowed reintroduction of elephant hunting and the introduction of a culling programme. Soon after that, initiatives among the CITES parties were made to have elephants listed in Appendix I. This was enacted in 1990, effectively closing all trade among CITES parties in consumptive products for the species. Botswana, which was against the listing, undertook a study to compare the economic values of the options for use of its elephant resource. Barnes (1990) estimated and documented the contribution that use of elephants for wildlife-viewing tourism, trophy-hunting tourism, hunting by citizens, and culling, could make to Botswana's national economy. This was followed by analyses for 1990 and 1992 of the effects that the international policy environment had on these values (Barnes, 1992; 1996a). The studies involved detailed financial and economic, budget/cost-benefit models of wildlife viewing activities in elephant areas, trophy hunting, and elephant culling, as developed by Barnes (1998). These models were based on empirical evidence from users, including data from the elephant use activities in Zimbabwe. The proportions of value attributable specifically to elephants were estimated as representing 41 per cent of wildlife viewing value, and 37 per cent of trophy hunting value. The models provided measures of the private profitability for the investor, as well as the net contribution of the activity to the national income. The net present value of various combinations of this income over 15 years, taking into account policy and plans for development of utilisation in the wildlife sector, were estimated, as summarised in table 1 (see Barnes, 1996a and 1998 for the details on the research methods employed).

As indicated in table 1, among the list of options for elephant use in Botswana in 1989, the combination with the highest value is Scenario 6, which contained all possible uses except hunting by citizens. To a large extent, elephant-viewing tourism, trophy hunting, and elephant culling were complementary spatially, allowing the highest values to be generated. The introduction of trophy hunting and culling of elephants was assumed to have a moderate effect on the values of elephant viewing through disturbance. In 1990, after the Appendix I listing, trophy hunting under quota was still permitted, and the option of culling was still a possibility, with some products marketed domestically and to non-CITES parties. Since 1990, culling could therefore add very little to the economic use value of Botswana's elephants, implying that the CITES listing effectively reduced the use value of elephants by some 47 per cent, as represented by the decline in value from P293 million in 1989 to P155 million in 1990 (table 1).

Scenario (option)	15 year present value @ 6%[a] (Pula million: 1989)[b]	
Viewing only with no consumptive uses	108.9	108.9
Viewing with trophy hunting only	153.2	153.2
Viewing with hunting by citizens only	130.7	–
Viewing with culling only	248.7	110.5
Viewing, trophy hunting, hunting by citizens and culling	282.3	–
Viewing, trophy hunting and culling	293.5	155.3

[a] Cumulative contribution to gross national income by year 15, after discounting at 6% per annum and after partial shadow pricing

[b] In 1989 Pula 1.00 was equal to ZAR 1.32 and US$ 0.51; Pula inflation factor from 1989 to 2007 is 3.50

Table 1: Present values of increases in Botswana's gross national income over 15 years, attributable to options for elephant management (1989 and 1990 analyses) (source: Barnes, 1996a; 1998)

Expenditure category[c]	15 year net present value @ 6%[a] (P million, 1992)[b] Utilisation option			
	Viewing only (no consumptive use)	Viewing with trophy hunting only	Viewing with hunting by citizens only	Viewing with culling only
Base case (costs rising from P16 to P242 per square km over 15 years)	123.5	181.5	122.6	181.2
Slow increase (costs rising from P16 to P510 per square km over 15 years)	84.0	142.0	83.2	141.8
Medium increase (costs rising from P16 to P510 per square km in first 10 years)	–1.5	56.5	–2.3	56.3
Fast increase (costs rising from P16 to P510 per square km in first 5 years)	–20.0	37.8	–20.9	37.6

[a] Value added over 15 years to national income, net of government expenditures, after discounting at 6% and after shadow pricing (April, 1992)

[b] In 1992 Pula 1.00 was equal to ZAR1.34 and US$ 0.47; Pula inflation factor from 1989 to 2007 is 3.02

[c] Different patterns of increase to a stable maximum for government expenditure on elephant management over the northern range (49 000 square kilometres)

Table 2: Effect of different scenarios for government expenditure on elephant management on economic net present values of different options for elephant utilisation in Botswana (1992 analysis) (source: Barnes, 1996a; 1998)

A second analysis carried out two years later, in 1992, showed similar results (Barnes, 1996a). Culling was not able to generate additional national income due to the restrictions on the ivory market. Elephant trophy hunting could, however, increase the value added by between 36 per cent and 58 per cent, depending on how much it disturbed elephant viewing activities. At the same time a cost-benefit analysis was conducted (Barnes, 1996a), comparing predicted national income streams generated from different possible use options with predicted government expenditure streams for elephant conservation. Future net income streams with management costs increasing to P242 per km^2 over 15 years generated positive returns in national income for all options. When costs were increased to P510 per km^2 (i.e. US$246.$km^{-2}$, after taking inflation and exchange rate fluctuations into account), as might occur with a surge in poaching, the inclusion of elephant trophy hunting was an important factor in ensuring a positive return for investment in elephant conservation. Table 2 shows the results of this analysis.

Table 3 shows the breakdown of value in terms of potential contribution to national income for all the different elephant products when all uses were included under conditions prevailing in 1989, 1990 and 1992. The salient point is that the culling values, which would have amounted collectively to 40 per cent of the total elephant use value in 1989, were reduced to negligible levels after that. The analysis of Barnes (1996a; 1998) provided evidence of the negative impact of the Appendix I listing on the economic viability of elephant conservation in Botswana. Combating elephant poaching for ivory was the prime motivation for the Appendix I listing, but this eliminated all culling values. It is noteworthy that values attributable to ivory (ivory sales and ivory carving in table 3) made up only 42 per cent of the total value of culling which was lost with the listing. Southern African countries have been trying to re-establish ivory markets within the CITES framework, but even if this is successful, it is unlikely that the 1989 markets for other elephant culling products, such as hides, could be revived.

Culling as a use option appears to have irreversibly lost the economic viability it had in 1989. In addition, culling as an activity has increasingly faced opposition from an animal rights perspective (see Chapter 9). Recent elephant utilisation policy in Botswana has allowed for a combination of elephant viewing and elephant trophy hunting only, with culling retained as a possible option for management purposes only. Since loss of culling value has resulted from attempts to conserve elephants, an argument could be made for compensation through the capture and transfer to Botswana of international non-use values for elephants.

Work on the economics of consumptive tourism (i.e. recreational hunting) in Namibia and Botswana (Novelli *et al.,* 2006) has shown that trophy hunting occupies a spatial niche that is complementary to and does not oppose or displace wildlife viewing tourism. The inclusion of elephants in trophy hunting quotas adds significant value to trophy hunting tourism. In addition to the elephant trophy fees, income from daily hunter fees is enhanced by the inclusion of a high-value elephant in the hunting bag. Using data from a northern Botswana trophy hunting enterprise model (Turpie *et al.,* 2006), and comparing values from trophy hunting in Botswana, where elephants are important (ULG Northumbrian, 2001), and Namibia, where less valuable plains game species are important (Novelli *et al.,* 2006), it was possible to impute a proportion of hunting income to elephants. Based on these calculations we estimate that some 44 per cent of the income from an elephant-inclusive hunting experience in northern Botswana is attributable to elephants.

	Year of analysis		
	1989	**1990**	**1992**
Total present value[b] (Pula million, 1989)[c]	293.5	155.3	133.0
Use category (%)			
Tourism – viewing	44.2	70.1	71.3
Tourism – trophy hunting	16.4	26.0	26.5
Culling – raw ivory	8.7	2.3	–
Culling – ivory carving	7.9	–	–
Culling – fresh or dried meat[d]	0.8	1.2	0.8
Culling – meat processing[e]	11.6	–	0.3
Culling – dry salted hides	6.6	–	0.6
Culling – hide tanning	3.7	–	0.2
Culling – live sale (calves)[f]	0.2	0.4	0.3
Total	**100.0**	**100.0**	**100.0**

[a] Management option 6, which included viewing, trophy hunting and culling for each year of analysis

[b] Present values for June 1989 and October 1990, and net present value for April 1992; all at 1989 prices

[c] In 1989 Pula 1.00 was equal to ZAR1.32, and US 0.51; Pula inflation factor from 1989 to 2007 is 3.50

[d] Carcass value after field recovery and field dressing

[e] Including (in 1989) use of meat as feed in crocodile breeding and rearing for production of skins and meat, and (in 1992) production of carcass meal

[f] Sale of calves between six months and one year old

Table 3: Proportional contributions of different products to the economic present values of elephant uses[a] in Botswana in the 1989, 1990, and 1992 analyses (sources: Barnes, 1996a; 1998)

No such comparative studies for South Africa have been conducted, but the live sale of elephants and the occasional hunting thereof on private land are permitted and the values known. Table 4 provides an overview of the average prices and number of trades over the past three years for various categories of animals. The trade in the number of live animals is restricted since conservation areas have commonly reached their carrying capacities. Trades are therefore restricted to private game farms. Similarly, the number of animals available for hunting is restricted by the fact that only animals from private game farms are eligible. The price per elephant, whether as a live sale or for a hunt, is very high, but this is attributable to the restricted nature of the market. It is therefore not possible to derive a total market value for all elephants in South Africa from these numbers. This is also the case in South Africa's neighbouring countries. The trophy values in the neighbouring countries are much lower, though. In Zimbabwe, for example, the trophy fee, set by government, for an elephant was US$10 000 for 2006/2007.

This figure is lower than that of 2000/2001, which was US$15 000, due to a decline in the quality of the animals. In a recent government auction for individual hunts in the Zambezi Valley safari areas, where a private individual can buy an elephant hunt as part of an associated bag of species, elephant hunts were sold for between US$25 000–30 000 per animal. In Botswana (2007), elephant trophy fees are US$18 000 and in Tanzania an elephant hunt (including all fees) is estimated to be US$23 000. In Mozambique the trophy fee for an elephant is only US$5 000, but this low value could be a reflection of the lack of a CITES trophy quota for that country (Cumming, pers. comm.).

The parties at the 12th Conference of Parties (CoP) to CITES in 2002 agreed to a one-off sale of 30 tons of ivory originating from the Kruger National Park. The prospective buyers had to register with the CITES Secretariat, fulfilling various requirements as laid down by the Conference. Only Japan and China indicated an interest in buying the ivory. To date (November 2007) only Japan has been verified as an acceptable trading partner. China will most probably be verified as a trading partner during the Standing Committee meeting scheduled for July 2008. CITES approved of the trade taking place at the CITES Standing Committee meeting in the Netherlands in June 2007. A further one-off sale has been approved by the 13th CoP of CITES (June 2007), which includes legally obtained ivory stock from South Africa, registered with the CITES Secretariat by 31 January 2007. Before the sale can take place, the ivory must be verified by the CITES Secretariat to be eligible for sale within the CITES framework and agreement.

Category	Live sales Price per animal (ZAR)*	Number	Category**	Hunts Price per animal (ZAR)*	Number
Trained animals	575 000–1 100 000	_	15–20 kg	290 000	10
Juveniles	50 000–500 000	–	20–25 kg	325 000	7
Cows plus family	15 000	150	30–35 kg	430 000	2–3
Bulls: approx. 20 kg**	70 000	30	35+ kg	500 000	2
Bulls: approx. 30 kg**	100 000	20	–	–	–

* Numbers quoted in rand, but most trading takes place in US$ and an exchange rate of ZAR7.2 per US$ has been used
** Weight of tusks

Table 4: Average prices and number of elephants traded in South Africa per year over the period 2005–2007* (Grobler, pers. comm.)

Can people living in areas adjacent to and in elephant-containing ecosystems benefit in any way from the presence of the elephants? One mechanism through which elephants can benefit local communities is through community-based natural resource management (CBNRM) programmes. CBNRM programmes that aim to partially devolve property rights over wildlife to communities on communal land have been under development in nearly all southern African countries since the 1980s, and are well developed in Namibia, Zimbabwe, and Botswana. Wildlife use, involving elephants for both wildlife viewing and trophy hunting, is commonly associated with these programmes. CBNRM in Namibia (Libanda & Blignaut, 2007), and in Botswana, involve both non-consumptive and consumptive tourism, but in Zimbabwe's CAMPFIRE programme, over 80 per cent of income derives from trophy hunting, which in the 1990s was dominated by elephant values (Bond, 1994; 1999). This figure seems to have risen above 90 per cent in recent years (Muchapondwa, 2003).

Elephants are therefore quite important as generators of income both nationally and for local communities in Botswana, Namibia, and Zimbabwe. However, they also generate costs in the form of damage to crops and infrastructure wherever they occur outside of fenced conservation areas. Sutton (2001) and Sutton *et al.* (2004) conducted a detailed household survey to measure the costs and benefits of living with elephants in the Caprivi Region of Namibia. Sutton determined that in the agro-pastoral system, which predominates in this region, elephants generate fewer damage costs than other wildlife, and that livestock actually causes more crop damage than all wildlife put together. Nevertheless, elephants still manage to reduce crop yields significantly. Jones & Barnes (2007) used crop damage data in crop enterprise

models to show that average crop losses due to elephants reduced net profits for small-scale crop growers by some 30 per cent. Crop damage varies spatially, and in areas where it is the highest (some two or three times the average), crop profits can be eliminated altogether. Barnes (2006) used a similar crop enterprise approach to estimate the value of crop losses due to elephants in the Okavango Delta area of Botswana. Here, damage levels were generally higher, and average small-scale, rain-fed crop production profits were reduced by some 75 per cent, and even entirely eliminated in some cases.

	Elephant crop damage cost level		
	Basic damage cost	**2 x damage cost**	**3 x damage cost**
Trust profit	604 200	333 600	–155 900
Community net benefit	1 199 400	928 800	439 300
Gross output	2 578 300	2 578 300	2 578 300
Gross national income (GNI)	2 002 900	1 777 600	1 349 800
Net national income (NNI)	1 894 400	1 669 100	1 241 400

[a] In 2006 Pula 1.00 was equal to ZAR 1.14, and US$ 0.16; Pula inflation factor from 2006 to 2007 is 1.06

Table 5: Impact of elephant crop damage costs on the measures of private and economic viability for a model CBNRM community trust investment in the Okavango Delta, Botswana (Pula per annum, 2006)[a] (source: Barnes, 2006)

Of importance here is the degree to which elephant damage costs incurred by communities can be offset by the benefits they derive from use of elephants through CBNRM. Models of community investments in CBNRM, developed by Barnes *et al.* (2001; 2002) were used to compare the wildlife crop damage costs with the utilisation benefits incurred by communities in both of the Caprivi and Okavango delta study sites. Table 5 and figure 2 (derived from Barnes, 2006) show the results for a typical CBNRM investment in the Okavango delta. The impacts of various crop damage levels (based on average figures) on the profits made by the community trust, the community members as a group, and the contribution made by the investment to the gross and net national income, were measured. Generally, benefits outweighed costs for all measures. In the case of the community trust, losses were only incurred when damage costs of three times the average levels were sustained over time. Jones & Barnes' (2007) results for the Caprivi Strip, Namibia, also established that CBNRM benefits generally outweighed crop damage costs. Various policy options are available to address elephant and wildlife damage costs. These studies suggested that

human-elephant conflicts could be internalised with CBNRM programmes. For a further discussion on the human–elephant link within a CBNRM context, please see Chapter 4.

While it appears that in southern Africa rural people at the community level can derive positive net benefits from wildlife, do they actually derive direct financial gains from it? Libanda & Blignaut (2007) found that in Namibia households do generally benefit significantly from CBNRM and that sufficient institutional mechanisms are in place to ensure broad-based support for the programme, as indicated by the rapid growth of the CBNRM programme from its inception in 1996, to the end of 2006, when it included 50 CBNRM areas and covered an area of 118 705 km^2. The area under CBNRM management comprises 15 per cent of the land surface of Namibia and is adding to the 16.5 per cent of the land surface area that is already formally protected. CBNRM areas already host 37 per cent of Namibia's rural population and a further 31 conservancies are in various stages of development, clearly indicating the widespread interest in, and support for, the programme.

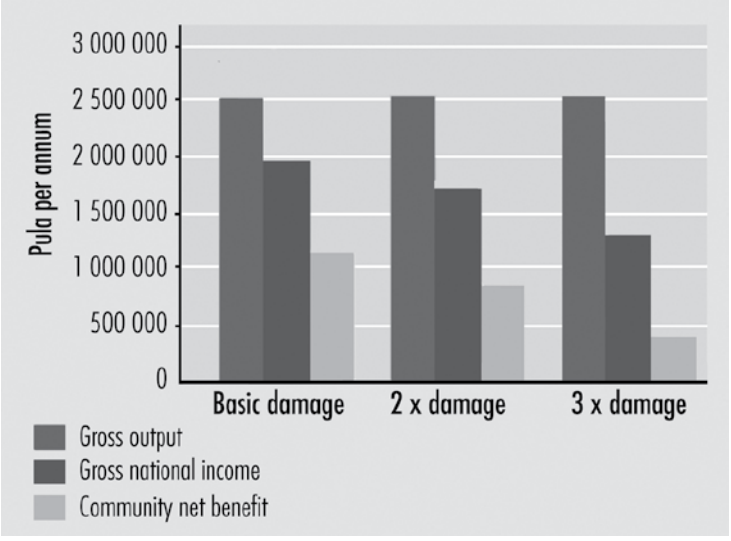

Figure 2: Impact of crop damage costs due to elephants on the economic gross output, the contribution to the gross national income, and the private community net benefits for a model CBNRM community trust investment in the Okavango Delta, Botswana (Pula per annum: 2006) (Barnes, 2006)

In contrast, this success of CBNRM is not unequivocally shared in Zimbabwe. Muchapondwa (2003) and Muchapondwa *et al.* (2003) conducted contingent

valuation studies in Mudzi District, a CAMPFIRE district since 1992, where households' willingness to pay for the preservation of elephant was measured. Some 570 households, randomly selected from within two similar wards in Mudzi District were surveyed, and, along with the willingness to pay bids, variables such as household size and income, sex, age, and education of household head, distance from an elephant reserve, size of intruding elephant herds, existence of mitigation, support for government conservation, participation in agriculture, and labour spent on mitigation were tested. The studies found that 34 per cent of households were willing to pay for elephant preservation, with a median willingness to pay (WTP) of Z$300 or US$5.45. This was 3.87 per cent of median annual income. However, 62 per cent of households had a negative willingness to pay for elephant – they were willing to pay to have elephants removed from their area, with a median WTP of Z$98 or US$1.78. This was 1.27 per cent of median annual income.

The results indicated that the community as a whole had a net positive willingness to pay for elephant preservation, but that the majority of community members did not support elephant preservation. This suggested that any net benefits that the community might have derived from CBNRM must not have been reaching many households. Muchapondwa *et al.* (2003) recommended external transfers to households in Mudzi to increase incentives for elephant conservation. The willingness to pay values estimated by Muchapondwa *et al.* (2003) can be said to represent non-use values, namely, any or all of option, bequest, or existence values. In the CBNRM context, they are likely to be made up largely of option values. Apart from these findings on local non-use values, no other studies appear to have been carried out.

Other examples

While we have emphasised the studies estimating the economic value of elephants in southern Africa thus far, a large number of other, non-regional, studies have been conducted as well, a selection of which is summarised in table 6. It must be noted that values derived in these studies are not always comparable, either between themselves or with the studies listed above, since different methods and measures are used.

Using an open-ended stated preference technique, Vredin (1997) estimated the median Swedish household's willingness-to-pay (WTP) for the preservation of African elephants, which is an attempt to capture the non-use values of elephants. With a resulting median value of SEK100 (= US$14.92) per household for the year 1996, it was estimated that the aggregated WTP of the Swedish

population for the preservation of the African fauna and flora (using the African elephant as indicator) is SEK383 million (=US$53.7 million). The main motives stated were: existence value (30 per cent of valid observations), care for future generations (28 per cent – bequest values) and own experiences (18 per cent – option values). This WTP is sensitive to changing income, as follows: a 1 per cent increase in income would lead to a 0.3 per cent increase in WTP (Hökby & Soderqvist, 2003). When taking this income elasticity into account as well as an average growth rate of 2.8 per cent, and changes in population since 1996, but with all other things being equal, aggregated WTP in 2006 has increased to SEK420 million (US$57 million). At average 2006 exchange rates, this amounts to US$14.73 per household per year. Currently, there are 470 000–690 000 African elephants in the wild (WWF, undated). Assuming 500 000 elephants and extrapolating to all 150 million European and US households (see Bulte *et al.,* 2006), this amounts to an indicative total WTP of US$2.2 billion per annum, or US$4 420 per elephant per annum. These numbers are, however, only indicative of the fact that the WTP for elephant conservation is potentially significant. They cannot be used in absolute terms since they are based on too many assumptions.

The estimated total gross tourism viewing value of elephants, in particular, was estimated at between US$25 and 30 million in Kenya in 1989 (Brown & Henry, 1989). This value was based on the travel costs of European and North American visitors and their stated purpose of travel. With an estimated 16 000 elephants in Kenya in 1989 (Ivory Trade Review Group, as quoted on the website http://www.american.edu/ted/elephant.htm), and using a low value of US$25 million per annum, that amounts to a mean WTP of US$1 562 per elephant in Kenya. Assuming declining travel costs and rising income over time this figure can be used as indicative for current values, but with low levels of confidence. Assuming that only three-quarters of Africa's elephants (375 000) are accessible to tourism this provides an indicative value of US$585 million or US$3.91 per European and US household per year. This is probably a low estimate, as up to 90 per cent of African elephants occur in southern and eastern Africa (Blanc *et al.,* 2007), both of which regions are readily accessible to international tourism. With low levels of confidence in these numbers – due to the fact that the studies on which they are based are dated and were carried out by various researchers in a variety of places using different methods – all of these discrepancies make comparisons difficult.

What has been valued	Valuation technique	Source	Values	Remarks
WTP of Swedes to preserve the population of African elephant	• Open Ended Contingent Valuation Method	Vredin (1997)	1996; US$53.7 million for all Swedes	1500 Swedish residents in age group 19–75
	• Linear aggregation	Hokby & Soderqvist (2003)	Median: SEK 100 per household	
The cost of preventing a decline of elephants from severe commercial poaching for their ivory	Defensive Expenditure Method (Cost of protection)	Leader-Williams (1994)	1981: US$215 per km^2 (adjusted to 1994 values: $340 per km^2)	The relationship between spending and success in protecting elephants was significant but only explained 32% of the variance
Tourism value of elephants in Kenya	Travel costs	Brown & Henry (1989)	1989: US$25–30 million pa, $1 562 per elephant	Estimating consumer surplus from European and North American visitors
Conservation of 650 elephants in Amboseli NP	Marginal cost of PES scheme to conserve elephants	Van Kooten & Bulte (2000) Bulte *et al*. (2006)	US$10 per acre per year (US$2 470 per km^2) or US$175 per elephant per year; equal to an estimated minimum of US$0.60 per European and US household per year for all African elephants	Current estimates of the African elephant population amount to some 500 000 head. Assuming a minimum benchmark cost of $175 per elephant per year, the total benefits of elephant conservation should amount to $87.5 x 106 per year. Dividing by the number of households (150 x 106) this amounts to $0.60 per household per year (Bulte *et al*. 2006)
Value of ivory exports from Africa	Market price	Ivory Trade Review Group(as quoted http://www.american.edu/ted/elephant.htm) Cobb (1989)	1979: US$36.89 million 1987: US$19.18 million 1979–1987: >US$500 million	Estimated 1.3 million elephants killed for their tusks during 1970s and 1980s
Ivory value	Market price	Vredin (1995)	1987: US$2 734 per elephant	

What has been valued	Valuation technique	Source	Values	Remarks
Trophy value	Market price	Vredin (1995)	1989: US$2 366 per elephant	
Relocation of elephants from KNP to Shamwari Game Reserve	Market price	Wilderness Conservancy (http://www.wilderness conservancy.org/projects/ ongoing.html)	US$2 850 per elephant	

Table 6: Valuation studies on African elephants (excluding studies from southern Africa)

Another way to value elephants is to estimate the minimum costs to sustain an elephant or elephant population. This would normally provide a measure of minimum value. The minimum cost to conserve elephants in Luangwa Valley, Zambia, during a time of intensive poaching was estimated at around US$215 per km^2 in 1981 values, and when adjusted for inflation amounts to US$340 per km^2 in 1994 terms (Leader-Williams, 1994). Using the same average 4.5 per cent annual increase in costs from 1981 to 2006 as used by Leader-Williams (1994), current cost levels are estimated at around US$600 per km^2. Assuming desired density of two elephants.km^{-2} in savanna habitat – which is high – this amounts to a cost for elephant conservation of US$300 per elephant or US$150 million per annum. In relation to the number of households in Europe and the US this amounts to US$1 burden per household per annum. These results should be interpreted with caution as only 32 per cent of conservation success could be explained by spending levels in the original study (Leader-Williams, 1994, 31). This implies that more spending, i.e. a bigger budget, is insufficient to assure elephant conservation, but institutional factors and management practices play a significant role as well.

When elephants cross protected area boundaries into adjacent human-inhabited areas, the costs of protection increase. In a study on the minimum cost of implementing a payments for ecosystem goods and services (PES) scheme in the Amboseli National Park of Kenya it was estimated that Maasai farmers needed compensation equal to US$10 per acre per year for roaming elephant populations in their croplands (Bulte *et al.*, 2006). For the 650 elephants of the Amboseli Park, this amounts to a compensation cost of US$175 per elephant. Assuming that this study is representative of all African farmers confronted with elephants (a very strict assumption) and that all of the 500 000 elephants in Africa can migrate across protected area boundaries (a clear worst-case situation), this amounts to a maximum of US$87.5 million per annum in compensation payments. For comparison, this amounts to a theoretical burden of US$0.60 per household per annum for all European and US households, which implies that if all these households pay US$0.60 per year, sufficient money could be collected to offset the damage caused by the elephants to crops.

Care should be taken interpreting this number since it is based only on one study consisting of 650 animals, but, indeed, it does indicate that the value from tourism (estimated above as US$3.91 per European and US household per year) is significantly more than the damage cost caused by elephants. This appears to create a unique opportunity for the implementation of a PES system.

The cost of translocation is also an indication of the socio-political WTP for the conservation of elephants. In South Africa, costs of up to US$2 850

per elephant were reported for translocation within the country (Wilderness Conservancy, no date); see also Chapter 5 for a detailed discussion. The total WTP for elephant relocation has not yet been estimated.

Vredin (1995) estimated the ivory value per elephant at US$2 734 (1987 prices). According to a recent report by CWI (2007), ivory prices for unworked pieces of ivory range from US$121 to US$900 (average US$390) per kilogram. Another recent release by CITES stated that the black market value of African ivory is approaching a high of US$700 per kilogram (CITES, 2007). It is well known that ivory per elephant is declining rapidly, and currently estimated at between 7 kg and 12 kg of ivory per African elephant (Van Kooten, 1995; Hunter *et al.,* 2004). Multiplying this by the price range of US$121–900 provides an estimate of US$850–US$6 300 per elephant. At an average price of US$390.kg^{-1} the current average value is estimated at around US$2 725 per elephant. Given the illegal nature of the ivory trade, it is very difficult to estimate the number of elephants involved. Nevertheless, Hunter *et al.* (2004) used one set of data, and careful extrapolation methods, to estimate that the ivory from between 4 862 and 12 249 African elephants is required annually to supply the unregulated markets in Africa. Although it is only a best guess at this stage, this would imply a market of between US$4.1 and US$77.2 million annually. This represents a theoretical burden of between US$0.03 and US$0.51 per European and US household. The trophy value of elephants was closely matched to the value of ivory and estimated at US$2 366 at 1989 prices (Verdin, 1995).

ASSESSING ELEPHANTS' CONTRIBUTION TO THE ECONOMIC VALUE OF ELEPHANT-CONTAINING ECOSYSTEMS

The suite of economic values of elephants are summarised in table 7. Though these values are by no means definitive and are often based on outdated data and various assumptions, using different valuation techniques, a clear picture appears. The consumptive benefits (e.g. ivory, trophy hunting) of the African elephant are much less than its non-consumptive (e.g. tourism) and non-use (e.g. existence, option, and bequest) values. The stated WTP for the preservation of the African elephant for just the Swedish population (US$57 million) is only 28 per cent less than the high-end estimate for the value of the total ivory market (US$77 million). If we hypothesise that this same WTP is shared by all European and American households – which are more or less, relatively speaking, on the same welfare level when compared to the average African household – then the high-end value of the ivory market is only 3.5 per cent of the potential Euro–North American WTP for the preservation of the African elephant. This analysis

also points out that a compensation programme for both the direct damage costs of elephants to farmers and lost ivory income (a combined cost of US$165 million per annum) is 7.5 per cent of the estimated WTP for preservation by European and American households. Such a voluntary conservation aid programme would also save an additional US$150 million in protection costs. Obviously, there is little confidence in the absolute level of these numbers, or how much of this market could actually be realised, or what South Africa's portion of it could be, but they are sufficiently high to indicate that options for alternative scenarios exist when considering the potential scope for the creation of a market for the preservation of the African elephant.

Type value	Comparative value per US & EU household (US$)[1]	Value per elephant (US$)[2]	Total estimated value per annum (US$)
Mainly existence, bequest and experience value	14.73	4 420	2.2 billion
Non-consumptive tourism value	3.91	1 562	585 million
Protection costs against poaching	1	300	159 million
Compensation costs to surrounding land owners	0.60	175	87.5 million
Offsetting consumptive value of ivory	0.03–0.51	2 730	4.1–77.2 million
Consumptive value of trophy hunting	n/a	2 360	n/a
Translocation costs	n/a	2 850	n/a
Trade in live elephants[3]	n/a	2 000–70 000	n/a
Hunting values[3]	n/a	40 000–70 000	n/a

[1] For comparison all values are expressed in terms of 150 million European and US households willing to pay, see Bulte *et al*., 2006 for a similar approach

[2] Values adjusted to reflect 2006/07 estimates

[3] These values, from the South African studies, are inflated due to the restricted market

n/a Not available

Table 7: Summary of main economic values of African elephants

The formally measured and accounted-for direct consumptive use values of the African elephant are low, as is to be expected given the heavy impact of the CITES ban. As noted by Barnes and his colleagues, the realised TEV, excluding non-use values, of elephants has declined due to the CITES listing of elephants, probably by as much as 47 per cent. Although the non-consumptive, indirect, and non-use values of elephants are high (Vredin, 1997; table 7), the CITES listing has reduced the real cash flow to both nations and communities.

This is because there are currently few mechanisms to retrieve or capture the non-use values. What is required is measures to protect, compensate, translocate, and even consume elephants, in a sustainable fashion, and, concurrently, for local communities, the nation, and the elephants to derive direct, measurable, and tangible benefits from all such activities. Within the development of such a 'conservation, preservation and sustained use' market, and of institutions to support it, Far Eastern countries can likely play an important role, especially related to the direct 'consumption' of elephant tusks. Additionally, if communities do not directly benefit from the presence of elephants, whether through consumptive or non-consumptive use or a combination thereof, indications are that they will not support elephant conservation in future (see the example from Zimbabwe). If, however, they are integrated, and made part of the 'solution', then indications are that they would readily support conservation (see example from Namibia). The experiences of these countries offer South Africa excellent learning references.

What is also apparent is that an inclusive conservation package that allows for all the possible economic benefits to be realised would be easily offset by the sum of economic benefits that could be gained. The challenge remains to create an efficient institution that would be able to capture these gains – that is, the economic rent – and distribute this to the benefit of both landowners and elephants. Evidence from all the studies cited previously suggests that international willingness to pay for elephant conservation in African countries exists, which implies that South Africa has a range of options to choose from. Barnes *et al.* (2002) supports this view and maintains that much of the hitherto substantial international NGO and donor support for CBNRM is a form of non-use values. Additionally, contingent valuation studies among wildlife viewing tourists in Botswana and Namibia (Barnes, 1996b; Barnes *et al.,* 1999) revealed a significant willingness to pay for wildlife conservation. The tourists surveyed generally had trip consumer surpluses and were willing to pay more for their trips than they had paid, a view supported by South African studies as well (Turpie, 2003; Turpie & Joubert, 2001; Geach, 1997). This implies that the value tourists received from viewing the wildlife was more than the economic cost of hosting them. The surplus, which constitutes economic rent, is attributable to the wildlife (elephants) and, if retained (captured) these rents could be used to advance conservation. At least a portion of the tourists' willingness to pay for conservation could thus come out of these surpluses, and may be defined as direct non-consumptive use value. It is important to note, however, that the estimated non-use values, as summarised in both tables 5 and 7, are mostly only hypothetical values. Until institutional mechanisms are created through

which such hypothetical values can flow and be materialised to the advantage of both people (through CBNRM or otherwise) and elephants, and to the nation as a whole, they remain hypothetical.

Economists (e.g. Bulte *et al.,* 2006; Van Kooten & Bulte, 2000; Kahn, 1998; Barbier *et al.,* 1990) seem to share the view that the use of markets through a well-designed institutional arrangement is a much better way of managing a precious resource over the long term, than an outright ban. This is so because markets offer more management options and flexibility than command and control mechanisms. Barbier *et al*. summarise this thought as follows (Barbier *et al.,* 1990, 147):

> The future of the African elephant is dependent upon the taking of immediate action. The ivory trade ban must be considered an interim measure, not a solution. Sustainable populations of the African elephant, as with so many other endangered species, will depend upon the development of reforms which constructively utilize the trade, rather than attempts to combat it. Institutional reforms to this end must be addressed now.

The development of market options has to be considered also from the perspective that official development aid, especially predominant in East Africa, is not sustainable in the long run and cannot sustain or improve conservation (Van Kooten & Bulte, 2000; Norton-Griffith, 2007). A further stimulus for the development of markets is provided by the emergence of the Far Eastern markets as significant role-players within the global ivory trade. This implies that the political-economic gridlock concerning the ban on trade in ivory cannot be maintained indefinitely. Leakage – both the legal and illegal trade in ivory – is likely to occur since sanctions and bans are imperfect measures in the long run. It is much more prudent to manage proactively and to introduce the use of markets and incentives measures in a controlled environment rather than to be confronted with the effects of leakage. Since the economic system is a self-organising system (Krugman, 1996) that requires adaptive management, markets and incentive measures are much more efficient and effective in achieving such desired behavioural change, if constituted and institutionalised appropriately, than are traditional command-and-control measures. In this context the use of market-based and command-and-control measures can occur in conjunction with each other for a period of transition, allowing markets to operate within a controlled environment and, progressively, to mature.

The time for such institutional change is ripe now. Almost two decades since the African elephant's listing as an endangered species, its numbers

have increased by 50 per cent. Concurrently, much experience has been gained in incorporating CBNRM into the conservation framework and thereby distributing conservation benefits broadly, and this could include the sustainable direct use or extraction of elephants (Damm, 2002). Such direct use will reduce at least the growth in the number of elephants, but, as has been observed in Botswana, the numbers are likely to be relatively small. It should be noted that the sustainable use of elephants is, at least theoretically, not in conflict with the non-use values, but could instead be an important complement.

In parallel to the development of CBNRM and other institutional arrangements over the past two decades, much has been learnt since the late 1980s and early 1990s on how to establish and operate markets for ecosystem goods and services (Pagiola & Platais, 2007). Such a market would allow for the transfer of money, especially from Europe and the USA, to capture some of the non-use values of elephants. In so doing, the economic value of elephants can be optimised by capturing all the values (direct consumptive, direct non-consumptive, indirect, and non-use values) and, additionally, by releasing finances to both conserve the elephants, and increase their range to include human-occupied areas (Van Aarde & Jackson, 2007; Van Aarde *et al.,* 2006). This option would inject new streams of income into rural communities, all across South Africa, especially to those living in areas adjacent to elephant-containing ecosystems, many of whom have a formal land claim on currently protected land. This offers an opportunity to link the formal (first) economy of South Africa with the informal (second) one, and to inject finances into the second economy by embracing the two as partners and fellow custodians of the natural environment and national heritage. This option is becoming increasingly viable due to current and probable future socio-demographic changes, as South Africa undergoes a rapid increase in urbanisation and possibly even de-population of the rural areas.

CONCLUSION

Some values of the African elephant are clearly expressed in the market, such as tourist expenditures on elephant viewing, or the direct costs of trophy hunting; the direct use benefits from elephants include ivory, although banned, and other animal products. However, non-use values are generally not captured as income or observed, and are hence difficult, but not impossible, to determine. The willingness to pay to conserve elephants for future generations on the part of many people who may never even see an elephant in their lifetimes, is

Box 3: Key research questions

What is the economic value of elephants in South Africa?

- What is the most appropriate, desirable, and feasible institutional arrangement and market mechanism to realise the suite of economic values of elephants?
- How could elephant markets, realising the direct, indirect, and non-use values of elephants, benefit local populations adjacent to elephant containing ecosystems?
- What are the likely impact of the emerging ivory market in the Far Eastern countries on South Africa and the impact thereof on the elephant management options for South Africa?
- How can markets be constructed to assist in reducing the risk and uncertainty in managing elephants and elephant containing ecosystems to the advantage of both elephants and people?

generally only partially captured through donations and thus largely remains unexpressed. An interpretation of economic value thus goes beyond exchange values as measured through market-based transactions.

Although there are no studies on the TEV of elephants in South Africa, there is a rich knowledge base thanks to work done in Botswana, Zimbabwe, and Namibia. Based on these studies, there is evidence of (1) an increase in the proportional contribution of non-consumptive values to the TEV of elephants, but (2) a decline in the overall economic value derived from elephants after the CITES ban on trading in elephant products. There is mixed evidence of the extent of elephant damage to local communities' crops and infrastructure from studies done in Botswana and Namibia. In some cases it was less than the damage by livestock, but in other cases substantial losses were incurred. In Kenya it was estimated that benchmark damage costs to the Maasai amounted to US$2 470 km^{-2}.year. In South Africa it is more than likely that costs are substantially lower due to our formal elephant management system in fenced-in conservation areas. A list of pertinent research questions with specific reference to South Africa is listed in box 3.

The success of institutions to compensate local communities, on the one hand, for their loss in income of elephant and elephant products and, on the other, for damage costs, is also mixed. There is evidence of some success in

distributing the economic value of conservation through CBNRM schemes in Namibia, but much less in Zimbabwe. The proper function of institutional success is a prerequisite for the effective internalisation of damages.

Based on evidence of international willingness to pay for the conservation of elephants, and the recent development concerning markets for ecosystem goods and services, ways have to be found to internalise this expressed willingness to pay to advance elephant conservation. Traditional policy options are limited in their scope as regards achieving this objective, but significant evidence exists that there is potentially sufficient international support to develop market-based alternatives. These high expressed non-use values for elephants are based on three factors, namely the fact that elephants exist – in other words that they *have to be preserved* for future generations; the ecological role they play within ecosystems; and the fact that people want to have the option to enjoy benefits from them in future. The preliminary meta-analysis presented in this chapter suggests that the non-use values from Europe and the US are three to four times higher than tourism values, 25 times higher then the benchmark compensation payments required to land owners, and almost 30 times higher than a high-end estimation of the total ivory market. This is a trend supported by De Boer *et al*. (2007). There is therefore abundant scope for the creation of markets and institutional strengthening.

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REFERENCES

Barbier, E., J. Burgess, T. Swanson & D. Pearce 1990. *Elephants, economics and ivory*. Earthscan, London.

Barnes, J., J. Cannon & K. Morrison 2001. Economic returns to selected land uses in Ngamiland, Botswana. Unpublished report, Conservation International, Washington, DC, USA.

Barnes, J.I. 1990. Economics of different options of elephant management. *Proceedings, Kalahari Conservation Society Symposium: The Future of Botswana's Elephants*. Gaborone, Botswana, 10 November 1990, 60–66.

Barnes, J.I. 1992. The economic use value of elephant in Botswana in 1992. Unpublished paper, Department of Wildlife and National Parks, Gaborone, Botswana.

Barnes, J.I. 1996a. Changes in the economic use value of elephant in Botswana: the effect of international trade prohibition. *Ecological Economics* 18, 215–230.

Barnes, J.I. 1996b. Economic characteristics of the demand for wildlife viewing tourism in Botswana. *Development Southern Africa* 13(3), 377–397.

Barnes, J.I. 1998. Wildlife economics: a study of direct use values in Botswana's wildlife sector. Ph.D. thesis, University College, University of London, London.

Barnes, J.I. 2006. Economic analysis of human-elephant conflict in the Okavango delta Ramsar site, Botswana. Unpublished report, Natural Resources and People, Gaborone, Botswana.

Barnes, J.I., C. Schier & G. van Rooy 1999. Tourists' willingness to pay for wildlife viewing and wildlife conservation in Namibia. *South African Journal of Wildlife Research* 29(4), 101–111.

Barnes, J.I., J. MacGregor & L.C. Weaver. 2002. Economic efficiency and incentives for change within Namibia's community wildlife use initiatives. *World Development* 30(4), 667–681.

Blanc, J.J., R.F.W. Barnes, G.C. Craig, H.T. Dublin, C.R. Thouless, I. Douglas-Hamilton & J.A. Hart 2007. *African elephant status report: an update from the African Elephant Database*. IUCN, Gland.

Blignaut, J., J. Aronson, P. Woodworth, S. Archer, N. Desai & A. Clewell 2007. Restoring natural capital: a reflection on ethics. In: J. Aronson, S. Milton & J. Blignaut (eds) *Restoring natural capital: the science, business, and practice*. Island Press, Washington, DC, 9–16.

Bond, I. 1994. The importance of sport-hunted African elephants to CAMPFIRE in Zimbabwe. *Traffic Bulletin* (WWF/IUCN) 14(3), 117–119.

Bond, I. 1999. CAMPFIRE as a vehicle for sustainable rural development in the semi-arid communal lands of Zimbabwe: incentives for institutional change. Ph.D. thesis, Department of Agricultural Economics and Extension, University of Zimbabwe.

Brown, Jr., G. & W. Henry 1989. *The economic value of elephants*. Discussion Paper 89-12. London Environmental Economics Centre.

Bulte, E.H., R.B. Boone, R. Stringer & P.K. Thornton. 2006. *Wildlife conservation in Amboseli, Kenya. Paying for nonuse values*. Roles of Agriculture Project Environment Series, December. Agriculture and Development Economics Division, Food and Agriculture Organisation of the United Nations.

Care for the Wild International (CWI) 2007. US exposed as leading ivory market. *Care for the Wild International,* 5 June.

Child, G. & B. Child 1986. *Some costs and benefits of controlling elephant populations in Zimbabwe*. Proc. 7th and 8th Sessions of the AFC Working Party on Wildlife Management and National Parks, Arusha, Tanzania, 1983, and Bamako, Mali, 1986, FAO, Rome, Italy, 74–77.

Child, G. & J. White 1988. The marketing of elephants and field-dressed elephant products in Zimbabwe. *Pachyderm* 10, 6–11.

CITES 2007. Cites decision promotes illegal ivory trade. *Official Bulletin of Species Management Specialists Inc*. 14th conference of the parties to CITES, The Hague, Netherlands, 3–15 June 2007.

Cobb, S. (ed.) 1989. The ivory trade and the future of the African elephant. Volume 1: Summary and conclusions; Volume 2: Technical reports. Unpublished report of the Ivory Trade Review Group, Oxford, UK.

Damm, G. 2002. *The conservation game: saving Africa's biodiversity*. Rivonia: Safari Club International African chapter.

De Boer, F., J.D. Stigter & C.P. Ntumi 2007. Optimising investments from elephant tourist revenues in the Maputo Elephant Reserve, Mozambique. *Journal for Nature Conservation.* In press.

Ekins, P., S. Simon, L. Deutsch, C. Folke & R.S. de Groot 2003. A framework for the practical application of the concepts of critical natural capital and strong sustainability. *Ecological Economics*, 44, 165–185.

Farley, J., & E.J.B Gaddis 2007. Restoring natural capital: an ecological economics assessment. In: J.S. Aronson, S. Milton & J. Blignaut (eds) *Restoring natural capital: the science, business, and practice*. Island Press, Washington, DC, 17–27.

Geach, B. 1997. The Addo Elephant National Park as a model of sustainable land use through ecotourism. MSc thesis, University of Port Elizabeth.

Hökby, S. & T. Soderqvist 2003. Elasticities of demand and willingness to pay for environmental services in Sweden. *Environmental and Resource Economics* 26, 361–383.

Hunter, N., E. Martin & T. Milliken 2004. Determining the number of elephants required to supply current unregulated ivory markets in Africa and Asia. *Pachyderm* 26, January–June, 116–128.

Jones, B. & J. Barnes 2007. WWF human wildlife conflict study: Namibian case study. Unpublished report, Macroeconomics Programme and Global Species Programme, WWF, Gland, Switzerland.

Kahn, J. 1998. *The economic approach to environmental and natural resources*. Dryden.

Krugman, P. 1996. *The self-organising economy*. Blackwell, Oxford.

Leader-Williams, N. 1994. The costs of conserving elephants. *Pachyderm* 18, 30–34.

Libanda, B. & J. Blignaut 2007. Tourism's local benefits for Namibia's community based natural resource management areas. *International Journal for Ecological Economics and Statistics.* In press.

McCauley, D.J. 2006. Selling out on nature. *Nature* 443(7), 27–28.

Muchapondwa. E. 2003. The economics of community-based wildlife conservation in Zimbabwe. Ph.D. thesis, Department of Economics, Göteborg University, Sweden.

Muchapondwa. E., F. Carlsson & G. Kölhin 2003. Can local communities in Zimbabwe be trusted with wildlife management? Application of CVM on the elephant in Mudzi Rural District. Unpublished paper, Department of Economics, Göteborg University, Sweden.

Norton-Griffith. M. 2007. How many wildebeest do you need? *World Economics* 8(2), 41–64.

Novelli, M., J.I. Barnes & M. Humavindu 2006. The other side of the tourism coin: consumptive tourism in Namibia. *Journal of Ecotourism* 5, 62–79.

Pagiola, S. & G. Platais 2007. *Payments for environmental services: from theory to practice*. World Bank, Washington.

Smith, A. 1997. *The wealth of nations*. Penguin, London.

Sutton, W. 1998. *The costs of living with elephants in Namibia*. Proceedings of the Workshop on Cooperative Regional Wildlife Management in Southern Africa, University of California, Davis, California, USA, August 13–14 1998, 57–71.

Sutton, W.R. 2001. The economics of elephant management in Namibia. Ph.D. thesis, University of California, Davis, California.

Sutton, W.R., D.M. Larson & L.S. Jarvis 2004. *A new approach for assessing the costs of living with wildlife in developing countries*. Research Discussion Paper No. 69, Directorate of Environmental Affairs, Ministry of Environment and Tourism, Windhoek, Namibia.

Turner, R.K., D.W. Pearce & I. Bateman 1994. *Environmental Economics. An Elementary Introduction*. Harvester Wheatsheaf, New York.

Turpie, J.K. 2003. The existence value of biodiversity in South Africa: how interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. *Ecological Economics* 46, 199–216.

Turpie, J.K. & A. Joubert 2001. Estimating potential impacts of a change in river quality on the tourism value Kruger National Park: an application of travel cost, contingent and conjoint valuation methods. *Water SA* 27(3), 387–398.

Turpie, J., J. Barnes, J. Arntzen, B. Nherera, G-M. Lange & B. Buzwani 2006. *Economic value of the Okavango Delta, Botswana, and implications for management*. Department of Environmental Affairs, Gaborone, Botswana.

ULG Northumbrian 2001. Economic analysis of commercial consumptive use of wildlife in Botswana. Unpublished report, Botswana Wildlife Management Association (BWMA), Maun, Botswana.

Van Aarde, R. & T. Jackson 2007. Megaparks for metapopulations: Addressing the causes of locally high elephant numbers in southern Africa. *Biological Conservation* 134, 289–297.

Van Aarde, R., T. Jackson & S. Ferreira 2006. Conservation science and elephant management in southern Africa. *South African Journal of Science* 102, 385–388.

Van Kooten, G. 1995. Elephant economics in the rough: an ivory trade model. Draft paper, Department of Economics, University of Victoria.

Van Kooten, G. & E. Bulte 2000. *The economics of nature*. Blackwell, Oxford.

Vredin, M. 1995. *Values of the African elephant in relation to conservation and exploitation*. Umeå Economic Studies No. 390, Department of Economics, Umeå University.

Vredin, M. 1997. *The African elephant: existence value and determinants of willingness to pay*. Umeå Economic Studies No. 441, Department of Economics, Umeå University.

Wilderness Conservancy n.d. Available at: http://www.wildernessconservancy.org/projects /ongoing.html.

WWF nd. History of the African elephant. Available at: http://www.wwf.org.uk/filelibrary/pdf /africanelephant_tl_0804.pdf .