AN ECONOMETRIC ANALYSIS OF THE WILDLIFE MARKET IN SOUTH AFRICA

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By

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Dedicated to my loving parents,

John and Heidi,

for all their invaluable inspiration and guidance,

and for showing me the value of life - all life

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ABBREVIATIONS

'000	thousands of
%	percent
2SLS	two-stage least squares
CA	conjoint analysis
CBA	cost-benefit analysis
CC Africa	Conservation Corporation Africa
CHASA	Confederation of Hunter's Associations of South Africa
CPI	consumer price index
CV	critical value
CVM	contingent valuation method
DEAT	Department of Environmental Affairs and Tourism
eg	*exempli gratia* (for example)
etc	*et cetera*
F-ratio	value of F determined using F-test
GDP	gross domestic product
GLS	generalised least squares
ha	hectare
I	income (as variable in equation)
ie	*id est* (that is)
IID	identically and independently distributed
IUCN	International Union for the Conservation of Nature
IV	instrumental variables
K	ecological carrying capacity (measured in LSU)
km^2	square kilometres
KNP	Kruger National Park
KZN	KwaZulu-Natal
KZN NCS	KwaZulu-Natal Nature Conservation Service
ln	natural logarithm
LSDV	least squares dummy variable
LSU	large stock unit
Ltd	Limited

NMNL	nested multinomial logit
NPV	net present value
OLS	ordinary least squares
P	Botswana pula
P	price (as variable in equation)
PA	protected area
pers comm	personal communication
PHASA	Professional Hunters' Association of South Africa
PPI	producer price index
prob	probability value
Q	quantity (as variable in equation)
R	South African rand (US$1 = R8.49 on 10/02/03)
R^2	coefficient of determination
RUM	random utility model
SA	South Africa
SE	standard error
SEEA	system of environmental and economic accounts
SNA	system of national accounts
SSPR	Sabi Sand Private Reserves
STATSSA	Statistics South Africa
SUR	seemingly unrelated regression
TCM	travel cost method
TEV	total economic value
t-test	statistical test using t-distribution
UK	United Kingdom
US	United States
US$	United States dollar
VAD	value added (to national income)
WTP	willingness to pay

EXECUTIVE SUMMARY

Since there has been no comprehensive nationwide economic study on the expanding wildlife market in South Africa, the present study ventures into a relatively unexplored economic terrain in the hope that more studies will be stimulated in that direction.

The study focuses, firstly, on presenting an economic overview of the current trends within the wildlife market of South Africa, attempting *inter alia* to demonstrate how the sustainable utilisation of wildlife can contribute to both conservation and the economy. Secondly, it is endeavoured to describe the demand for wildlife species at game auctions across South Africa, using a range of econometric modelling techniques.

An analysis of information from various sources reveals that wildlife areas can be estimated to cover around 14% of South Africa's surface area, with land being converted back to "nature" at an average of 6.7% per annum, for most of the previous decade. The gross economic value of the wildlife market in South Africa is conservatively estimated to be R1.4 billion (in 2001 prices). The wildlife markets that are significant in this regard are: hunting, wildlife-viewing tourism, live game sales, and wildlife products and processes.

In order to consolidate an understanding of the drivers of demand for various wildlife species, both cross-sectional and panel data demand models are estimated. Results indicate *inter alia* that rare wildlife species, such as the white rhino and sable antelope, can be seen as "luxury" and buyers are not price responsive for these, while for the oryx (a more commonly available species), buyers were price responsive (in 1999). It is submitted, however, that rarity is not the only factor affecting price responsiveness. Other factors such as buyer profile, competitive buying, price level and actual availability at auctions, can also play decisive roles. In conclusion, it is clear that an extensive analysis of the wildlife sector of South Africa must be undertaken to expose the major regulation and enforcement needs, and to determine where appropriate economic incentives can be applied.

CHAPTER 1

INTRODUCTION

1.1 BACKGROUND AND RATIONALE

Private investment in wildlife ranching has been on a steady increase in South Africa in the last few decades. The last eleven years, in particular, have seen a significant boom in the wildlife industry, for instance with the turnover at game auctions increasing from around R17 million[1] in 1991 to R81 million in 2001 (in constant 2000 prices). This translates to nearly a 400% increase in real gross auction income[2] over the period 1991 to 2001. The ever-increasing demand for wildlife ranching has driven up the prices of wildlife significantly. For instance, a hunting trip including a leopard trophy, cost in the region of US$ 75 000 in 2000 (Hunter, 2001), while five years ago it cost only US$2 880 ('t Sas Rolfes, 1995).

In South Africa, roughly 6% of the total land area (about seven million hectares) is set aside as state nature reserves and parks (Turpie & Siegfried, 1996). The Convention of Biodiversity states that each country should have a minimum of 10% of their land proclaimed for conservation of biodiversity. South Africa has potentially[3] at least around 14% (see section 3.1.2.2), due to the added number of private wildlife ranches that exist today. That means that a significant amount of biodiversity[4] and conservation[5] presently rest in the hands of private landowners. It is thus extremely important to establish whether the private wildlife

[1] Data from Eloff (2001) were deflated using the producer price index (PPI).
[2] Income is defined throughout as "money received over a period of time either as payment for work, goods, or services, or as profit on capital" (Rooney, 1999).
[3] Although all wildlife ranches do not necessarily have conservation of biodiversity as a main objective, the long-term viability of wildlife-viewing tourism and hunting depends on a healthy natural environment. It is thus in the ranchers' best interest to conserve the biodiversity on their ranches.
[4] As stated in the Convention of Biological Diversity, *biodiversity* refers to diversity within a species, diversity between species, and the diversity of an ecosystem.
[5] The three basic requirements of effective conservation as in the World Conservation Strategy are: to conserve life-support systems, to conserve biodiversity, and to ensure that the use of renewable resources is sustainable.

market is economically viable, both now and in the long term. Can the major role that it currently plays in the conservation of South Africa's biological diversity be sustained?

To date, there has ostensibly been no study on the nationwide economic perspective of this expanding wildlife market in South Africa. In 1987, Muir-Leresche (in Barnes, 1998: 90), discussing marketing of wildlife products in sub-Saharan Africa, stressed the need for more investment in research on the demand side of the equation within wildlife utilisation. Barnes (1998) avers that there is still a vast backlog of research to be done on both the supply and demand sides, within wildlife economics in Africa. He further contends that policy analysis[6] will ultimately fail if not backed by significant, ongoing contributions to empirical research in wildlife economics.

The present study therefore has two main aims:

1) To estimate the gross value of the wildlife market in South Africa, based on a review of available information. This includes providing an overview of the different wildlife markets in South Africa, showing their valuable contribution to both the economy and conservation.
2) To describe the demand for wildlife species at game auctions across South Africa, using a range of econometric modelling[7] techniques.

Due to the fact that wildlife ranching and its related wildlife markets have only recently been expanding at a steadily rapid pace, little research has been conducted in this area. There is currently no clear picture as to the full extent of the wildlife industry, and are no reliable estimates for the gross income of the wildlife market. This study aims to review all available information regarding the wildlife market in South Africa, in order to estimate a gross value.

[6] Barnes (1998) asserts that, due to the vast backlog in research, a number of people, particularly in South Africa have aimed to apply policy analysis tools directly to wildlife and environmental matters without undertaking research.

[7] *Econometrics* is the branch of economics concerned with the empirical estimation of economic relationships (Intriligator *et al*, 1996). Other options of methods within wildlife economics include cost-benefit analysis and linear programming.

It is necessary to value the contribution that this sector makes to the economy, for a number of reasons. These include *inter alia* the following:

- Values gained or lost under different resource-use options need to be considered if optimal decisions are to be made that yield *economic efficiency* (an overall net gain to society). Such decisions could be related to the trade-off between development and preservation of an area, or between two competing wildlife uses such as tourism and hunting.
- It is imperative that these values be included in *wildlife accounts* as part of the system of national accounts[8] (SNA).

Since no research has yet been conducted on the economic demand for live wildlife species in South Africa, the present study attempts to describe demand for wildlife species using a range of econometric techniques. Broadly, econometric models can have three principal purposes. These are: *structural analysis*, *forecasting* and *policy analysis*[9]. Due to data limitations, this study focuses on structural analysis only, with the purpose of understanding the drivers of demand, the sensitivity to price and income changes, and the variation in demand between various wildlife species. One of the econometric approaches followed can be classified as a relatively new *revealed preference technique*[10] for estimating the gross value of wildlife species traded at game auctions.

1.2 OUTLINE OF THE STUDY

This study is divided into two parts: (1) a review of the wildlife market in South Africa, culminating in an estimate of the gross value of the wildlife market, and (2) the estimation of

8 See section 2.2.5 for an overview of this concept.

9 Intrilligator *et al* (1996) define these as follows: *structural analysis* is the use of an empirically estimated model for the quantitative measurement of economic relationships; *forecasting* is the use of an estimated model to predict quantitative values of variables outside the sample of data observed; and *policy evaluation* is the use of an estimated model to choose between alternative policies.

10 The *revealed preference technique* is a valuation technique that involves observing people's willingness to pay (WTP) through their behaviour.

demand models for wildlife traded at game auctions in South Africa, utilising a variety of econometric modelling techniques.

The first part is covered in chapter 2, while the second part is covered in chapters 3 to 5.

Chapter 2 begins by presenting some theoretical aspects of wildlife economics, including *total economic value*. It continues with a brief review of the wildlife market in South Africa, focusing on the wildlife-viewing, hunting, and game auction sectors. The chapter culminates in an estimate of the gross value of the wildlife market and the gross value added per hectare.

In chapter 3, cross-sectional models are estimated with the aim of describing demand for wildlife species traded at game auctions across South Africa. Results of the cross-sectional demand models are presented using the *Oryx gazella*, commonly known as the oryx or gemsbok, as an example.

Chapter 4 introduces *panel data econometrics* and includes varied applications of the panel data demand models that have been reviewed for this study. A brief theoretical overview of the various model options that are suitable for determining demand for wildlife species, is then presented.

Chapter 5 includes the specification and estimation of *one-way error demand models* for wildlife species traded at auctions. The first section estimates demand for 14 wildlife species, while the second section estimates demand for four wildlife species. Models are subjected to various hypothesis tests, and price and income elasticity results are discussed.

Chapter 6 concludes the study with a synopsis of the themes and results of the various chapters.

CHAPTER 2

THE ECONOMICS OF THE WILDLIFE MARKET IN SOUTH AFRICA

2.1 INTRODUCTION

An economic review of the wildlife market in South Africa is here provided with the aim of arriving at an estimate for the gross value added from wildlife utilisation in South Africa. To place this study in perspective, some of the theories and concepts of wildlife economics are briefly elucidated.

2.2 WILDLIFE-ECONOMIC THEORY

2.2.1 Wildlife economics: Its position within environmental, ecological and conservation economics

Environmental economics focuses on those resources provided by nature that are indivisible, for example, an ecosystem or an estuary, while *natural resource economics* focuses on those natural resources that can be divided up into increasingly small units and can be allocated at the margin (Kahn, 1998). *Ecological economics* has been said to address the conflicts that arise between the interests of environmentalists and neoclassical economists, with the ultimate goal of achieving sustainable development[11] (Turpie & Siegfried, 1996). Czech (2000) asserts that ecological economics is a fundamentally alternative approach to

[11] The term *sustainable development* was coined by the World Commission on Environment and Development, and defined as development that 'meets the needs of the present without compromising the ability of future generations to meet their own needs.' Many definitions agree that it concerns the long-term health of the environment, and the welfare of future generations (Van Kooten, & Bulte, 2000).

mainstream or neoclassical economics, whereas environmental economics is an application of neoclassical theory to environmental issues.

Conservation economics, a focused area of ecological economics, addresses the economic issues relating to activities whose primary role is the conservation of biological diversity (Turpie & Siegfried, 1996). Due to the fact that *wildlife economics,* which focuses purely on the utilisation and preservation of wildlife, forms part of conservation economics, it thus also falls within the field of ecological economics.

2.2.2 The total economic value (TEV) of wildlife and wildlands

2.2.2.1 Economic perspective of value and prices

From an economic perspective, value is determined by people and by people's willingness to make trade-offs. With goods sold in a market, people's *willingness to pay* (WTP) is reflected by the monetary price that they pay for them. This trade-off need not involve money as such: other goods or opportunities may be sacrificed instead. In Figure 2-1, the inverse demand curve for a market good represents a marginal WTP function.

Total WTP for Q units is represented by 0AEQ. This represents the total value or total benefit associated with the Q units of the good. Net value is equal to area 0AE, which consists of consumer surplus[12] and producer surplus[13]. Otherwise stated, it is the area 0AEQ less the opportunity cost[14] 0EQ (Kahn, 1998). For non-market goods, a similar approach can be followed, although WTP must first be estimated with a variety of valuation techniques.

[12] *Consumer surplus* is the difference between a consumer's WTP for a certain quantity of a good, and the amount actually paid. It is the area under the demand curve and measures the surplus enjoyed by the demanders of a good. Consumer's surplus is often seen as a standard measure of utility (Varian, 1996).

[13] *Producer surplus* represents the benefit gained by society through having the resources utilised in their most productive manner (Kahn, 1998). It is the area above the supply curve and measures the surplus enjoyed by the suppliers of a good.

[14] *Opportunity cost* is the cost of the foregone opportunity, the next best alternative.

Economic value is therefore clearly not the same as revenue. Revenue is the price of a good times the quantity of the good sold, or area 0PEQ. Thus, total value exceeds revenue through the consumer surplus. Marginal revenue measures the change in revenue when one changes the quantity sold.

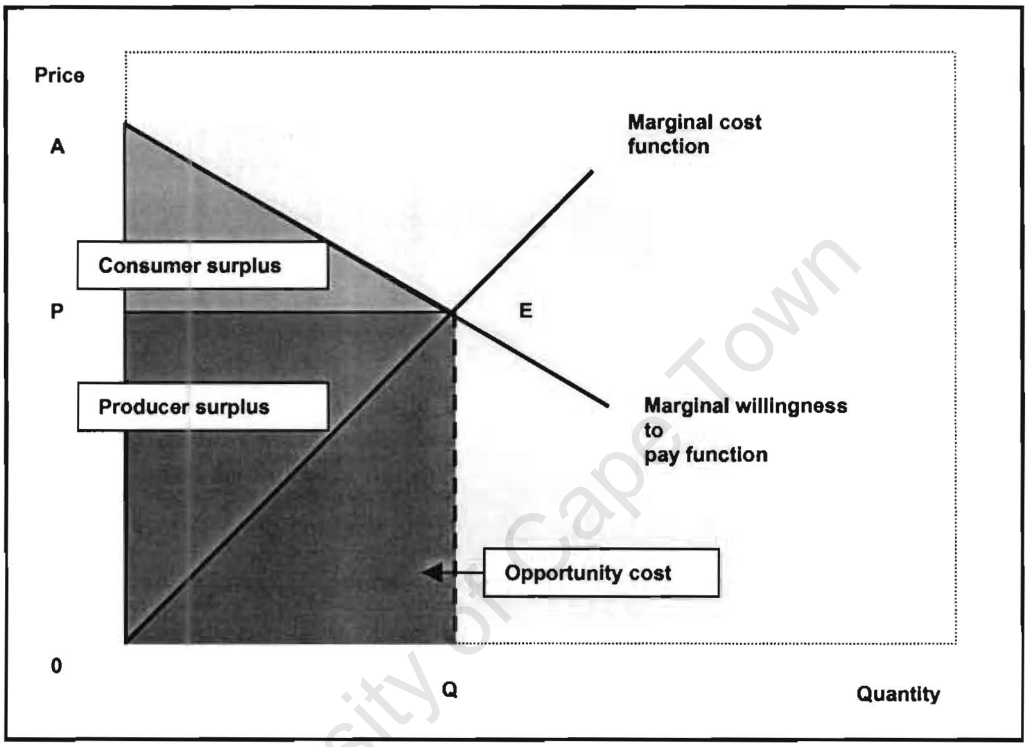

Figure 2-1: Graphical representation of inverse demand curve and various associated terms

2.2.2.2 The total economic value (TEV) concept

Of utmost importance within wildlife economics, is assessing the *total economic value* (TEV) of wildlife species and game reserves. Georgiou & Pearce (1997) furnish five reasons why valuing environmental goods and services is so important. These are: the significance of the environment in national development strategies; modifying the national accounts; setting national and sectoral priorities; project, programme and policy evaluation; and economic valuation and sustainable development.

TEV is a particularly *anthropocentric* approach, and includes use and non-use values. It views natural resources as existing for human benefit and utility only. *Biocentric* approaches incorporate the intrinsic value[15] of ecosystems and their components, and assume that ecosystem components have an intrinsic right to exist unaltered by humans (Kahn, 1998).

As can be seen in Table 2-1, use values are divided into direct (consumptive and non-consumptive), indirect and option values. Non-use values are, in turn, divided into existence and bequest, and option values. *Direct use values* are derived from the actual utilisation of a resource, and contribute tangible value in the form of income (Barnes, 2001). This is the only part of TEV that has previously been captured in national accounting, and thus in gross domestic product (GDP). For wildlife species, consumptive direct use would include hunting and the sale of venison, while non-consumptive direct use would include wildlife-viewing tourism. *Indirect use values* would include the value derived from ecosystem functioning and the role of various species within a specific ecosystem.

Existence value involves an individual's "utility" having increased merely from the knowledge of the existence of an environmental resource (e.g. an elephant) even though the individual may never derive a direct use from the resource. *Bequest value* refers to deriving utility by knowing that species are being conserved for future generations to benefit therefrom. An *option value* may be classified as a use or non-use value, as it values the desire to preserve the option of maintaining a resource for the future (Kahn, 1998).

Direct use values are captured from the following forms of *wildlife utilisation*, as enumerated by Barnes (1998):

- Wildlife-viewing tourism
- Safari-hunting tourism
- "Game meat" consumption (subsistence)
- Commercial wildlife production (harvesting, farming, ranching), and
- Trade in, and processing of, wildlife products

[15] *Intrinsic value* is that value not defined in terms of human satisfaction, but that exists due to nature's own "right to survive".

Table 2-1: Detailing the total economic value (TEV) concept

TOTAL ECONOMIC VALUE				
Use values		*Variable*	*Non-use values*	
Direct	**Indirect**	**Option**	**Existence**	**Bequest**
Consumptive use				
Examples: Venison meat and medicine	Processes which support other economic activities	Willingness to pay to maintain the option of future use	Species that are valued for their own right without reference to an economic use	Bequest resources for future generations
Non-consumptive use				
Examples: Wildlife viewing and recreation	Examples: Elephant converting woodland into grassland			

Barnes (1998) furnishes a schematic view of how the various wildlife uses in Botswana compare with biodiversity conservation (see Figure 2-2 below). Much of South Africa's wildlife ranches primarily focus on providing a service to wildlife-viewing tourists and/or hunters. This inevitably encourages ranchers to maintain fairly high levels of biodiversity, as their clients favour healthy ecosystems and diverse species. It can thus be argued that many of South Africa's wildlife ranches provide higher net economic values and are more in line with conservation than current wildlife ranching in Botswana.

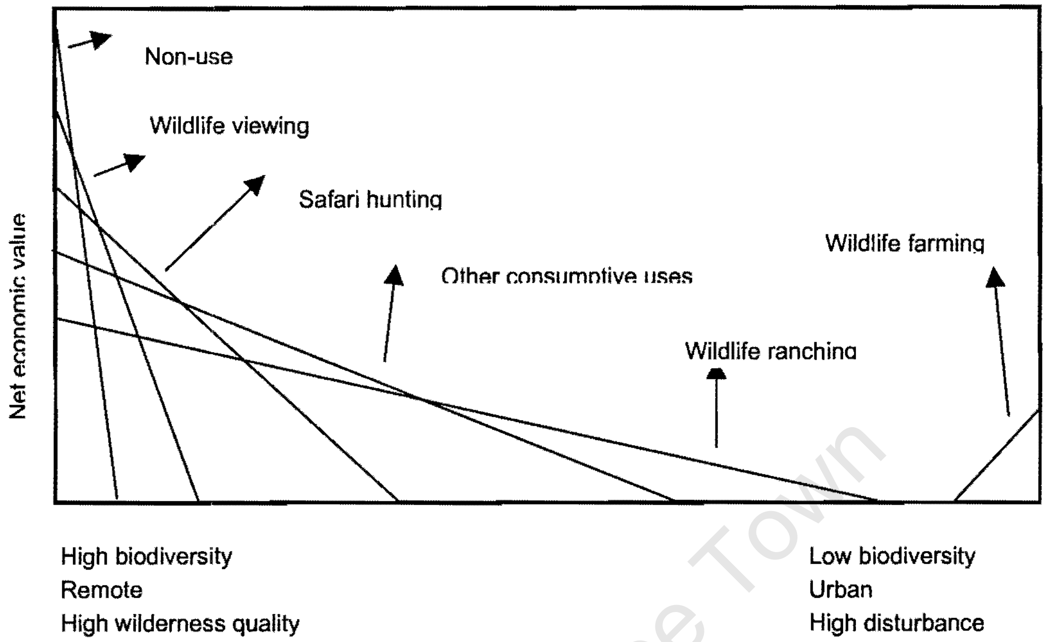

Figure 2-2: **Hypothetical land rent triangles for different wildlife and rangeland use activities in Botswana (along a gradient of environmental quality, showing zones where specific activities can be given priority)**

Source: Barnes (1998)

2.2.3 Wildlife markets: Capturing the value of biodiversity conservation

Trauger (1999, in Freese & Trauger, 2000) maintains that the conservation of biological diversity is emerging as a major societal goal. He contends that progress toward that goal is strongly influenced by the fact that commercial markets based on wildlife and other biodiversity values are large, growing and diversifying.

Although commercial markets for consumptive use of wildlife have led to population declines and a few extinctions, appropriate commercial markets are viewed as important for providing economic incentives to conserve wildlife and, more broadly, biodiversity. As Pearce and Moran (1994, in Freese & Trauger, 2000) aver, the emergence and diversification of new markets for biodiversity and the development of economic valuation techniques for non-market biodiversity values are providing new economic incentives and policy tools for biodiversity conservation.

Wildlife-viewing tourism and nonprofit conservation organisations are two major market mechanisms for wildlife conservation that are not based on consumptive use. The largest organisation in North America to take a direct market-based approach to conservation is The Nature Conservancy. As of 1998, they have used land acquisition and conservation easements to protect 4.25 million ha of land. In Southern Africa, this can be compared to Conservation Corporation Africa (CCA), which protects 70 500 ha of land for wildlife conservation in South Africa alone. A new market mechanism for biodiversity conservation, viz green-labelling, enables consumers to express their preference for products that are harvested or produced in a manner that benefits, or at least minimises degradation of, the natural environment and biodiversity. Producers and marketers of green-labeled products are then rewarded by obtaining a greater market share and by being able to charge a premium for their product (Freese & Trauger, 2000). Other biodiversity markets include bioprospecting and the trade of carbon offsets, which in return compensates landowners for protecting their natural environment (most commonly forests).

Markets favour specialisation in the production of those components of an ecosystem where the return on investment is greatest, whereas ecosystem health and biodiversity are generally placed at risk when nutrients and energy are redirected toward one or a few populations. The result is that biodiversity markets can be seen as a proverbial double-edged sword (Freese & Trauger, 2000). If efficiently and effectively managed, they can collectively constitute a very powerful tool for wildlife management and biodiversity conservation. However, if poorly managed, they can readily lead to overexploitation, degradation and loss of species.

2.2.3.1 Opportunity cost and land-conversions

When converting a natural area to an alternative use, part of the opportunity cost includes the loss of a number of the vital ecosystem goods and services. As Isaacs (2000) asserts, this loss can be termed an *externality*[16] of land conversion. Wildlife-related externalities of land-use practices are indeed considerable, including diminishing habitat, declining wildlife populations, and increasing rates of extinctions. As Krutilla (1967, in Isaacs, 2000) and Pearce & Moran (1994, in Isaacs, 2000) point out, if such costs are excluded or ignored, the production and consumption of marketable goods and services and the accompanying environmental deterioration, will be excessive.

The nonexclusive nature of many ecosystem functions means that markets for their provision will not likely arise. Randall (1987, in Isaacs, 2000) and Dixon & Sherman (1990, in Isaacs, 2000) aver that, in this situation, the value of these amenities will be ignored and the quantity or quality supplied will be insufficient.

In conclusion, increasing the returns from wildlife and protected areas will raise the opportunity cost of land conversion and result in reduced loss of natural area (and species populations).

2.2.3.2 Managing wildlife populations for maximum sustainable use

Game populations increase slowly at first, but once a critical number is reached, the growth rate is exponential and, accordingly, numbers increase rapidly. However, above a certain level, competition for resources, lower fertility and increased mortality result in a levelling-off of the population growth and net growth is therefore zero. A population at this level oscillates around a fluctuating upper level, which is the maximum biomass of game that an

[16] *Externalities* exist whenever the welfare of some agent, either a household or firm, depends directly on his or her activities and on activities under the control of some other agent (Tietenberg, 1996); or an externality exists "when action of one agent has an effect on an argument in the utility function (consumer) or production

area can sustain. This is called the *ecological carrying capacity*. The ecological carrying capacity is the level of the game population that is likely to exist in unmanaged large natural areas, such as the national parks. At this level, drought and disease can affect numbers quite considerably, with a consequent severe decline in population numbers. If a game population is maintained below the ecological carrying capacity by harvesting[17], and the net growth of the population is maximised, it is said to be at an *economic carrying capacity*. There is no fixed economic carrying capacity, but there is a point, called the *maximum sustainable yield*, where the population can be harvested, equal to half the intrinsic growth rate. In contrast to cattle farming, where the full ecological carrying capacity of the land is usually used (often referred to as K and usually measured in large stock units, LSUs per hectare), in wildlife ranching, only between 50% and 70% of K is used (ABSA, 2002). The annual harvesting of surplus game should ideally keep the game population at economic carrying capacity.

2.2.4 Techniques for valuing wildlife goods and services

A number of techniques exist that enable one to value non-market goods and services. They can be used to measure both use and non-use values. These monetary evaluation methods can be divided into demand and non-demand curve approaches (see Table 2-2). The two major classes of *demand curve techniques* are the revealed preference and stated preference techniques. *Revealed preference techniques* involve observing people's willingness to pay (WTP) through their behaviour. Examples are *hedonistic pricing* and *wage studies*, and the *travel cost method*. Stavins (1996) explores the use of *panel data* in valuing environmental goods and services (see section 4.2.1.1 for a review of this study). *Stated preference approaches* do not have a link to actual behaviour and rely on respondents' stated WTP, based on a range of hypothetical questions posed. This includes the *contingent valuation*

function (producer) of another agent and the first agent does not pay (negative) or receive (positive) an amount equal to the welfare effect on the second agent (Freeman III, 2003).

[17] Harvesting implies either the capture of game for resale at game auctions or, alternatively, the hunting of game for trophy or venison purposes. These days, game is not only bought and sold through live game auctions, but also over the internet. This has a number of specific advantages, such as less stress to the animals. For further details see www.bidorbuygame.co.za

method (CVM) and more recently, *conjoint analysis* (CA)[18] and *attribute-based methods. Non-demand curve techniques* include *inter alia* dose-response methods, replacement costs and mitigation behaviour.

Table 2-2: Valuation techniques

	VALUATION TECHNIQUES	
Market goods & services	**Market prices method** Using prevailing prices for goods & services	
	Efficiency (shadow) prices method Market prices adjusted for market imperfections and policy distortions	
Non-market goods & services *Demand curve approaches*	Revealed preference technique	> *Hedonic pricing method* > *Wage studies* > *Travel cost approach* > *On-site cost approach* > *Panel-data demand approach*
	Stated preference technique:	> *Contingent valuation methods* > *Contingent ranking or conjoint analysis*
Nondemand curve approaches	Cost based valuation	> *Damage costs* > *Prevention costs* > *Replacement costs* > *Relocation costs*
	Related good method	> *Bater exchange method* > *Direct substitute approach*
	Production function approach Estimating marginal value product	
	Benefit transfer approach Transfer values from other studies by Adjusting for country differences	

Source: Adapted from the classification as given by Barbier *et al* (1997)

[18] These choice-based methods evolved from evaluation methods for marketing and transportation. They present the respondent with a set of alternatives, each characterized by levels of attributes and a cost. (Haab & McConnell, 2002).

2.2.5 Wildlife accounts as part of the system of integrated environmental and economic accounts (SEEA)

Environmental assets

Wildlife species in reserves or wildlife ranches constitute a type of asset: an environmental asset. They should therefore be treated by their owners as any other accounting asset would be. They are assets to their owners, the country and the world, and thus their sustainability and economic contribution should be recorded and monitored. These crucial environmental assets have been absent from past national accounts (in South Africa and most other countries[19]), and any depreciation of natural capital stocks has been ignored, as also the loss in human well-being. Wildlife farmers should therefore account for all wildlife species (environmental assets) that provide actual or potential economic benefits to them, with a positive monetary value in the balance sheets (United Nations, 1993). Any decrease in the monetary value of the opening stock through an accounting period (not including sales) should be shown as a cost, or as a depreciation figure, in the income and expenditure account.

Satellite accounts

Data on the total economic value of the species and the reserves can be fed into natural resource accounts, within the *system of integrated environmental and economic accounts* (SEEA). These accounts are at present seen as satellite accounts to the *national accounts*, within the system of national accounts (SNA). Information collected on values derived from wildlife utilisation can be fed into biodiversity and tourism accounts, where *biodiversity accounts* would consist of wildlife accounts for the major fauna and flora of a region or country. The two major steps involved in natural resource accounting are the compilation of *physical* natural resource accounts, and the valuation of natural resources: compiling the

[19] Recently work in Botswana and Namibia has embraced the wildlife sector into the natural resource accounts (Barnes, 1998). This has not yet been done in South Africa.

monetary accounts (United Nations, 1993). As with national accounts, these satellite accounts would be updated each year, thus allowing one to track changes, weaknesses and progress.

Although South Africa is rich in biodiversity – indeed it is ultimately one of South Africa's richest assets - no work to date has focused on the compilation of biodiversity accounts[20] in South Africa, and very little economic valuation has been done. If South Africa wishes to maintain biodiversity and to minimise depletion and unsustainable exploitation, role-players need to understand the current situation and must isolate the vulnerabilities, thus focusing attention where it is most needed. One of the easiest ways of accomplishing this is through analysing the various biodiversity accounts and pinpointing the weaknesses. Thereafter, optimal environmental regulations, economic incentives, policies and management can be decided upon and instated. Ultimately, one needs to look at the national accounts and all of the satellite accounts, in an integrative and holistic manner, to ensure that the most sustainable development path is followed.

It is hoped that this study will contribute towards the compilation of wildlife accounts for South Africa, through an analysis of the market value of species, and the aggregation of incomes derived from wildlife utilisation in South Africa.

2.3 CAPTURING THESE VALUES: CASE EXAMPLES FROM AFRICA

Since Barnes (1998) furnishes a very detailed review of wildlife economics work in Africa, such a review will not feature here. A brief review for South Africa, however, will be provided in section 2.4, so as to indicate what research has been undertaken in relation to the economics of wildlife conservation, and where research needs may presently lie.

[20] In South Africa work has begun (relating to fynbos accounts), which focuses on the value of the flora within the Cape Floral Kingdom. No attempt has yet been made at estimating the value of the grasslands, savannah and bushveld biomes.

To gain some perspective, results from three African studies, with varied applications, are next provided to represent some of the current thinking[21] as regards the economics of wildlife conservation.

2.3.1 A study of direct use values in Botswana's wildlife

Barnes (1998) undertakes a comprehensive overview of the direct use values of Botswana's wildlife resources. This includes cost-benefit analysis (CBA) models for wildlife viewing, safari hunting, community-based wildlife use, wildlife ranching, ostrich farming, crocodile farming and ranching, elephant utilisation and wildlife product processing. In addition, he uses the contingent valuation method to estimate demand for wildlife-viewing tourism, and proposes a linear programming model for optimising the contribution of use activities to national income. Finally, various planning and policy options are analysed within the wildlife sector.

Table 2-3: Financial and economic comparisons from the results of the CBA models (1991 prices, in Pula)

	Internal rate of return (IRR) (over 10 years)		**Economic net present value (NPV)** (pula, @ 6% over 10 years)	
Wildlife use	Financial (%)	Economic (%)	per km^2 of land	per P '000 initial capital
Wildlife viewing	18	28	10 177	1,551
Safari hunting	16	38	694	2,230
Community[1]	26	67	589	5,225
Wildlife ranching	6	7	600	44
Cattle farming	9	2	< 0	< 0
Ostrich farming	18	19	2 301 548	950
Crocodile farming	18	11	2 565 398	525
Community use[1]	15	17	22	931

Source: Adapted from Barnes (1998)

Note: [1] Community (Chobe enclave project), and community use (Ngwaketse project)

[21] Some of the earliest work in Africa includes Mitchell's work on analysing the contribution of wildlife viewing to national income in Kenya (1968) and a cost-benefit analysis of Amboseli National Park (1969). Research in Africa has included studies on the value of hunting and its contribution to the viability of wildlife ranching, and the social value of game reserves (using shadow prices). More recent developments have been the application of revealed and stated preference techniques. A shadow price is "an estimate of the value that a good or service would have if a market could be established for it" (Stauth & Baskind, 1992).

The findings (in Table 2-3) confirm that wildlife use in Botswana can contribute positively to national income. Barnes (1998) maintains that this can happen without loss in biological diversity, and thus the wildlife market can assist both economic development and biodiversity conservation. He avers that the survival of wildlife in Botswana depends largely on its ability to generate economic value. Refer also to section 2.4.6.3.1 and Appendix 2.5 for aggregate values derived from the different wildlife sectors in Botswana.

Table 2-4 below shows the important contribution that safari hunting has made to the viability of the wildlife sector in Botswana.

Table 2-4: **Showing economic returns in the wildlife sector and the effect of the exclusion of consumptive use**

Case scenario	Economic NPV (pula '000 000) over 15 years	Economic NPV (pula '000 000) over 30 years
All wildlife uses		
Low cost	3.1	434.3
Base cost	-13.8	348.6
High cost	-32.5	234.9
Very high cost	-53.1	83.1
Non-consumptive wildlife uses		
Low cost	-68.2	205
Base cost	-85.1	119.3
High cost	-103.8	5.6
Very high cost	-124.4	-146.2

Source: Adapted from Barnes (1998)

2.3.2 Protected area subsidies: insufficient and ineffective

Hanks (2001) asserts that the perilous state of many of Africa's protected areas[22] (PAs) and valuable species, is largely due to the increasingly serious budgetary shortfalls of Africa's PAs. Additional complications include: ineffective and inexperienced management and

[22] As Hanks (2001) reports, protected areas are recognized by IUCN (1994) as "areas of land or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means".

leadership, coupled with lack of accountability; widespread reluctance to enforce legislation; lack of suitably qualified and motivated staff; competing needs such as health and housing; and civil unrest[23] and political volatility.

Table 2-5 shows budgetary shortfalls of PAs in various African countries, for which data have been sourced.

Table 2-5: Budgetary shortfalls in US$ per km^2 of PA

Country	PAs (km^2)	Shortfall (US$)	Shortfall/PA km^2
Angola	81 812		Substantial
Botswana	104 986		Capital dev. Shortfall
Burundi	1 462	220 230	151
Cameroon	20 977	1 243 000	59
Gabon	7 230		Substantial
Kenya	44 376	3 082 070	69
Mozambique	69 790	821 226	12
Namibia	112 153	7 412 000	66
Sierra Leone	1 531	2 000 000	1 306
South Africa	66 386	24 500 000	1 289[1]
Tanzania	268 030	2 465 681	9
Togo	4 290	288 719	67
Uganda	49 135	258 356	5

Data source: Hanks (2001)

Notes: [1] The 2001 loss in operating income for KNP, as stated in Hanks (2001), is used (and its size in km^2), due to total shortfall for SA not yet being known.

These budgetary shortfalls show the serious need for PAs to capture their value through alternative mechanisms and funding sources. Furthermore, these government subsidies are an ineffective means of earning income, as they allow PAs to be poorly managed, not needing to be competitive and economically viable. IUCN (2002) provides a number of options of funding sources, which need to be captured to finance PAs sufficiently. These include: *global mechanisms* including *inter alia* carbon offsets, global bonds, and bioprospecting; *national-level mechanisms* including tourist taxes, grants, "debt for nature" swaps, and public-good

[23] Kanyamibwa (1998) in Hanks (2001) reports for example that the Rwandan civil war (1990-1994) caused Akagera National Park to lose about 90% of its big mammals, as well as natural habitat due to refugees settling in the area.

environmental payments (to landowners); and *site-level mechanism* including two-tiered entrance fees to PAs, concessions, sale of wildlife and wildlife products, and filming rights.

2.3.3 Price sensitivity of wildlife-viewing tourism to Gonarezhou National Park, Zimbabwe

Goodwin *et al* (1998) also point out that government revenues are limited in Africa and that there are strong demands for expenditure on other public goods such as education, housing and health. They aver that where the principal beneficiaries of PAs are foreign tourists, rather than local people, the case for reforming public expenditure on national protected areas is especially strong.

Charges for access to public PAs have traditionally been based on the philosophy that national heritage should be available at no cost, or at a nominal price. Revenue maximisation has not been a primary objective of government policy and entrance charges have been set with social objectives in mind. Consequently, park revenues fall below park operating costs, and entrance fees are often below the level that visitors are willing to pay. The increase in international tourism has raised the issue of whether national governments, especially those of developing countries, should be subsidising the use of natural heritage by foreign tourists (usually from "rich" countries). Under-charging raises the cost for locals (through taxes) of maintaining the PAs, and fails to maximise revenue form international tourists (Goodwin *et al,* 1998).

Gonarezhou National Park operates on a system of dual pricing to prevent the exclusion of domestic tourists. *Dual pricing* recognises that domestic tourists already contribute to the maintenance of public PAs through the national treasury and that it is the domestic population that bears the opportunity costs of setting land aside for the conservation of biodiversity. Some foreign tourists may take offence at this strategy, but the argument that national parks and PAs are maintained at the expense of the nation's citizens needs to be advanced (Goodwin *et al,* 1998).

Goodwin *et al* (1998) consider the price sensitivity of foreign visitors to changes in the entrance fees of three national parks in developing countries. The results, from a series of WTP questions, are shown for Gonarezhou National Park in Table 2-6. Entrance fees (in 1997) were US$0.90 for domestic tourists and US$5 for international tourists. These fees are set bureaucratically, and are not in response to market forces.

As is evident, most respondents are prepared to pay twice the current entrance fee, but as soon as they would be asked to pay four times the current fee, their demand would decrease significantly. The demand can therefore be seen to be least elastic at low prices since the proportional fall in demand associated with the first price doubling (from current to x2) is less than the proportional fall for either of the next two price doublings. It is also clear from the results that the most revenue will be generated when entrance fees are around twice their current level. Charging levels more than this will not benefit total revenue, due to the decline in demand.

Table 2-6: Proportion of respondents WTP hypothetical increases in entrance fees, and estimated change in revenue

Entrance fee	Proportion of sample WTP (%)	Projected revenue (as a proportion of current revenue (%)
current	100	100
x2	79	158
x4	24	97
x8	8	65

Studies such as this show how public PAs can benefit from optimal pricing policies, by maximising their potential revenue earnings.

Having explored some of the concepts and themes within wildlife economics and having highlighted a variety of issues involved in this field, it is appropriate next to focus on the economics of wildlife in South Africa. In the following section, the wildlife market of South

Africa is reviewed, culminating in estimates for the gross value added to the economy from wildlife utilisation.

2.4 A BRIEF REVIEW OF THE WILDLIFE MARKET IN SOUTH AFRICA

2.4.1 Overview

2.4.1.1 Introduction

At present, there exists no easily available systematic representation of what is currently known or unknown, researched or unresearched, within the wildlife sector. It is, however, certain that a dynamic analysis and description of the entire wildlife market in the context of environmental, natural resource, and ecological economics are necessary, so that optimal wildlife management decisions and informed purchasing decisions are made, and so that appropriate environmental laws and policies may be instated. This study aims to contribute in this regard, although it focuses only on the economic angle of wildlife conservation, and not on the ecological and natural systems and interrelationships therewith. Throughout this chapter, it will be attempted to demonstrate what work has been done in South Africa in relation to the economic viability of wildlife utilisation, collaborating these efforts so as to derive estimates for the value of the wildlife sectors and the wildlife market.

In South Africa, the main wildlife sectors are: wildlife-viewing tourism, safari-hunting, the sale of wildlife at game auctions, breeding, and the sale and processing of wildlife products (such as trophies). Unlike Botswana, where most of the wildlands are state land[24], South

[24] Areas designated for, or proposed for, wildlife in Botswana (the "wildlife estate") amount to 127 000 square kilometres or 39 percent of the country's land surface. These areas include the national parks, game reserves and areas designated as "wildlife management areas" (WMAs). In addition, the whole country is divided into controlled hunting areas. The primary protected areas for wildlife are the national parks and game reserves. These four national parks and four game reserves occupy some 103 800 square kilometres or 17.8 percent of the country's land surface. No consumptive uses are permitted inside national parks, while game reserves have less constraints on resource use and can be de-proclaimed more easily. There are 12 WMAs, although only nine of these have been gazetted. Total land area zoned for WMAs is some 123 190 square kilometer or 21.2 percent of

Africa has a significant amount of protected land under private ownership (refer to section 2.4.1.2.2) As already mentioned, the extent of land used for wildlife ranching is still on the increase (more recently particularly in the Eastern Cape) and estimates have placed the expansion of the wildlife ranching industry at 25%[25] per annum over the last decade (ABSA, 2002).

2.4.1.2 Wildlife ranching in South Africa

Wildlife ranching and game farming are here defined as the keeping and management of wildlife species in semi-controlled or controlled conditions for consumptive and/or non-consumptive purposes. The distinction made here is that farming involves controlled breeding, whereas in ranching breeding is left to occur naturally (Barnes, 1998). In game farming, a farmer usually concentrates on the breeding of a specific animal species (for instance, blesbok and ostrich farming) or the breeding of rare animals (such as roan, sable or disease-free buffalo). Naturally, the level of control within wildlife ranching will differ. The larger the ranches are the more "natural" the environment for the wildlife.

In South Africa, mixed-species wildlife ranching usually incorporates wildlife-viewing tourism and/or safari hunting, as well as live sales (either private or at game auctions). One reason for the promotion of wildlife ranching is that it can, in appropriate regions replace cattle or sheep farming, as well as other alternative land uses, contributing more to national income and to the conservation of wildlife and wildlands.

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the country's land surface. Tribal land covers some 71 percent of the country while the second most important category of land tenure is state land, which covers 23 percent of the land surface. Freehold land covers only six percent of the country - this is under the control of private landowners (Barnes, 1998).

[25] No explanation is given as to how this estimate is derived. See sections 2.4.3.3 and 2.4.4.2 for the growth in the various different wildlife sectors and section 2.4.1.2.2 for the growth in the number of hectares under wildlife ranching in South Africa.

2.4.1.2.1 The trend towards wildlife ranching

Wildlife ranching is flourishing in South Africa. Unlike Botswana[26], South Africa is not constrained by poor market development, lack of capital and management skills, low natural game densities, low and unstable carrying capacities, and lack of land tenure security. Rather, a number of structural changes have taken place over the last decade favouring investment in wildlife ranching. As stated in ABSA (2002), these include:

- The deregulation of the agricultural sector.
- The agricultural sector lost its political power in Parliament after 1994, with the result that agricultural subsidies[27] have effectively disappeared.
- The new labour laws have increased costs and altered relationships between farmers and workers.
- Potential cost impact of AIDS and malaria that is re-emerging.
- Bush encroachment, with the result that economic carrying capacities are declining.
- Expensive cattle disease control.
- Stock losses have increased dramatically over the past two decades.
- Land claims.

Sub-Saharan Africa is among the poorest agricultural regions in the world. Europe's total agricultural output per hectare is at the top with $1 026/ha. The world average is $266/ha as compared to $69/ha in sub-Saharan Africa. This clearly shows that traditional agriculture is not a very suitable land practise in sub-Saharan Africa. Moreover, in the best cattle grazing areas of South Africa, one large stock unit (LSU) needs four hectares of grazing land per year on a sustainable basis, while in New Zealand one LSU needs only 0.67 ha. In short, the dry and unpredictable climate, and large areas of land with sandy infertile soils, make it difficult

[26] Barnes (1998) maintains that exclusive wildlife ranching occurs only on about 20 properties in Botswana, ranging in size from 6 000 to 75 000 hectares. In addition, around 250 properties involve primarily livestock production, although some supplementary game use takes place. These range in size from 3 000 to 100 000 hectares.

[27] South Africa's *net earnings in agriculture* have dropped from R110/ha in 1990 to R80/ha in 2000, and this can mainly be attributed to the decline in subsidies. The effective agricultural subsidy in South Africa is around 4%, while in Europe and Japan it is around 45% and 53% respectively (ABSA, 2002).

for South African cattle farmers to compete with farmers abroad without subsidies and import protection (ABSA, 2002). With such a low productivity, and in the light of the recent structural changes, it is evident that cattle farming is certainly not where South Africa's comparative advantage lies. Is it not possible that South Africa's most important asset is its biodiversity, including its wildlife and protected areas?

2.4.1.2.2 The extent of wildlife ranching

It is estimated that 20 million hectares of land have been transferred back to nature (ABSA, 2002). This is possible if it includes land that is either publicly or privately owned. Conservation areas under private ownership cover around 10 364 154 hectares, 8.49% (Eloff, 2000) of South Africa's surface area (see Table 2-7), while publicly owned conservation land covers around 6 638 600 hectares, 5.44%[28] (Hanks, 2001). Thus wildlife areas can be estimated at 13.93% of South Africa's surface area[29]. In addition, around 4.51%[30] of land is utilised under mixed cattle and wildlife ranching. Assuming around half of this is for wildlife ranching, one reaches around 20 million hectares of land that is set aside for conservation in South Africa.

Currently, there are about 5 000[31] wildlife ranches in South Africa, and more than 4 000 mixed wildlife and cattle ranches (ABSA, 2002). This is in sharp contrast to the 1960s, when there existed only around 10 wildlife ranches (Van Hoven & Zietsman, 1998). Van der Waal & Dekker (2000) show how there was a conspicuous increase in new exemption permit issues in Limpopo from 1983 to 1991, totalling 1 794. Growth in the number of wildlife ranches and in the number of hectares under wildlife ranching in South Africa has been 51%

[28] ABSA (2002) maintains that of the country's total land area 5.8% is for all officially declared conservation areas.
[29] This, however, does include some ranches, which also stock livestock. This is clear from Limpopo wildlife ranch estimates given by Eloff (2000) and Van der Waal & Dekker (2000).
[30] Wildlife ranches and mixed wildlife and livestock ranches cover 13% of South Africa's land area (ABSA, 2002). Taking this 13% and subtracting wildlife ranches, which cover 8.49%, leaves one with 4.51% for mixed ranches.
[31] As can be seen in Table 2-7, Eloff (2000) estimates 5 061 wildlife ranches. These do, however, include mixed farms.

and 47% respectively, from 1993 to 2000 (see Table 2-7). If the growth in the number of hectares under wildlife ranching (47%), were to indicate growth in the wildlife industry in South Africa, and was averaged out over the period 1993 to 2000, annual growth in the wildlife market could be estimated to be 6.71%[32].

Table 2-7: Summarising growth in the extent of private wildlife ranches in South Africa

Year	No. exempted wildlife ranches	Size (ha)	% of SA land area	% of agricultural land
1960	10[1]			
1993	3 357	7 039 992	5.77[3]	8.49
2000	5 061	10 364 154	8.49	12.5

Data sources: [1] Van Hoven & Zietsman (1998)

[2] Eloff (2000): data for 1993 and 2000

[3] Department of Environmental Affairs and Tourism (DEAT) national land-cover map (1999) - SAs surface area

Included below, in Table 2-8, is a representation of the number and coverage of wildlife ranches in each province of South Africa. 49% of all South African wildlife ranches are in Limpopo, followed by the Northern Cape (with 19.5%) and the Eastern Cape (with 12%).

Table 2-8: Showing the extent of wildlife ranching in South Africa by the year 2000, by province

Province	Number of exempted ranches	Hectares covered	% of total ranches	% of total ha
Limpopo	2 482[1]	3 325 652	49.04	32.09
Northern Cape	986	4 852 053	19.48	46.82
Eastern Cape	624	881 633	12.33	8.51
North West Province	340	364 935	6.72	3.52
Mpumalanga	205	276 016	4.05	2.66
Free State	180	147 743	3.56	1.43
Natal	90	168 841	1.78	1.63
Western Cape	82	265 205	1.62	2.56
Gauteng	72	82 076	1.42	0.79
TOTAL	***5 061***	***10 364 154***	***100***	***100***

Source: Adapted from Eloff (2000)

Note: [1] This figure can be compared to Van der Waal's & Dekker's (2000) estimate of 2 306 by 1998.

[32] Other growth rates that have been referred to include: "The wildlife ranching industry has been expanding at a rate of about 25% per annum during the last decade" (ABSA, 2002) As mentioned earlier, no reference is given as to how this was determined. Bernes-Lasserre (2001) reports that wildlife ranches have expanded by an average of 5.5% per annum - this is based on estimates of the number of wildlife ranches.

Graphical representations of the number of wildlife ranches per province and the size of private wildlife lands (in hectares) are provided in Figure 2-3 and Figure 2-4 respectively.

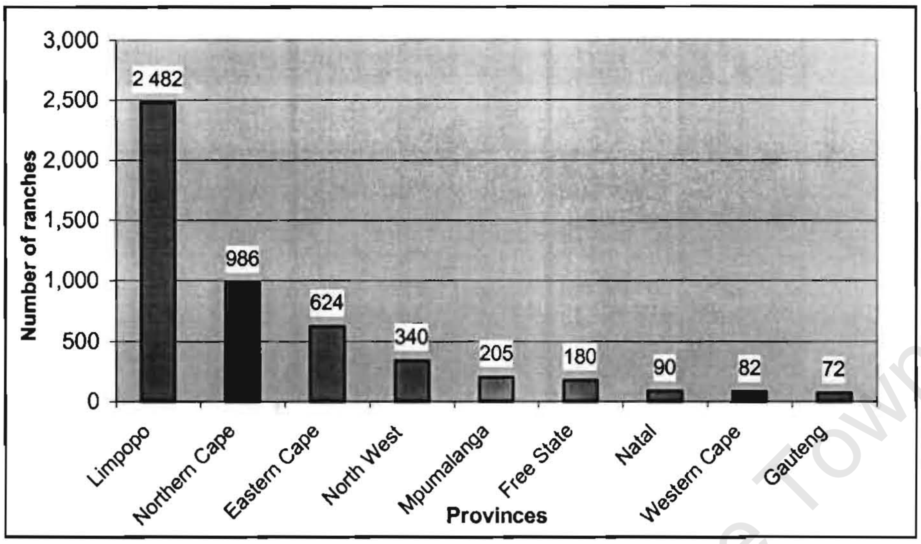

Figure 2-3: **The number of wildlife ranches per province by 2000**

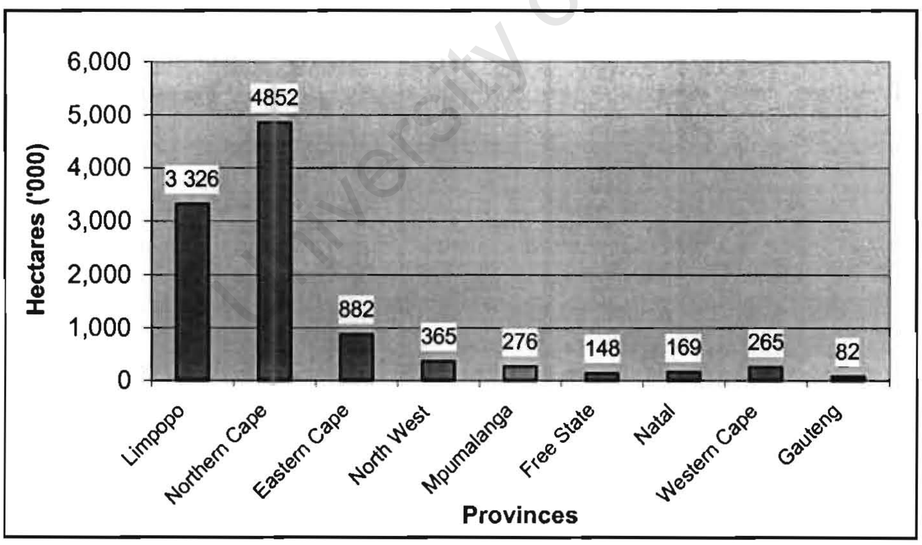

Figure 2-4: **The number of hectares under wildlife ranching per province by 2000**

2.4.1.3 A market approach to conserving biodiversity: Questions and concepts

Can the current market in South Africa be an example of '*biodiversity business*' in the sense that it promotes conservation of wildlife? If so, it needs to have both positive financial and biodiversity returns (ie it must ensure the conservation of biodiversity in the future).

Another way of explaining this is through the *'demonstration-capture' paradigm*. This firstly involves showing that ecosystems have economic value and provide valuable ecological services and goods, and secondly that these non-market values need to be captured, i.e. reaping real benefits and making conservation pay. The two combined are believed to be a 'powerful, practical and strategic tool for saving the environment' (Pearce, 1998). Demonstration without adequate capture is clearly insufficient, and businesses with rates of returns that are too low, will be forced to bring in more revenue or to shut down. On the other hand, those reserves or hunting ranches that for instance do not maintain healthy population levels of species or overgraze their lands, are neither conserving biodiversity nor ensuring that the use of wildlife is at a sustainable level. These can therefore not be seen as 'biodiversity businesses'.

This means that the market approach to conserving biodiversity has the potential to be very successful if appropriate internal business and environmental management exists. Appropriate regulation, enforcement, and policies within the wildlife industry are also of extreme importance. Indeed, 'thousands of small businesses, landowners, and threatened species are endangered by faulty regulations' (Suwyn, 1993, in Merrifield, 1996). Further analysis of the wildlife sector of South Africa could show us where the major regulation and enforcement needs are, as well as where appropriate economic incentives can be applied.

A review of the various wildlife markets is next undertaken, highlighting their extent and contribution to both the economy and conservation. Finally, an estimate is made of the total value of the wildlife market, and this is then briefly compared to Botswana and the United States.

2.4.1.4 Review of the economics of wildlife-ranching activities

A brief review of what work has been done on the economics of wildlife ranching in South Africa is provided. All South African work is surprisingly recent, with the first studies dating back to the mid-1980's. Tourism-specific studies will not be mentioned here, and can be found in Section 2.4.2.1.

Barnes (1998: 74) mentions various economic and financial studies that were completed over ten years ago - these are incorporated here. Several studies involve financial analysis of wildlife-ranching activities to determine profitability. Relatively brief financial analyses of wildlife-ranching budgets are undertaken by: Loxton Venn and Associates and Rural Development Services (Pty) Ltd (1985) in the North West Province; Davies and Chadwick (1991) in KwaZulu-Natal; and Grobler (1991) in the Eastern Cape. A fairly detailed analysis of wildlife-ranching budgets from a sample of 12 ranches is conducted by Behr (Joubert and Behr, 1986; Behr, 1988; Behr and Groenewald, 1990b). Berry (1986) undertakes a useful comparative study of different wildlife-use activities on a large private estate in the Northern Cape. Benson (1988), and Behr and Groenewald (1990a) provide physical, social, and gross income data on private-land wildlife users, using a detailed postal survey with some 1 500 responses.

More recently, a study undertaken by Van der Waal and Dekker (2000) assesses the extent and socio-economic impact of wildlife ranching in Limpopo. 1 844 questionnaires were distributed during 1998, of which only 118 (6.4% of total) were returned and completed. Information from exemption permits, issued for wildlife ranches from 1983 to 1998, was obtained from the Provincial Department of Agriculture, Land and Environment. This was used to determine the number, the spatial distribution and the surface area covered by wildlife ranches in Limpopo. Further results from this study have been summarised and can be found in section 2.4.6.2.1.

ABSA's economic research group have compiled the second edition of a book on "Game Ranch Profitability in Southern Africa" (ABSA, 2002). This includes quite an extensive coverage of the topic and thus much reference is made to this work throughout this chapter. ABSA (2002) developed optimisation models and estimated costs and benefits of different investments into wildlife ranching.

Eloff (2000, 2001) provides *inter alia* recent annual price and quantity data from game auctions, as well as calculations on various trends over time. An attempt at an estimate for gross income from the wildlife industry is also made for 2000.

Most recently, Ebedes *et al* (2002) edited a book on "Sustainable utilization – conservation in practice." This includes *inter alia* various economic realities of the wildlife industry. Bothma (2002) explores various sources of income from the wildlife ranching industry. Eloff (2002, in Bothma, 2002) provides a further estimate of the gross income from wildlife ranching[33].

2.4.2 The economics of wildlife-viewing tourism

2.4.2.1 A brief review

Once again, Barnes (1998) provides a thorough review of economic studies relating to wildlife-viewing tourism. These cover *inter alia* cost-benefit analyses relating to both the Kruger National Park and Pilansberg National Park, travel cost models, and more recently the contingent valuation method (CVM). In addition, 't Sas Rolfes (undated) provides an overview of conservation economics and analyses the value of the Kruger National Park, and its species. A marketing study by Schutte (2000) focuses on applying conjoint analysis to assess tourists' price sensitivity to changes in entrance prices into the Kruger National Park (KNP). Day (1996) applied a nested multinomial logit model to estimate the welfare benefits derived by visitors to the game reserves of KwaZulu-Natal. Unlike the travel cost method (TCM), this model focuses on the discrete decision between alternative recreational trips.

[33] These sources will not be referred to further, except in section 2.4.6.1, due to them being so recent.

2.4.2.2 Showcasing the benefits of tourism to conservation: In theory

2.4.2.2.1 Ecotourism and its contribution to biodiversity conservation

As Isaacs (2000) rightfully asserts, the term *ecotourism* is unfortunately somewhat vague. The term's being used with different meanings contributes to the vast differences in the estimates of international ecotourism expenditures, ranging from US$388 billion in 1988 to US$12 billion[34] in 1990 and US$416 billion[35] in 1994 (Isaacs, 2000).

It is submitted that ecotourism[36] should be defined as "tourism whose purpose is relatively undisturbingly to enjoy fauna and/or flora in their natural environment". As in Isaacs (2000), the definition here excludes hunting. Ecotourism is thus broader than wildlife-viewing tourism, as it does not merely focus on the enjoyment and appreciation of wild animals (particularly game species), but rather of any fauna or flora. Ecotourism in wildlife areas can therefore be said mostly to coincide with wildlife-viewing tourism.

The development of ecotourism is an effort to develop a market for one of the benefits of natural-area preservation, *viz* recreation. It is hoped that increasing the returns from ecotourism will raise the opportunity cost of land conversion and result in reduced loss of natural area. Isaacs (2000) mentions that Dixon & Sherman (1990) are of the impression that providing ecotourism is ideally consistent with providing other, complementary ecosystem functions. Conservationists hope that in this way ecotourism is a *proxy* market for other biodiversity values, including wildlife conservation.

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[34] Data source: Quammen, 1992

[35] Data source: Ecotourism society, unpublished data

[36] Ecology is derived from the Greek word *oikos* meaning house or habitation and *logos*, study. The Encarta dictionary defines ecology as "the study of interactions between living organisms and their natural or developed environment." Ecotourism has been defined as: "travelling to relatively undisturbed … areas with the specific objective of studying, admiring, and enjoying the scenery and its wild plants and animals [or] existing cultural manifestations found in the areas" (Boo *in* Luzar *et al*, 1995:545, found in Isaacs, 2000); or "a form of tourism that strives to minimize ecological or other damage to areas visited for their natural or cultural interest" (Rooney, 1999).

Negative externalities, however, must also be recognised in assessing and designing ecotourism services. These include: habitat deterioration, as land is converted to tourist facilities and road networks are developed; and tourist-related wastes and pollution, such as sewage, runoff, and air and noise pollution. In aiming to maximise profits an entrepreneur may not undertake costly measures to reduce erosion, pollution, wildlife endangerment, and congestion; hence the importance of appropriate regulations and/or economic incentives, ensuring that these negative externalities are maintained at desired levels.

2.4.2.2.2 Wildlife-viewing tourism: Economic, environmental and social objectives

Some of the factors that affect the attainment of an economically efficient level of tourism within the wildlife sector, are next discussed. As Barnes (1998) avers, the primary motive for the proclamation and development of national parks and game reserves has traditionally been the preservation of wild species and ecosystems. In the past it has generally been considered to be more of an ethical question than an economic one. More recently, however, ecological economists have begun to measure the economic value of wildlife conservation. This is due to the increasing need for conservation to pay for itself.

It is assumed here, as in Barnes (1998), that the objective for the development of wildlife-based tourism should be to maximise net total economic value (TEV) on a sustainable basis (over time). Maximisation of TEV therefore clearly requires a balance between the conflicting effects of preservation and utilisation. In addition, equity concerns need to be addressed and local communities need to be benefit from the tourism development.

Tourism carrying capacities

Lindsay (1986, in Barnes 1998: 88) defines the *tourism carrying capacity* as the physical, biological, social and psychological capacity of the protected area environment to support tourist activity without diminishing environmental quality or visitor satisfaction. Thus the economic tourism carrying capacity is that which provides the highest sustainable net economic value. Problems, however, can be reduced by regulating access to sites, and maintaining control of visitor activities and behaviour within a conserved area through the use of zoning and compulsory guiding (Goodwin *et al*, 1998).

Wildlife-viewing tourism can be developed to supply for one of two markets – upmarket, low-density tourism, or cheap-priced, high-density tourism. Upmarket, low-density tourism is generally associated with less environmental damage (higher preservation values) than cheap-priced, high-density tourism. As Barnes (1998) avers, the fact that, of the two, private landholders choose upmarket, low-density tourism, suggests that its financial return on investment is higher. This type of tourism is to be encouraged, where possible, although the danger of excluding poorer communities exists. High financial returns from high-priced tourism can, however, be distributed as income among local households and communities. This may be a more powerful tool for gaining support for conservation, particularly where local communities are poor.

A further solution to the "exclusion" problem may be to introduce *dual pricing* for national protected areas, which allows domestic visitors discounted entrance. By attracting domestic visitors, these facilities will encourage both local public support for protected areas and conservation, and the development of a domestic market to buffer seasonal and irregular fluctuations in international arrivals (Goodwin *et al*, 1998). In conclusion, while maximising TEV, a central objective should be to generate public support for conservation.

2.4.2.3 Showcasing the benefits of tourism to conservation: Some case examples

Methods for determining the economic value of the wildlife market

Hotelling noted that though the environmental resource is not itself a market good, the household in undertaking the recreational trip must also consume a number of complementary market goods (Day, 1996). For short trips, the household may need to spend no more than the costs of traveling to the recreational site, while for trips that involve staying overnight, one will need to purchase other market goods, such as accommodation. This allows one to determine values for the environmental resources (here, wildlife-viewing tourism) through estimating the gross revenues received from complementary goods. Gross economic value of the environmental resources can then be estimated either by taking gross revenue less costs of goods and services, or by determining value added[37] (VAD) ie their contribution to gross domestic product (GDP).

Towards ecotourism accounts: The macro picture

South Africa's total value of tourism was around R45 billion in 1998, or 6% of nominal GDP. The tourism sector employs 600 000 people, while another 500 000 jobs are created through the multiplier effect in related industries. Revenue from ecotourism has not been accurately estimated, but some estimates place the direct VAD from ecotourism[38] at least R1 billion. The multiplier effect stimulates around another R1 billion in industries such as airlines, taxidermy and hotels, bringing the estimated total VAD from ecotourism to R2 billion per annum (ABSA, 2002). As ecotourism is currently such a small percentage of the total market share of tourism, there is certainly significant potential for further growth. Included below are four examples of cases, which focus on different private and public game reserves/parks. These are provided here with the aim of contributing towards improved estimates for wildlife-viewing tourism income - a major portion of total ecotourism income for South Africa.

2.4.2.3.1 Case 1: Private Game Reserves and Sabi Sand Private Reserves (SSPRs)

In the early 1960's, photographic safaris were offered for the first time in this region at R20 per person per day, all inclusive. Prior to this, the region had hunting farms, and prior to that

[37] All VAD estimates are given in economic terms (unless specifically stated). This means that VAD estimates provided exclude government subsidies.

the region was a cattle-farming venture (Evans, 1995). The SSPRs[39], through their economic viability, have therefore made it possible to conserve this land for the last 40 years.

According to Evans (1995) at 100% occupancy, the various ecotourist lodges in the SSPRs generate, from accommodation income alone, R185 million per annum. At an average of 66% occupancy, this translates to the equivalent of R122 million per annum. Many private game reserves earn in excess of 90% of their revenue at certain times of the year in foreign currency, while the Kruger National Park generates only 10% of its revenue in foreign currency. Private game reserves earn these large amounts of foreign exchange, due to their popularity with the overseas travelers. Evans (1995) furnishes the following reasons why foreign tourists make use of the private game reserves: the assurance of seeing the "big 5" or at least most of its members; being taken on safaris on an open 4x4 in the company of a tracker, and a qualified ranger, who can teach one more about the flora and fauna; the luxurious accommodation and international level of service; and the high degree of exclusivity and personal pampering. It is for these reasons, then, that they can ask their high prices.

Another added benefit of the private game reserves is that their substantial international marketing teams and overseas sales agents do not benefit only private game reserves, but also tourism and nature-based tourism[40] across South Africa. This is because, of the 14 days' foreign exchange earned in South Africa, 11 days' is earned at other destinations (Evans, 1995).

2.4.2.3.2 Case 2: CC Africa's Private Game Reserves in South Africa

Conservation Corporation Africa (CC Africa) operates 27 luxury game or island lodges across six countries, and organises tours and safaris across eight African countries. In South

[38] It is not clear what ABSA regards as being included in "ecotourism".
[39] The SSPRs include MalaMala, Londolozi and SabiSabi

Africa, they currently operate five lodges in across the provinces Mpumalanga, KwaZulu-Natal and the Eastern Cape. Their conservation mission is "to demonstrate that a financially sound ecotourism business can make a significant contribution to the long-term conservation of land and wildlife in Africa, while also improving the lives of people" (CC Africa brochure). In this section, however, only the economic contributions are investigated, showing specifically the value captured from wildlife-viewing tourism for the South African lodges.

Tourism carrying capacities

In Barnes (1998) *tourism carrying capacities*[41] are assessed for high-priced tourism in Botswana's better quality protected areas. The size of each property involved is divided by the number of tourist beds available in order to obtain an estimate of carrying capacity. Tourism carrying capacities of between 510 and 980 hectares of viewing area per lodge bed is then calculated for the sample. These carrying capacities are relatively low as compared with an estimate for the Kruger National Park (KNP), where tourism is low-priced and there are 144 hectares per bed (Barnes, 1998). Tourism carrying capacities calculated for CC Africa lodges, range from 138ha/bed to 658ha/bed. Interestingly enough, even though Phinda Private Game Reserve is a high-priced reserve[42], it has only 156ha/bed, which is similar to that of KNP. The tourism carrying capacities for the South African CC Africa reserves can be found in Table 2-9.

[40] Nature-based tourism includes passive tourism, on the one hand, which includes the appreciation of scenic views and landscapes; and active tourism, on the other, which includes adventure tourism and ecotourism (Turpie *et al*, 2001).
[41] For comparison, information was gathered from nine successful, commercial game-viewing tourism operations in Botswana aimed at up-market clients from overseas and situated on private land. The average being 700 hectares per bed (Barnes, 1998).
[42] Phinda's lodge rates varied from US$340 (in 2002 low season) to US$590 (in 2002 high season) per person per night, all inclusive (www.ccafrica.com/destinations/southafrica/phinda/rates.asp).

Table 2-9: **Tourism carrying capacities per CC Africa lodge**

CCA lodge	Position	No of beds[1]	Hectares (ha)	ha/bed
Bongani	Bordering southern KNP	58	8 000	138
Kwandwe	Along Gr Fish River - ECape	24	15 800	658
Londolozi	In SabiSands	64	17 000	266
Ngala	In KNP, Orpen region	40	14 700	368
Phinda	Zululand, GSLWP[2]	96	15 000	156

Data source: CC Africa brochures (2001)

Notes: [1]Number of beds is assuming each room accomodates two people

[2]GSLWP is the Greater St Lucia Wetland Park, a natural world heritage site

Game lodge income and their value added

The real gross lodge income (in Rands 2000) for each CC Africa lodge, for the financial years ending June '01 and '02, is provided in Table 2-10, as well as real lodge income/ha. Total annual wildlife-viewing income (in real prices) from all five reserves was R76 million in 2001 and R49 million in 2002. A drop in real income of 35% is evident. This is due to the occupancy rates decreasing, with international tourism declining after the 11 September 2001 attacks. Londolozi and Phinda cater mostly for the international tourist, in particular American tourists, and thus these reserves experienced the greatest declines in income, of 37% and 50% respectively. This shows how sensitive the economic viability of game reserves is to the occupancy rates. This accords with Barnes (1998), who concludes that changes in the price of the tourism product affect the viability[43] of a tourism enterprise but not to the same extent as occupancy rates.

[43] Barnes also found through his (high-priced) tourist lodge enterprise model that at occupancies below 28 percent the enterprise is financially unattractive after 10 years.

Table 2-10: Real lodge income/ha for CC Africa lodges (in Rands 2000)

CCA lodge	Inc '01	Inc '02	Inc'01/ha	Inc'02/ha
Bongani	4 344 817	3 709 586	543	464
Kwandwe	0	3 364 852	0	511[1]
Londolozi	33 353 970	21 103 347	1 962	1 241
Ngala	10 476 947	7 056 627	713	480
Phinda	27 794 351	13 920 225	1 853	928

Data source: CC Africa headquarters

Notes: [1]Kwandwe only opened in October 2001 - income earned for 5 months was adjusted to annual income

Table 2-11 below shows the gross lodge income that could have been generated from the reserves at 100% occupancy rates for the year 2001/2002, which amounts to R118 million (in current prices)

Table 2-11: Gross income at 100% occupancy levels

CC Africa lodges	Potential Income
Bongani	6 148 800
Kwandwe	7 808 000
Londolozi	38 510 520
Ngala	16 704 240
Phinda	48 600 896
Total	**117 772 456**

Data source: CC Africa headquarters

Gross value added (VAD)/ha[44] from tourism lodges is estimated by subtracting an estimate[45] of cost of goods and services from gross income. Table 2-12 below shows how VAD/ha ranges from R325/ha for Bongani to R1 567/ha for Phinda. Average VAD/ha from CC Africa lodges is 764 and 563, for 2001 and 2002 respectively. Thus the economic value of wildlife-viewing tourism in these reserves is very high.

[44] Expressing income or VAD per hectare is an approach used by many environmental economists and researchers. It is not an ideal approach but is useful in that it allows for comparison between different (often competing) uses of the land. Its drawback, however, is that it does not allow for differentiating between the impact of, for instance, varying soil qualities or varying assets, such as luxury lodges versus cabins, on returns per hectare.

[45] This estimate is the average of KZN NCS's calculated percentages (see section 2.4.2.3.4) and the estimate of Turpie *et al* (2001), of 65%, within the Cape Floral Kingdom.

Table 2-12: Value added contributions per hectare for CC Africa lodges

CC Africa lodges	VAD'01/ha (Rands 2000)	VAD'02/ha (Rands 2000)
Bongani	380	325
Kwandwe	0	358
Londolozi	1 373	869
Ngala	499	336
Phinda	1 567	928

Data source: CC Africa headquarters

Consumer surplus

Willingness to pay by visitors for wildlife-viewing visits has not been determined for Phinda Game Reserve nor for CCAfrica's other reserves, and thus accurate consumer surpluses can also not be determined. Barnes (1998), through a 1992 questionnaire survey[46] of tourists in and around the northern national parks and game reserves of Botswana, found that the consumer surplus amounted to some 20.5% of trip expenditure. Barnes (1998: 98-99) maintains that a similar study, by Barnes *et al* (1997), in Namibia found the proportion to be 26%, while Brown *et al* (1995) using a different approach, the *dichotomous choice approach*, found the consumer surplus among tourists in Zimbabwe, to be 23% of expenditures. These are all surprisingly similar, taking into consideration the differences in approach as well as the potential questionnaire biases.

Thus, for a rough estimate of the aggregate consumer surplus derived from CC Africa's South African reserves, it would be possible to take 20.5%[47] of total annual tourist expenditures. This amounts to R15 573 867 and R10 076 700 for the financial years ending 2001 and 2002, respectively.

In conclusion, it is clear that these CC Africa lodges contribute significantly towards (GDP) and have a very high economic value per hectare. In addition, CC Africa supports local

[46] The contingent valuation approach was used here, and respondents were asked their actual expenditures in an open-ended question, and then their willingness to pay in excess of this was determined through the use of a payment card (Barnes, 1998).

community projects, while truly benefiting wildlife and nature conservation; their motto: "Care of the land, care of the wildlife, care of the people".

2.4.2.3.3 Case 3: National Parks and the Kruger National Park (KNP)

Figure 2-3 (below) shows the 18 South African National Parks, which cover 2.8% of South Africa (ABSA, 2002). Their combined gross revenue (in current prices) was R317.2 million and R322.6 million for financial years ending 2000 and 2001, respectively. This does not reflect growth in real terms. The cause of the decline in real income can be attributed to the devastating floods in KNP in February and March 2000 (Hanks, 2001).

Figure 2-3: Map of South Africa with the positions of the 18 National Parks http://www.parks-sa.co.za/

[47] Consumer surplus as % of trip expenditure for lodge users was 21.5%. It is assumed that visitors to these high-priced lodges would have similar WTPs as visitors to CC Africa lodges. 20.5% is thus conservative.

Income and VAD of KNP

KNP, South Africa's prime ecotourism destination, covers approximately 1.9 million hectares (van der Waal & Dekker, 2000). Gross income (in 2000 prices) was R194 788 274 in 1994, equating to a gross income per hectare of R103/ha. The VAD/ha can hence be estimated to be R72/ha.

Table 2-13: Gross income and VAD for KNP (in 2000 prices)

Using 1994 gross income:	Rands (2000 prices)
Gross income[1]	194 788 274
Gross income/ha	103
VAD[2]*/ha*	72

Notes: [1]Gross income for 1994 was taken from t 'sas Rolfes (undated)

[2] VAD uses the estimate of 70% of gross income contributing to VAD

It is also interesting to note that KNP generates only 10% of its revenue in foreign currency, whilst it is widely acknowledged that some private game reserves generate around 90% (Evans, 1996).

Sabi Sands Private Reserves, which are 2.5% the size of KNP, generate (at full occupancies) significantly more revenue than KNP's current potential (Evans, 1996). Evans (1996) reports that the highest occupancies ever reached by KNP were during the month of July 1994 when actual bed occupancies exceeded 97% for the month. He states that he was further informed that, although possible revenue for the month was R9.5 million, only R5.7 million (60% of potential) had been receipted. An explanation given for this was that discounts were granted during the busiest season of the year. He further notes that these concessions appear to cost KNP, annualised, the equivalent of approximately R46 million. This is occurring while losses are being made. Hanks (2001) points out that KNP operating losses were R10.9 million in 2000, and R24.5 million in 2001 (in current prices).

National Parks must imperatively remain economically viable to ensure their continued conservation efforts. Management needs to be more sensitive to the fact that their parks must maintain sufficient profits in order to be sustainable.

Price sensitivity

Schutte (1999) investigates the role of price sensitivity and pricing for local visitors' demand for accommodation in KNP. His study utilises conjoint analysis (CA), involving respondents' rating various pairwise comparisons on a nine-point scale. Results include average utility values for different attribute levels, relative attribute importance, and price elasticities.

The results indicate that the demand for entrance to and use of KNP by tourists was inelastic[48] with respect to the fee charge. As Lumsdon (1997, in Schutte, 1999: 74) maintains, any tourism offerings that are price sensitive, ie the level of demand is sensitive to price changes, are so mainly because of readily available substitutes. Those that are not price sensitive tend to be either at the luxury end of the market or instances where the supply is limited. The latter applies to KNP.

2.4.2.3.4 Case 4: KwaZulu-Natal Nature Conservation Service (KZN NCS)

KZN NCS have as their vision "to ensure that the natural heritage of the parks, wildlife, land and seascapes within KwaZulu-Natal are sensitively protected as a source of spiritual and physical sustenance for all its peoples". KZN NCS's protected areas and biosphere reserves are reported to have numbered 227 in 2000, covering 1 576 566 hectares (19%) of the KwaZulu-Natal province (KZN NCS Annual report, 2000).

Table 2-14 (below) depicts the different types of income generated by the KZN NCS's protected areas. Accommodation income contributes the most to self-generated gross income, with 49% and 42% for financial year ends 2000 and 1999, respectively. Some of the protected areas and wildlife non-use values are captured through donations. These total 6%

[48] Similarly, the introduction of a major increase in park fees to Moremi Game Reserve, in Botswana, had no effect on the general growth pattern in visitor numbers. In fact, fees collected increased dramatically. This indicates that demand for entrance, and use of Moremi was highly price-inelastic through the range of the fee change (Barnes, 1998).

for 2000 (R6 578 751), and 4% for 1999 (R5 054 877), of self-generated gross income. Self-generated gross income totals R108 631 523 and R135 351 511 for financial year ends 2000 and 1999, respectively.

Table 2-14: Gross income by type for KZN NCS (in 2000 prices)

	99/00	98/99
Gross income type		
Surplus from game sales	6 065 191	9 101 530
Accommodation	53 094 009	56 507 843
Admission	8 230 073	9 533 199
Trails, rides and tours	5 632 107	6 293 103
Rentals, hire and concessions	2 163 296	4 987 767
Natural resources	19 379 411	22 916 289
Sundry income	2 657 608	3 957 411
Interest received	4 831 078	16 999 490
Donations[1]	6 578 751	5 054 877
Gross self-generated income	108 631 523	135 351 511
State subsidy	154 789 105	146 089 502
Gross income (2000 prices)[2]	263 420 628	281 441 013
Gross income/ha[3] ***(incl subsidies)***	***167***	***179***
Gross income/ha (excl subsidies)	***69***	***86***

Data source: KZN NCS Annual report 1999/2000

Notes: All amounts have been deflated and are in real prices (in 2000 Rands)

[1] Donations capture some of the non-use values of the protected areas

[2] This includes the state subsidy

[3] Using 1 576 566 hectares

Gross income from sale of live game (including KZN's annual game auction) totalled R20 962 038 and R21 703 756 in real terms for financial year ends 2000 and 1999, respectively (data from KZN NCS Annual report 1999/2000). Table 2-15 shows how economic value added per hectare is determined. On average, across KZN NCS land VAD/ha is 127/ha and 146/ha for financial year ends 2000 and 1999, respectively. Value added is thus 34% and 62% of turnover.

Table 2-15: KZN NCS Value added per hectare (in 2000 prices)

Value added (2000 prices):	99/00	98/99
Turnover	278 317 476	294 043 240
Less: Cost of goods and services	-77 976 884	-64 201 824
Financial value added	*200 340 592*	*229 841 416*
Less: Subsidies	147 049 650	131 904 000
Economic value added	***53 290 942***	***97 937 416***
Value added/ha	***34***	***62***

Data source: KZN NCS Annual report 1999-2000

Table 2-16 provides estimates of value added from nature-based tourism per hectare. Nature-based tourism is seen to include passive and active nature-based tourism[49]. As mentioned earlier, passive nature-based tourism is the appreciation of scenic landscapes, while active nature-based tourism includes ecotourism and adventure tourism. Ecotourism is defined as in section 2.4.2.2.1 and adventure tourism includes the direct-use of natural landscapes, such as horse-riding, hiking and climbing.

Table 2-16: Estimating nature-based tourism per hectare (in 2000 prices)

Value added from nature-tourism	99/00	98/99
Accommodation	37 165 807	39 555 490
Admission	5 761 051	6 673 240
Trails, rides and tours	3 942 475	4 405 172
Gross VAD from nature-tourism	***46 869 333***	***50 633 903***
Nature-tourism VAD/ha	***30***	***32***

Data source: KZN NCS Annual report 1999-2000

Local communities have also benefited from the protected areas. Income earned by adjacent communities from employment totalled R104 863 402 for 1999/2000, while natural resources gathered by the communities in KZN NCS's protected areas were worth R6 545 350. In addition, KZN NCS was responsible for obtaining donations worth R31 250 000 for community conservation projects (KZN NCS Annual report, 2000).

[49] Breakdown taken from Turpie *et al* (2001).

2.4.3 The economics of hunting

2.4.3.1 A brief review

Barnes (1998: 124-125) reports that Ferrario (1985) mentions an analysis of the safari-hunting market by the Worldwide Outfitters Guide (Safari Club International), according to which the total African market was then split as follows: 42% South Africa, 25% Zimbabwe, 12% Namibia, eight percent Sudan, seven percent Botswana, two percent Tanzania, two percent Zambia, one percent Cameroon, one percent Central African Republic. Thus, South Africa then held a considerable portion (42%) of the African hunting market.

Barnes (1998) states that some of the earliest work relating to the economic value of safari hunting, was by Jahnke (1972) in Uganda. Jahnke derived useful measures of the economic value of safari hunting by tourists and recreational hunting by residents. A recent study in South Africa, on the financial values associated with recreational greywing francolin shooting in the Eastern Cape (Crowe *et al*, 1994), showed this activity to have financial importance to farmers and the economy, as well as indirect use value (Barnes, 1998: 75).

2.4.3.2 The contribution of hunting to conservation

Due to the income-generating potential of hunting and other wildlife-use activities, many tracts of land are restored to their semi-natural state, as private protected areas[50]. This conserves land and ecosystems that would otherwise not have been conserved. Even if the land is no longer "pristine" it is conserving biodiversity that would not have been conserved if the land were to have been converted to alternative land uses, such as irrigated agriculture or town development. "Harvesting", through hunting and/or the sale of live game, or the culling of species, ensures that species' population levels do not exceed ecological carrying

[50] Freese & Trauger (2000) report that the Migratory Bird Hunting Permit (from its inception, in 1934, to 1996) has generated more than US$442 million, which has been used to purchase or lease more than 1.6 million ha of wetlands.

capacities. When it becomes necessary to lower population levels, hunting can be a very beneficial option[51], as it generates significant income.

The contribution of hunting to the economic viability of the wildlife market

Safari hunting is generally considered one of the more profitable wildlife-use enterprises, and, in South Africa, safari and trophy hunting account for the biggest contribution to gross income within the wildlife sector[52]. Hunting results in both very high economic returns per unit of wildlife consumed and very high economic returns per tourist-day (Barnes, 1998). Trophy hunters, in particular, are highly profitable visitors, as their daily expenses (US$350 per hunting day excl accommodation) are much greater than that of the majority of wildlife-viewing tourists. Thirdly, hunters have to book well in advance, either directly with the owner or through hunting associations. This arrangement reduces business risk considerably and therefore makes long-term planning and wildlife management easier (ABSA, 2002).

Barnes (1998: 124) reports that Luxmoore (1985), Child (1984, 1988), Joubert and Behr (1986), Behr (1988), Behr and Groenewald (1990) all demonstrate that the financial viability of wildlife ranching in southern Africa is very often reliant on this form of wildlife use[53]. The financial viability of community-based natural resource management (CBNRM) projects in southern Africa have been very dependent on the inclusion of safari hunting, particularly that of elephant (Barnes, 1998). This means that often land would not be conserved if it were not for safari or community-based hunting. Thus hunting can assist in the conservation of

[51] In Tanzania in 1988, visiting hunters shot 4 000 animals, local culling amounted to 30 000, while crop protection removed 7 000. Poachers took an estimated 410 000 head of game that same year. Likewise, in the United Kingdom, 22 000 foxes were killed by hunting; 100 000 killed by cars; 80 000 shot by farmers; and 30 000 snared in 2000. Banning hunting may simply imply more killings with *inter alia* shotguns and snares. (ABSA, 2002).

[52] Freese & Trauger (2000) point out that according to Sparrowe (1993), approximately 75% of the costs of wildlife management at the state level in the United States is from hunting-related revenues.

[53] Refer to Table 2-4 in section 2.3. These results of Barnes (1998) show that the exclusion of hunting activities in Botswana lowered economic net present values (NPV) considerably. For high and very high costs, the exclusion of safari hunting causes the economic NPV after 30 years, to drop respectively from P234.9 million to P5.6 million, and from P83.1million to P-146.2 million.

wildlife species and can play a major role in the recovery of many African wildlife populations.

2.4.3.3 Hunting as a source of revenue from wildlife and its economic contributions to South Africa

In 2000, 2 120 professional hunters and 1 053[54] hunting outfitters were registered with the Professional Hunters Association of South Africa (PHASA), while by 2001, 3 589 professional hunters[55] and 2 671 hunting outfitters were registered. This represents an increase of 69% in registered professional hunters, and of 154% in registered hunting outfitters, over 2001. According to Potgieter (2001) there are around 200 000 South African hunters, of whom more than 20 000 are affiliated with the Confederation of Hunter's Associations of South Africa (CHASA).

On average, foreign trophy hunters hunt in South Africa for about 10 days, bag six animals, and spend around R40 000 on hunting fees (PHASA data). In addition, they pay the price of trophies successfully hunted. In 2001, 5 304 trophy hunters, and in 2000, 4 020 trophy hunters, visited South Africa, with the majority of these coming from the USA, followed by South America, Germany and Spain. South Africa is the major supplier of trophies in Africa, with around 85% of all Africa's trophy exports coming from South Africa (ABSA, 2002). Figures 2-5, 2-6 and 2-7 compare foreign clientele number, total animals hunted and hunting days by province, respectively. In 2001, Limpopo appears to have had the most foreign hunting clients (1 644) and the most number of animals hunted by foreign clientele (10 287), while Eastern Cape has had the second most, with 1 183 foreign hunting clients and 8 941 animals hunted.

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[54] PHASA data on foreign hunt statistics. All 2000 data represent PHASA data from the period 1/11/99 to 31/10/00, and similarly 2001 PHASA data represent data from 1/11/00 to 31/10/01.

[55] To become a professional hunter, one must complete a training course at a professional hunting school. On completion of this, one obtains a certificate and is licensed. To become a hunting outfitter then, one must have had three years of experience as a professional hunter, and one's facilities need first to be inspected (SA Hunting Guide 2002).

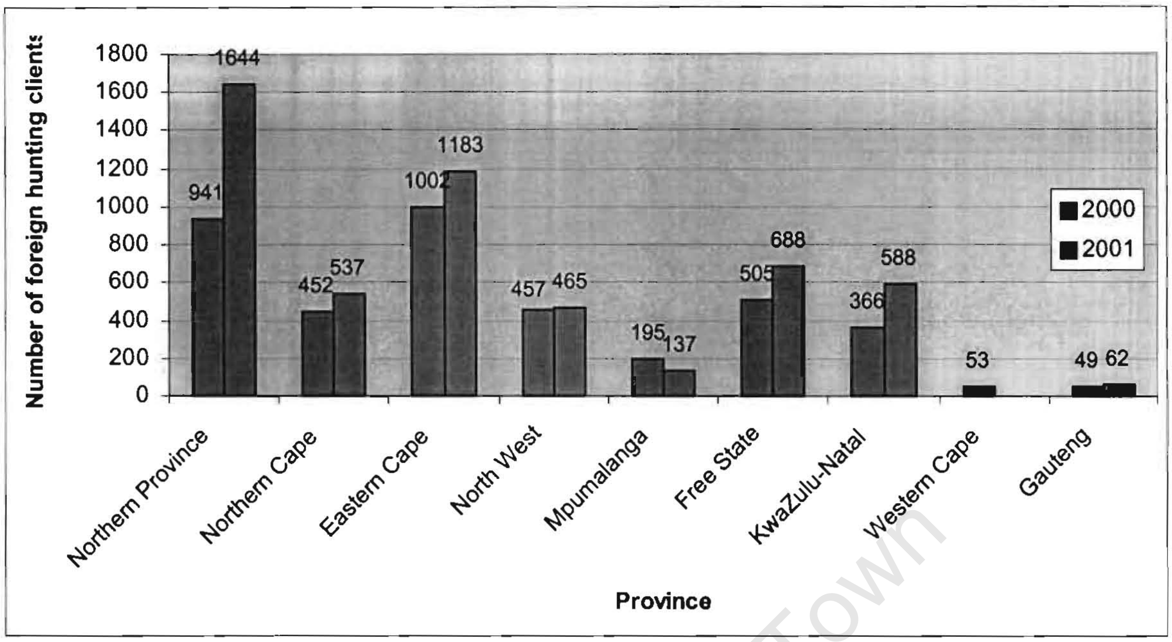

Figure 2-5: **Two-year comparison of the number of foreign hunting clients per province**

Data source: PHASA

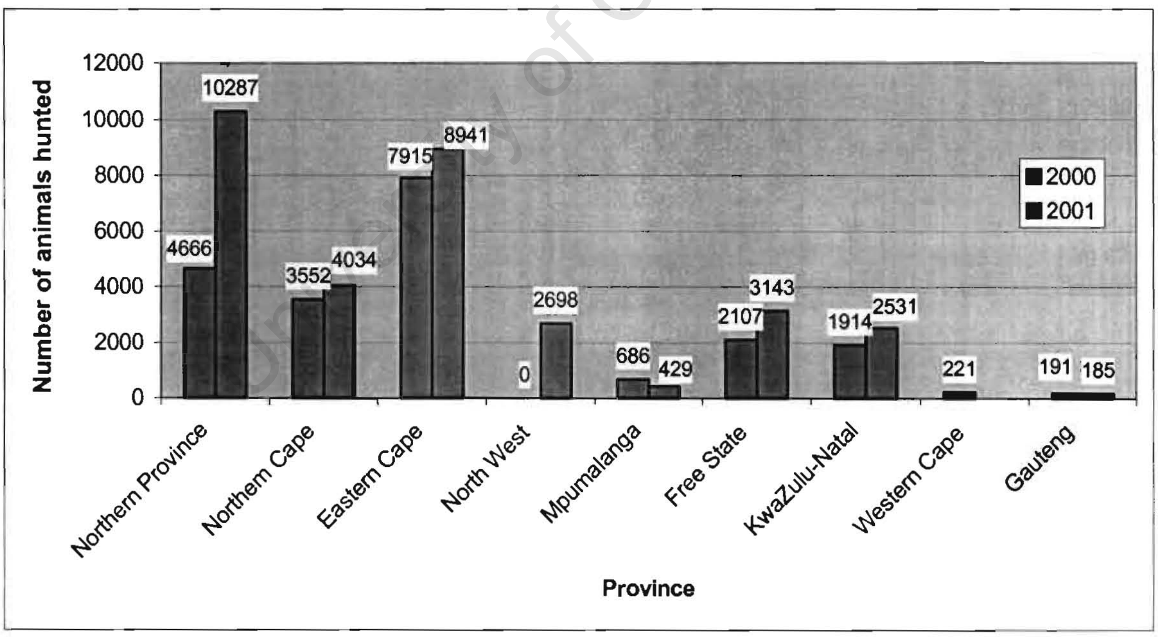

Figure 2-6: **Two-year comparison of the number of animals hunted by province**

Data source: PHASA

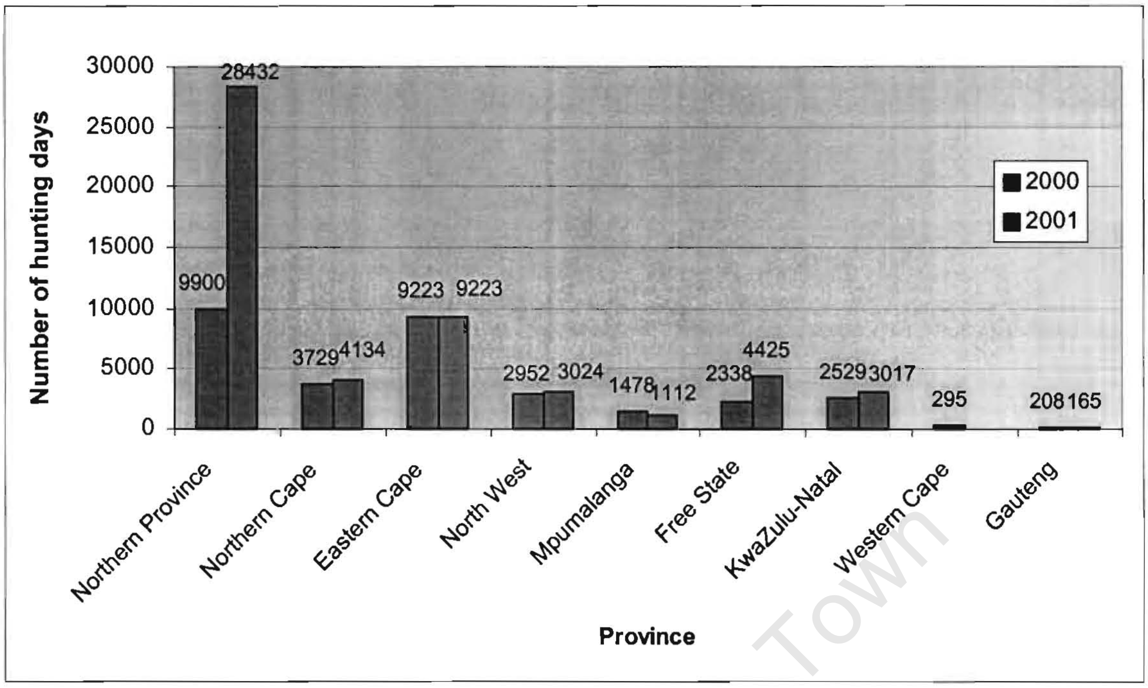

Figure 2-7: **Two-year comparison of the number of hunting days by province**
Data source: PHASA

From Figure 2-7 it is evident that foreign clients that visit wildlife ranches in the Eastern Cape are hunting similar quantities of wildlife species as clients in Limpopo, while staying for much shorter periods of time. In 2001, the average length of a hunt in Limpopo was 17.29 days, while in the Eastern Cape it was 7.8 days.

Table 2-17 shows the percentage of hunting days and animals hunted by foreign clients per province for 2000. An estimate of percentage income earned from foreign tourists by province, based on an equal weighting of the number of animals hunted and hunting days, is also provided.

Table 2-17: **Estimate of % of income earned by province from international hunters in 2000**

	% of hunting days	*% of animals*	*% of revenue*
Limpopo	30	22	26
Northern Cape	11	17	14
Eastern Cape	28	37	33
North West	9	8	9
Mpumalanga	5	3	4
Free State	7	10	9
KwaZulu-Natal	8	9	8
Western Cape	1	1	1
Gauteng	1	1	1

Data source: PHASA

Table 2-18 provides gross income figures (in current prices) from foreign hunting clients. The table shows how gross income (including: daily fees, accommodation and cost of

Table 2-18: **Gross hunting income from foreign clients for South Africa (with professional hunting outfitters)**

1 November 1999 to 31 October 2000	1 November 2000 to 31 October 2001
DAILY RATES	
32 652 hunting days @ $350/day R85 711 500	53 532 hunting days @ $350/day R173 124 280
TOTAL FOR ANIMALS HUNTED & ACCOMMODATION DURING HUNT	
R231 140 325	R414 080 066
GROSS HUNTING INCOME (current prices)	
R316 851 825	R587 204 346
GROSS HUNTING INCOME PER HECTARE (current prices)	
R31	R57
PROPOSED $10 LEVY[1] ON TROPHIES	
23 378 animals hunted R1 753 350	32 246 animals hunted R3 036 316

Data source: PHASA (data excludes taxidermy work)

Note: [1] Levy (to be imposed on all trophies) to be fed into PHASA's Wildlife Conservation Fund, which will be used in PHASA's efforts to promote and maintain the wise consumptive use of the natural resources of South Africa (SA Hunting Guide 2002).

[2] PHASA used an exchange rate of R7.50=US$1 (for 1999/2000) and R9.24=US$1 (for 2000/2001)

animals hunted) totals R317 million and R587 million, for the years ending 2000 and 2001, respectively. Average gross income per client is therefore calculated to be around R79 000 (in 2000) and R103 000 (in 2001).

Figure 2-8 depicts the considerable increase in foreign exchange earned over recent years from foreign hunters in South Africa. In real terms (in Rands 2000) the growth has been at an average of 47.5%[56] per annum for the period 1996 to 2001.

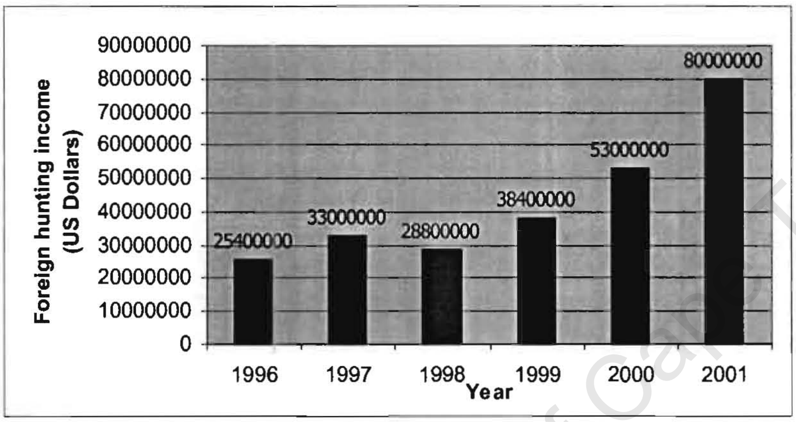

Figure 2-8: **Gross hunting and taxidermy income from foreign hunters in dollars (US$)**
Data source: PHASA (2002)

Table 2-19 provides the estimated gross incomes for 2000 and 2001 (in constant 2000 prices) derived from trophy and venison hunting (including local and foreign hunters) in South Africa. Gross value added from hunting, expressed per hectare, is R52/ha and R67/ha for 2000 and 2001 respectively. These VAD/ha estimates are conservative, as the calculation assumes that all hectares under wildlife ranching are utilised for hunting - this is not the case. It is clear from these estimates that hunting, as a form of wildlife utilisation, contributes a significant amount of VAD to the economy. In addition, taxidermy work for foreign clients generated R348 million (in constant prices) in 2001.

[56] Calculated using US$/Rand exchange rates as at 30 December each year and the PPI with a 2000 base year.

Table 2-19: **Estimates for gross income and VAD per hectare for local and foreign clients (constant 2000 prices)**

1 November 1999 to 31 October 2000	1 November 2000 to 31 October 2001
GROSS HUNTING INCOME (constant 2000 prices)	
R766 851 825	R998 789 108
GROSS HUNTING INCOME PER HECTARE (constant prices)	
R74	R96
GROSS VALUE ADDED PER HECTARE (constant prices)	
R52	R67
GROSS VALUE ADDED FROM HUNTING (constant 2000 prices)	
R536 796 278	R699 152 376

Data sources: PHASA and Eloff (2000)

Notes: [1] Prices were deflated assuming a 7% change in the PPI index from 2000 to 2001

[2] Hectares were assumed to be 10 364 154, Eloff's (2000) estimate of wildlife ranches in 2000

[3] Local hunting income, of R450 million from Eloff (2000), is assumed to be the same in real prices in 2001

2.4.3.4 The role of ethics and the legal system in wildlife sustainability

Hunting does harbour potential dangers. Freese & Trauger (2000) maintain that the selective hunting of males, particularly trophy males, may have multiple negative effects on genetic diversity, the health of targeted populations, and ecological processes. For instance, some are of the opinion that lion hunting is likely to tear apart the social fabric of a particular lion pride for years to come, and the lion population may go into a downward spiral (ABSA, 2002). It is thus advised that the population levels and genetic diversity of hunted species be monitored (particularly species on, or near to being on, the endangered list). The decision can then be made as to which species should be allowed to be hunted, depending on their population levels and the viability of different breeding groups. Hunting should only be advocated if it will assist in their conservation. If it leads, in any way, to their demise or to nonviable population groups, it should be further controlled through appropriate regulations, or it should be banned[57].

Regarding ethical hunting, although unethical hunters may be only a small percentage of the total hunting community, they cause disproportional damage to hunting. The Professional Hunter's Association of South Africa (PHASA) therefore has as an objective to assist and promote ethical hunting. PHASA *inter alia* advocates "fair chase" and promotes the sustainable utilisation of wildlife in their code of conduct (South African Hunting Guide, 2002). The view has been expressed that a code of conduct that effectively enforces minimum standards in the hunting industry is still clearly lacking in Africa (ABSA, 2002). It is submitted that it is also clear that there is a significant dearth of laws pertaining to hunting and the enforcement thereof.

2.4.4 The economics of live sales

2.4.4.1 Live sales as a contributor to wildlife conservation

The sale of live animals, if regulated and enforced appropriately and sufficiently, can certainly contribute to wildlife conservation. Firstly, if there is a market where revenue can be earned from the sale of wildlife, there is an incentive for landowners to maintain healthy ecosystems and stable population levels. In this way, natural breeding will take place and selling wildlife at "sustainable off-take levels" is possible. In addition, this market enables land that was previously used for alternative land uses, to be restocked with suitable wildlife

[57] Lion hunting (trophy males) in Botswana was banned in 2001, causing an apparent loss of income of \$2.5 million (from the removal of 53 lions, previously on the quota list for 2001). This was apparently due to the uncertainty around the effect of hunting males on the social dynamics of the prides and the consequences for breeding populations, and the possible effects of the feline immunodeficiency virus (FIV). It is estimated that lion populations number between 2 500 and 3 000 in Botswana. The estimates for Africa are around 15 000 to 20 000, with only three healthy populations remaining: Northern Botswana, Serengeti and the Kruger National Park (Michler, 2001). Furthermore, 1 000 lions died in the Serengeti from canine distemper (Michler, 2001) – with vulnerabilities like that should we not protect the lion populations from additional sources of mortality, until more is known about diseases such as FIV? Considering that estimates for leopards in South Africa range from 500 to 1 500 (Bezuidenhout, 1999), is the current practice of leopard trophy hunting sustainable, or could it lead to major population decreases and their ultimate extinction?

for that region. In this way, various species can be brought back into regions where they had once become extinct. Another potential benefit is the improvement in genetic diversity within a species, ultimately making the populations more resilient. Without the trade of live game and the relocation of species, many populations confined to smaller game reserves would become inferior and weaker.

Auction prices

There may be significant differences between auction, venison hunting and trophy hunting prices. Appendix 2.1 provides a comparison of live auction prices and Eastern Cape trophy prices for various wildlife species, showing trophy prices often to be twice as much as the live auction prices. For some species, especially those of the cat family, trophy prices are significantly higher than live prices. From the seller's perspective, the gross income obtained from game at auctions is decreased by the cost of game capture. Transport costs and losses, however, are borne by the buyer (Game catalogue, 2002). Trophy hunters from abroad usually pay high prices. However, the natural supply of trophy-quality animals is usually limited, so that the overall impact on the cash flow of a ranch for this source is not that large, especially for small wildlife ranches. Prices for game species are not uniform across one species. Many factors play a role in the marginal price differences between two individuals of the same species. These include: characteristics of the species, such as age, sex, horn size, health, injuries, and whether it is part of a breeding group or not; and other factors such as location, availability, capturing, transport, season and reputation (ABSA, 2002).

2.4.4.2 The South African game auctions

The last ten years, in particular, have seen a significant boom in the private wildlife industry, with the turnover at game auctions (in real prices) increasing from R17 million in 1991 to R81 million in 2001. This translates to an average annual increase in income of 18.83%[58], or

[58] Calculated using annual auction income of each year from 1991 to 2000.

a five-fold increase over the last 11 years. A third of all game sales are believed to be sold at game auctions across South Africa, which numbered[59] 38, 43, 27, and 34, from 1999 to 2002, respectively. According to Eloff (2000), total annual live sales are therefore estimated at around R180 million (in constant 2000 prices). Figure 2-9 clearly depicts how the turnover at game auctions has increased considerably over the last decade. Game prices[60] and gross annual income have been deflated using the producer price index (PPI), with 2000 as the base year.

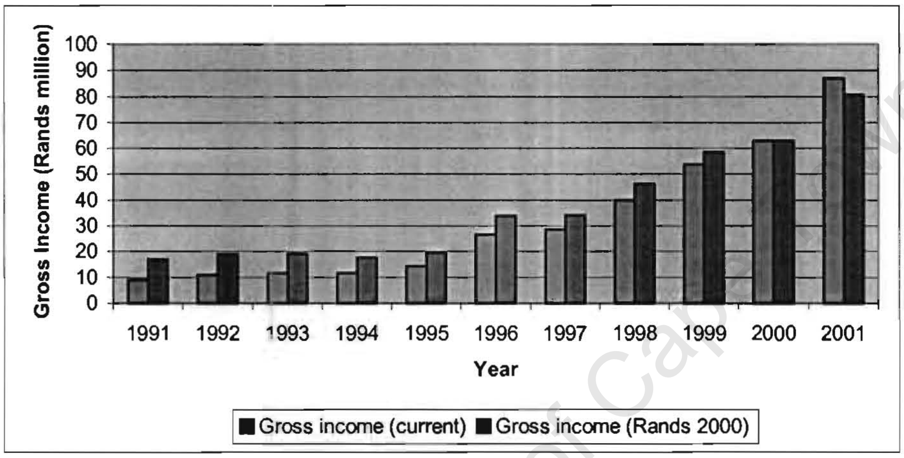

Figure 2-9: **Annual gross income at game auctions across South Africa (in current and constant 2000 prices)**

In nominal terms, the mean price of game increased by 11.3% per year since 1990, or 1.7% per year in real terms (ABSA, 2002). Appendix 2.2 provides 11-year averages of quantity demanded and average prices for 14 wildlife species (for which demand models are estimated in Chapter 5). From this, it is clear that scarcity plays a major role in the determination of the prices for the different wildlife species. Appendix 5.1 also provides graphical representations of the average prices and quantities demanded for these species over time. These graphs show how, for many of the species, both the prices and quantity sold have been increasing over the ten-year period 1991 to 2001.

[59] Frequency of game auctions was totaled from game auction information in monthly *Game & Hunt* magazines. These numbers add live and catalogue sales together, if occurring on the same day at the same venue. Gross income from the 34 game auctions in 2002 totals R77 544 554 in current prices (2002 data from www.sagro.org.co.za).

Figure 2-10 shows how the real prices of rare game species are still increasing. This means that the market for these species has not yet reached maturity. Real prices for disease-free buffalo seem to be stabilizing – this is most probably due to a decrease in their relative scarcity due to specialized buffalo farming[61] and breeding.

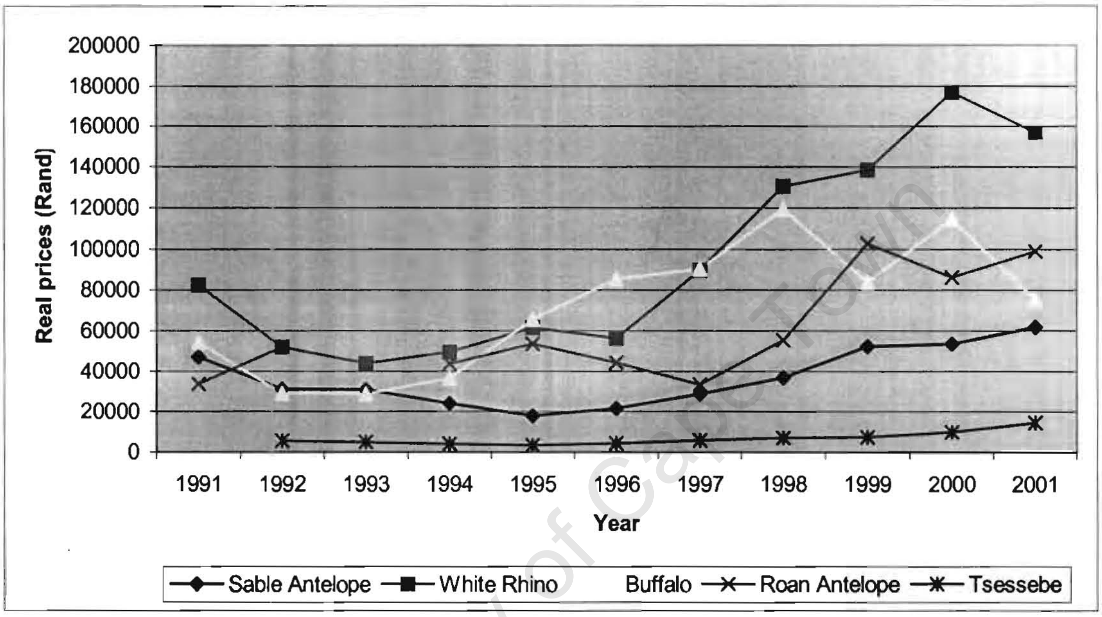

Figure 2-10: Trend in real prices of rare game species over time

Data source: Based on graphical representations from Prof T. Eloff

The drop in real prices between 1992 and 1995, particularly visible in Figure 2-11, is due to severe drought and economic recession at that time (Standard Bank, 2000). Figure 2-11 shows how the real prices of some of the less rare species are being maintained. The real price of nyala, however, is still on the increase.

[60] Annual game auction data from 1991 to 2001, including data on average prices and quantities sold of wildlife, were supplied by Prof Eloff.

[61] In ABSA (2002) under certain assumptions, it was estimated that returns on buffalo farming (as calculated by the real internal rate of return, IRR) range from 23% to 17% per annum. Due to this being a financially successful investment, disease-free buffalos are increasing in number.

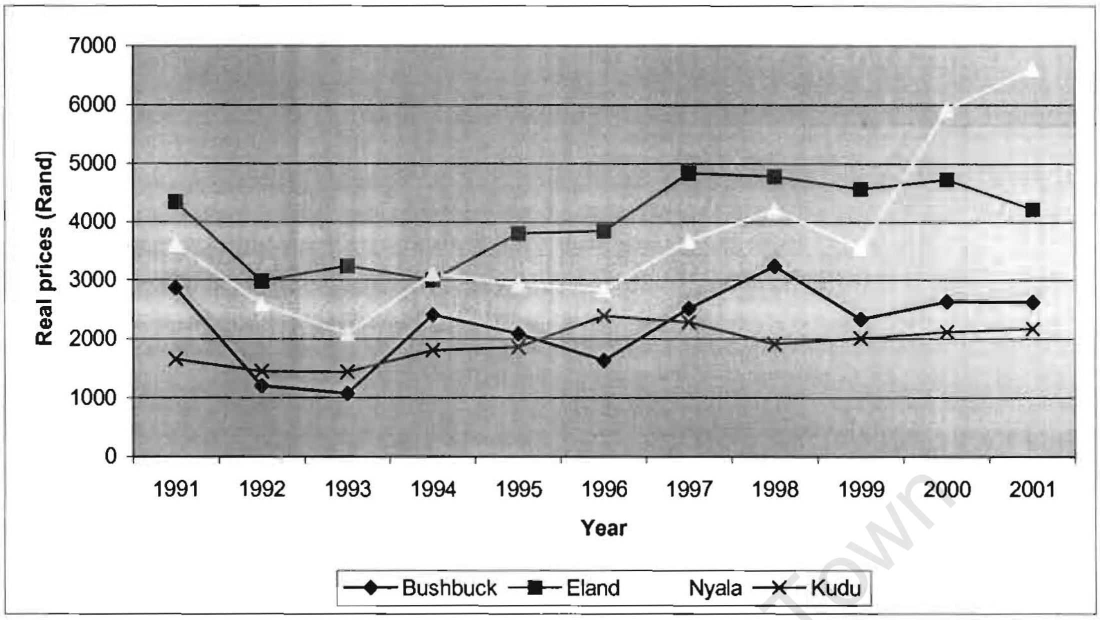

Figure 2-11: Trend in real prices of selected game species over time

Real prices of the more abundant game species have generally been declining since 1997/1998 (see Figure 2-12). This suggests that the market is nearing maturity for the abundant game species. As the Standard Bank (2000) study reports, the game market must be

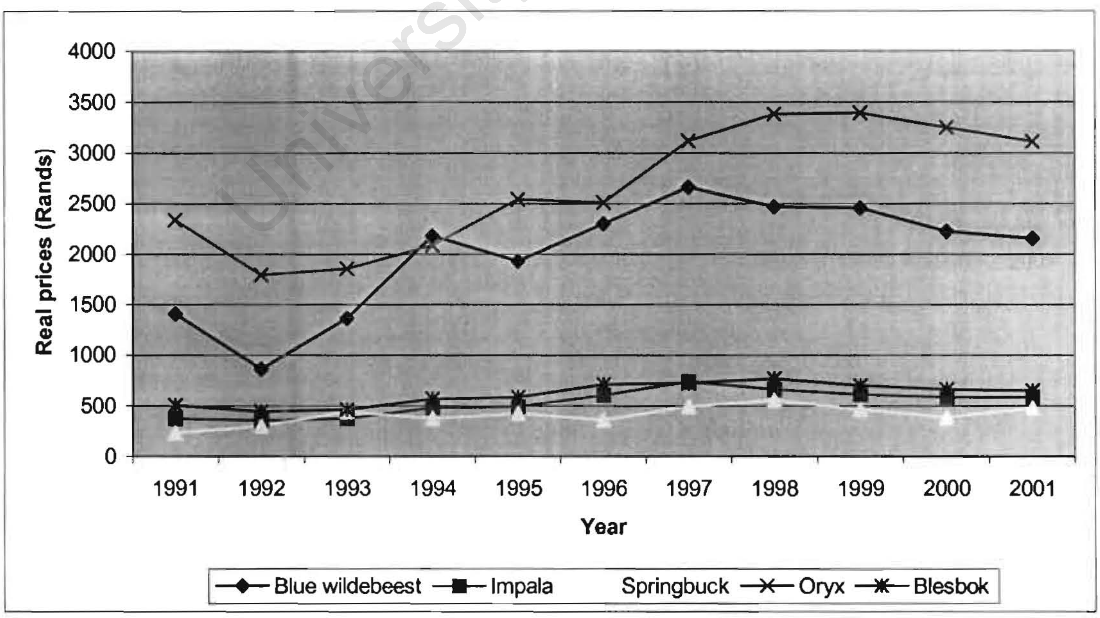

Figure 2-12: Trend in real prices of abundant game species over time

seen in relation to the cattle market, which is a prime example of a mature livestock industry (long periods of real price declines with shorter periods of real price increases). The wildlife industry needs therefore to enlarge current markets and to develop new markets if it wishes to remain in a growth phase with high returns.

2.4.5 The value of wildlife land

2.4.5.1 The increase in the demand for wildlife land

Reasons for the increase in the demand for wildlife ranching (and therefore wildlife land) are furnished in section 2.4.1.2.1. A more detailed investigation into some of the differences between wildlife ranching and cattle ranching is next provided.

Wildlife ranching versus cattle ranching

Barnes (1998) demonstrates, through financial and economic cost-benefit models, that cattle ranching has a higher financial internal rate of return than wildlife ranching (in Botswana) although the economic internal rate of return is significantly higher for wildlife ranching. Furthermore, the economic net present values (over a ten-year period) are positive for wildlife ranching, while these are negative for cattle ranching (refer to section 2.3.1, Table 2-3). This suggests that public transfers are necessary to make cattle ranching more attractive than wildlife ranching for investors. The reduction in subsidies in respect of cattle ranching, is indeed a major cause for the change over from cattle ranching to mixed farming and wildlife ranching.

Van der Waal and Dekker (2000) report that Behr and Groenewald (1990), through a national survey, found that two-thirds of respondents kept both cattle and wildlife on the same land. By 1998, Van der Waal and Dekker (2000) had demonstrated that 43.3% of respondents were practising a combination of cattle and wildlife production in Limpopo. There thus seems to be a continual changeover from mixed farming to "pure" wildlife ranching, as ranchers realise more returns from wildlife ranching. Wildlife ranching offers returns from a variety of markets, including: wildlife-viewing tourism, hunting, breeding and live sales, venison, biltong, taxidermy, natural resources and curios. Cattle farming, on the other hand, produces only milk, beef and hides. The cattle industry has reached maturity, while there are numerous opportunities for wildlife ranching to expand current markets and remain in a growth phase for quite some time still. One such market that has hardly been tapped in the global market, is that of venison production. It is submitted that as demand for lean meat and organic foods increases globally, venison will surely become a very popular protein option.

2.4.5.2 The market value of wildlife land and wildlife

The market value of wildlife land

According to ABSA (2002), if a wildlife rancher were solely interested in the *LSU equivalent value of game*, he would not be willing to pay more than R6 million for a 17 000ha lowveld property, since a higher price would make him uncompetitive compared with a wildlife rancher in the grassland region (who would pay R6 million for 6 000 ha of land). The *theoretical price* for lowveld land would therefore be limited to R353 per hectare (R6 million/17 000 hectares). However, in reality, lowveld land generally trades at anything between R3 000 and R6 000 per hectare, depending on *inter alia* the size and location. If the market value of land is R4 000 per hectare (see Table 2-20), the farmer needs to pay more than R68 million (see Table 2-21) for a large-sized lowveld ranch with the same number of LSUs as a large-sized grassland ranch (costing R6 million).

This large price difference between the *theoretical price* of R353 per hectare and the *actual price* of approximately R4 000 per hectare for lowveld land, represents the value that the market places on potential wildlife-viewing tourism income and increased aesthetic value. The owners of lowveld land can earn significantly more from wildlife-viewing tourism than owners in the grassland region - hence the considerably higher land prices. This, however, implies that it will, at least in terms of opportunity cost, be too expensive to concentrate on wildlife breeding in the lowveld region (ABSA, 2002).

As is evident in Table 2-20, actual average market prices for land are close to the theoretical prices in regions where wildlife-viewing tourism has not been developed or where agricultural alternatives to cattle farming are limited. In contrast, in regions like the lowveld, where land can be used for alternative purposes such as irrigated agriculture[62] or wildlife-viewing tourism, actual land prices deviate quite notably from their relative (theoretical) values.

Table 2-20: Actual and theoretical land prices for different wildlife ranch sizes in different ecological regions (Prices in Rands per hectare)

	Ranch size			
Ecological region	*Small-sized*	*Medium-sized*	*Large-sized*	*Theoretical price*
Grassland	1 200	1 050	1 000	*1 000*
Lowveld	4 500	4 200	4 000	*333*
Bushveld	1 200	1 150	1 100	*267*
Kalahari	220	180	160	*133*
Karoo	80	77	75	*73*

Source: ABSA (2002)

Note: Prices are inclusive of 14% value-added tax (VAT), but do not include game fencing, game stock or outbuildings on a ranch

Based on ABSA's (2002) assumed sizes of wildlife ranches and average market prices for land (given in Table 2-20), the total estimated land value per ranch ranges from 1.4 million for a small-sized ranch in the Karoo, to R69 million for a large-sized ranch in the lowveld (see Table 2-21 below).

[62] Particularly sugar cane, subtropical fruit and vegetable farming. Irrigated land in the Waterberg region, Limpopo, sells for anywhere between R10 000 and R20 000 (Maurits Blignaut, Regional Manager, Pam Golding, *pers comm* 2002).

Table 2-21: Market value of wildlife land for different wildlife ranch sizes and in different ecological regions (in Rand million)

Land market value	Ranch size		
Ecological region	*Small-sized*	*Medium-sized*	*Large-sized*
Grassland	1.4	4.2	5.7
Lowveld	16.2	50.4	68.6
Bushveld	5.4	17.3	23.6
Kalahari	1.9	5.4	6.9
Karoo	1.3	4.2	5.9

Source: ABSA (2002)

ABSA (2002) cautions that, unless the owner of a small or medium-sized wildlife ranch is fairly certain of the potential cash flow generated from wildlife-viewing tourism, he/she should be careful about what he/she is willing to pay for land. Furthermore, land typically represents more than half the total net asset value of a ranch in the lowveld and bushveld. Therefore, the internal rate of return (IRR) from wildlife ranching may be adversely affected if more is paid for the land than what it is worth in terms of its income-generating potential.

Recent developments in the Eastern Cape

Recently, the Eastern Cape has become the fastest changing area in South Africa and there has been much conversion from livestock to wildlife ranching. The Eastern Cape wildlife ranches have become very popular destinations for overseas hunters and wildlife-viewing tourists, due to their being easily accessible from the Garden Route, and their being in a malaria-free area. With the increase in demand for wildlife land, land suitable for wildlife ranching has been witnessing significantly increasing market prices. Land in the vicinity of Shamwari Game Reserve (a luxury private game reserve) was worth R475/ha in 1993, while, in 2002, it was worth R1 500/ha in current prices (B Fowlds, Farm Specialist, Pam Golding Properties, *pers comm* 2002). Shamwari's land value (including the wildlife and lodges) was worth R8 000/ha by 2002. Similarly, land bought for Kwandwe Game Reserve (the newest CC Africa reserve) cost R1 000/ha in 1998, while, by 2002, it was also valued at R8 000 (in current prices, including wildlife and lodges).

The market value of wildlife

Included below in Table 2-22 are estimates of the total average asset value of wildlife on various ranches that differ by size and ecological region. These were estimated by ABSA (2002) using average game auction prices for 2000 and large stock units (LSU) per species.

Table 2-22: Average asset values for wildlife stock on various ranches, and % of total asset value[1] of a wildlife ranch (over full investment period at constant 2000 prices)

Region	*Grassland*		*Lowveld*		*Bushveld*		*Kalahari*		*Karoo*	
Ranch size	*R ('000)*	*% of total*	*R ('000)*	*% of total*	*R ('000)*	*% of total*	*R ('000)*	*% of total*	*R ('000)*	*% of total*
Small ranch	731	28	1 382	7	3 067	33	1 172	30	851	29
Medium ranch	2 938	30	12 377	19	13 003	39	3 932	32	3 019	29
Large ranch	8 786	45	22 006	23	21 882	43	8 718	42	5 053	31

Source: ABSA (2002)

Note: [1] Total asset value includes land, fencing, game, and other assets.

As is evident, the asset value of wildlife on individual private wildlife ranches in South Africa ranges from R731 000 to R22 million. Multiplying these asset values by the number of wildlife ranches (5 061) in South Africa allows one to estimate a minimum and maximum[63] total asset value of wildlife on private land in South Africa. Assuming that all ranches are small, the total asset value of South Africa's wildlife on private land amounts to R3.7 billion. Since all ranches are not small, this is a minimum (and very conservative) estimate for the total asset value of wildlife on private land. It can also be expressed as an asset value of R357/ha[64]. Using this as an average across all wildlife areas[65] in South Africa, a conservative estimate for total asset value of wildlife across South Africa is R6 billion.

[63] A maximum estimate would be R111.3 billion.
[64] R3.7 billion divided by the number of hectares under exempted private wildlife ranches (10 364 154).
[65] This uses an estimate of 17 002 754 ha and excludes all 4 000 mixed game farms as mentioned in ABSA (2002).

2.4.6 The value of the wildlife market in South Africa

2.4.6.1 The value of the wildlife market at a national level

Table 2-23 provides a conservative estimate of nearly R2 billion (in constant 2000 prices) for annual gross income[66] generated from wildlife utilisation in South Africa. Gross income estimates of game sales income, local hunting and venison sales are taken from Eloff's (2000) estimates, as no other data on these were found. Foreign hunting and taxidermy income (in current prices) was supplied by PHASA[67] (refer to section 2.4.3.3 above for further details).

Table 2-23: Estimate of annual gross income generated from wildlife utilisation (in constant 2000 prices)

Non-consumptive incomes:	**623 668 611**
Gross wildlife-viewing income (minimum)	*443 668 611*
Gross game sales income	*180 000 000*
Game auction income	62 960 451
Consumptive incomes:	**1 357 196 262**
Total hunting income	*998 789 108*
Foreign hunting income	548 789 108
Local biltong & trophy hunting	450 000 000
Income from wildlife products & processes	*358 407 153*
Gross taxidermy income (minimum)	348 407 153
Gross taxidermy income from foreigners	348 407 153
Gross venison sales income	10 000 000
Total gross income from wildlife utilisation	**R1 980 864 872**

Source: Own calculations

For wildlife-viewing income, the KwaZulu-Natal Nature Conservation Services (KZN NCS) estimate for income per hectare from nature-based tourism, of R43/ha (in constant 2000

[66] Due to lack of data within the wildlife market, particularly at a time series level, most of the data used are 2000 or 2001 estimates. The annual gross income can thus be seen to be that for *circa* 2000.
[67] The sources from which these values were taken (Eloff and PHASA) do not, however, reveal the method by which they were obtained.

prices) is used. This includes incomes as shown in section 2.4.2.3.4. It does not include any subsidy income. That figure is then multiplied by the number of hectares estimated to offer wildlife viewing in South Africa. For this, public protected areas were added to 35.5%[68] of all "pure" wildlife ranching land, arriving at 10 317 875 hectares. This is a very conservative estimate per hectare, since it divides income from nature-based tourism by all the hectares under KZN NCS's management, while, in fact, not all the hectares are used for nature-based tourism. It can further be assumed that game reserves earn greater revenues per hectare from wildlife-viewing tourists than most protected areas do from all their nature-based tourists. Moreover, since KZN NCS is a public entity, their primary purpose is conservation and not to make profits – therefore, less income is probably currently derived than would have been if it were under private ownership. In comparison, CC Africa's average annual lodge income/ha is R870/ha (in constant 2000 prices). This shows that, under effective management and sound pricing policies, luxury game lodges can make 20 times more income per hectare than public entities, such as KZN NCS, from wildlife-viewing tourists.

The gross income estimate here can be compared with an estimate by Eloff (2000), which came to R823 million (in current 2000 prices) for the wildlife industry, and a more recent estimate of R713 million for wildlife ranches[69] (Eloff, 2002, in Bothma, 2002). The present author's estimate more than doubles the previous estimate of R823 million. The main causes for the increased gross income are improved estimates of hunting and taxidermy (through PHASA's data on the foreign exchange earnings of hunting and related activities), and more reliable data on wildlife-viewing tourism. Eloff's (2000) "ecotourism" estimate actually involved only a couple of phone calls to wildlife ranchers, and was not much more than a "guestimate" (T Eloff, Centre for Wildlife Economics, Potchefstroom, *pers comm*, 2002).

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[68] Van der Waal & Dekker (2000) estimated that only 35.5% of wildlife ranches in Limpopo offer ecotourism activities. If, instead, all hectares under public and private wildlife areas in South Africa (17 002 754ha) were to be multiplied by R43/ha, gross wildlife-viewing income would increase to R731 118 422. This said, it is clear that more research needs to be undertaken in South Africa to be able to determine reliable estimates for gross wildlife-viewing income.

[69] The gross income of R713 million for 5061 wildlife ranches was calculated by taking a gross income estimate for the wildlife industry of R843 million and then subtracting "peripheral" income of R130 million. "Peripheral" income is said to include: professional hunting fees (R50 million), wildlife capture (R40 million), Government fees (R30 million) and meat processing (R10 million) (Eloff, 2002, in Bothma, 2002). These incomes have been excluded from gross income calculations in Table 2-23, as it is not clear whether they are in fact additional incomes.

Table 2-24: Gross income/ha (constant 2000 prices) estimates for five respondents at the Pretoria game auction, 6 April 2002

Respondent no	Income/ha (constant 2000 prices)	Hunting income/ha	W-v tourism income/ha	Live sales income/ha
1	389	234	156	0
2	234	234	0	0
3	156	47	31	78
4	813	41	41	731
5	467	467	0	0
Average	**412**	**204**	**46**	**162**
Minimum	***156***	***41***	***31***	***78***
Maximum	***813***	***467***	***156***	***731***

Note: Prices deflated using a CPI index of 107 for the year 2001

Appendix 2.4 provides the data used to estimate the gross income/ha values that are provided in Table 2-24. A copy of the questionnaire used in obtaining the wildlife ranching data can be found in Appendix 2.3. Although the results above are not representative of all wildlife ranches in South Africa, they do show that wildlife ranching is indeed generating favourable incomes per hectare. These can be compared to an income of R117/ha that can be determined using the annual gross income figure in Table 2-23. The most income generated per hectare (R813/ha) is for a Northern Cape game farm that focuses on buffalo breeding. This shows that specialised farming can compare favourably (in respect of economic returns per hectare) to reserves catering for exclusive wildlife-viewing tourists.

According to Van der Waal & Dekker (2000) wildlife ranchers who live outside of Limpopo own around 50% of small ranches in that province. Furthermore, 30% of respondents mentioned they have the ranches for their own pleasure, and for their aesthetic value; and 27% of respondents saw owning a ranch as their contribution to conservation. This shows that many are not relying on an income from wildlife ranching. Thus, the average gross income earned per hectare could increase considerably if all ranches were to exist for the purpose of being financially profitable. The current situation can be contrasted with, for instance, cattle production and the gross income/ha derived from it, since production systems such as these are already striving to maximise revenues. Therefore, comparing gross income per hectare from wildlife utilisation to those from other land-uses, such as cattle production systems, may unfairly favour these alternative land-uses as more economically and financially viable industries.

Table 2-25 allows for a comparative analysis within the wildlife sector of different gross value added per hectare (VAD/ha) estimates. Eloff (2000), and Van der Waal and Dekker (2000), provide gross income/ha estimates of R58/ha and R61/ha, respectively, in current prices. Van der Waal and Dekker's (2000) income figure relies, firstly, on an estimate of total game sales in Limpopo, and, secondly, on assumed weightings placed on different utilisation categories. It does not rely on actual financial data of the ranches, as the respondents were reluctant to provide this.

Table 2-25: Comparative results of gross value added per hectare in the wildlife sector

Study & date	Scale &/or region	Size of reserve	VAD/ha (constant 2000)
Dekker (1998)	Mopani veld, Limpopo	? (semi-arid area)	R46 (current gross income)
Eloff (1998)	Wildlife industry, SA	+- 10 364 154 ha	R58 (current gross income)
Van der Waal & Dekker (2000)	Wildlife ranches, Limpopo	Median wildlife ranch: 1 150 ha	R61 (current 1998 gross income)
Eloff (2002, in Bothma, 2002)	Wildlife ranches, SA	10 364 154 ha	R69 (current gross income *circa* 2000)
Own study (2002) Data: KZN NCS (1999/2000)	KwaZulu-Natal (KZN)	1 576 566 ha	R136 (incl subsidies) R48 (excl subsidies) (average over 2 yrs)
Own study (2002) Data: KNP (1994)	Kruger National Park (KNP)	1 900 000 ha	R72 (incl subsidies)
Own study (2002) Data: CC Africa (2001/2002)	5 CC Africa lodges, SA	70 500 ha	R663 (average over 2 yrs)
Own study (2002)	***Private and public wildlife areas, South Africa***	***17 002 754 ha***	***R82*** (for *circa* 2000)

VAD/ha figures for KZN NCS, KNP and CC Africa are also provided. These were calculated using data and assumptions provided in section 2.4.2.3. Across South Africa's wildlife areas, gross VAD per hectare is estimated on average to be R82/ha[70]. As is shown in Table 2-26, the estimate for gross VAD from wildlife utilisation (for *circa* 2000) includes only direct use values captured[71] in the market, both consumptive and non-consumptive, and no indirect use values or non-use values. The total economic value (TEV) of the current wildlife sector would be significantly higher than the estimated direct use value of R1.4 billion. This is important to bear in mind, as it is the TEV that should be used to reflect accurately the costs and benefits of different land and wildlife use options (Barbier, 1992). The TEV of wildlife would also include *inter alia* all donations[72] made to game reserves and national parks, as well as the indirect use value of the ecological functions of wildlife species.

Table 2-26: Direct use value estimate of the wildlife market in South Africa (in 2000 rands)

Non-consumptive:	***436 568 028***
Wildlife-viewing tourism	310 568 028
Live game sales	126 000 000
Consumptive:	***950 037 383***
Hunting	***699 152 376***
Foreign hunters	384 152 376
Local hunters	315 000 000
Taxidermy	***243 885 007***
Venison sales	***7 000 000***
DIRECT USE VALUES:	***1 386 605 411***

Note: Assuming value added is 70% of each of the gross incomes in Table 2-23

[70] This is based on the total annual gross income provided in Table 2-23 and a wildlife area of 17 002 754 hectares. In addition, it is assumed that gross VAD is 70% of gross income.

[71] Ivory, for instance, is an example of a direct use value that is not captured in the market, due to international bans on the trade of ivory. Pre-ban ivory exports from Africa brought in revenue of US$35-45 million per annum (Barbier, 1992).

[72] Donations for KZN NCS were around R6.6 million (in 2000 prices) in the period 1999/2000. This translates to an extremely conservative estimate for the existence value of wildlife - R4.17/ha. Ideally, the existence value of wildlife could be determined by estimating the WTP for the conservation of wildlife and their habitats, by all the citizens of the world.

2.4.6.2 The value of the wildlife market at a provincial level

2.4.6.2.1 Wildlife ranching in Limpopo

According to Van der Waal and Dekker (2000), "the game-ranching industry has grown from a small beginning in the 1960's to an industry that is taking up a significant part of commercial agricultural land in Limpopo". These authors estimated the total number of wildlife ranches in Limpopo at 2 306[73] by August 1998, with a mean wildlife ranch size of 1 717 ha, and a median wildlife ranch size of 1 150 ha. It was further calculated that wildlife ranches in Limpopo covered a total of 3.6 million hectares by 1998 (an amazing 26% of Limpopo's total surface area). An approximation of the total gross income derived from ecotourism, was calculated using the annual median income of R21 000 per wildlife ranch and extrapolating to an estimated 819 wildlife ranches (35,5% of 2 306 were involved in ecotourism activities). Turnover from tourist accommodation on wildlife ranches therefore amounted to approximately R17 million per year in Limpopo.

In Van der Waal and Dekker (2000), live game sales in Limpopo were estimated at approximately R56 million during 1997. Using a weighting on different wildlife utilisations, annual turnovers were estimated at: R7 million from venison production, R82 million from local hunting, and R48 million from foreign hunting. Gross annual turnover therefore amounted to R221 million. This amounts to a gross income of R61 per hectare for wildlife ranches in Limpopo.

2.4.6.2.2 Estimated value of the wildlife market at a provincial level

No reliable gross income estimates from the total wildlife market by province can be made at this stage. It would not be correct to multiply average gross income per hectare estimates by

[73] Commercial farming units in Limpopo totaled 7 273 by 1996 (STATSSA, 2000). This means that 32% of all farms in Limpopo are wildlife ranches.

the number of hectares in a province, as the different ecological regions play a major role in determining the productivity of the land and the appropriate wildlife uses. If this method were applied, it would, for instance, greatly overestimate the income generated in the Northern Cape from wildlife utilisation (merely due to the wildlife ranches in the Northern Cape occupying so much wildlife land).

The only wildlife-market incomes that can currently be compared across the 9 provinces of South Africa (due to insufficient data) are those from game auctions and foreign hunting. Table 2-27 shows that in terms of percentage of total income generated from foreign hunters in 2000, the Eastern Cape earned the most (31%), Limpopo the second most (25%), and the Northern Cape the third most (13%).

Table 2-27: Estimate of foreign hunting income by province for 2000

	Foreign hunting income R (constant 2000 prices)	*% of income*
Limpopo	153 481 464	25
Northern Cape	82 602 490	13
Eastern Cape	192 279 796	31
North West	51 107 930	9
Mpumalanga	22 767 244	4
Free State	50 131 745	8
KwaZulu-Natal	49 182 843	8
Western Cape	5 705 776	1
Gauteng	4 509 024	1
South Africa	**587 204 346**	**100**

Data source: Estimates using %'s provided in Table 2-17 and PHASA data

As can be seen in Table 2-28 (below), of the total gross auction income for 2001, 31% was generated in Limpopo, 28% in KwaZulu-Natal and 11% each in the North West and the Free State. While the Eastern Cape is a major destination for foreign hunters, it does not presently generate significant earnings from game auctions (only 4% of total).

Table 2-28: Game auction income by province for 2001

Province	*Gross auction income R (current prices)*	*% of income*
Limpopo	27 415 994	32
Northern Cape	591 115	1
Eastern Cape	3 311 005	4
North West	9 885 379	11
Mpumalanga	7 231 400	8
Free State	9 593 560	11
KwaZulu-Natal	23 990 220	28
Western Cape	0	0
Gauteng	4 972 800	6
Total	**87 000 473**	**100**

Data source: Various issues of *SA Game & Hunt*

2.4.6.3 Brief comparative analysis with wildlife markets in two other countries

2.4.6.3.1 Botswana's wildlife sector

According to Barnes (1998), total gross output in the wildlife sector of Botswana is estimated as P124.5 million in 1986, and gross value added amounts to P53 million (both in constant 1991 prices). The gross output translates to around US$58[74] million or R160[75] million. See Appendix 2.5 for gross output and value added per wildlife use. It is difficult to compare the income generated from wildlife use in Botswana to that in South Africa, as South Africa has relatively reliable estimates only for around 2000. However, assuming annual growth rates between 5% and 25% for the wildlife market[76] (both these have been mentioned), leaves estimated gross income from the wildlife market in South Africa ranging from around R300 million (25%) to R1.3 billion (5%) in 1991.

[74] Assuming an exchange rate of P1.00 equal to US$0.47, which is given for the period of the study (Barnes, 1998).
[75] The 1991, end-year Rand/US$ exchange rate of R2.75 equal to US1.00 is used.
[76] As well as using a gross income from wildlife utilisation of R1 980 864 872.

2.4.6.3.2 Extent of wildlife market in the United States

Isaacs (2000) states that, in 1991, 3 160 000 people in the United States (US) spent $222 million on observing, photographing, and feeding wildlife (data from the United States Fish and Wildlife Service, 1993). Freese & Trauger (2000) report that expenditures associated with wildlife-related recreation totaled US $101 billion in the US in 1996, up from $63 billion in 1991. This estimate for gross income from wildlife utilisation is significantly larger than that in South Africa, even though South Africa has significantly more "large" wildlife species than does the US. In addition to these markets, millions of dollars are contributed by citizens of Canada and the US to nonprofit organisations to support a variety of use and non-use values of biodiversity.

2.5 CONCLUSION

The marketable value of many of South Africa's wildlife species has never been greater, while in the past wildlife has even been seen to have a "negative" value[77]. The growth in the wildlife ranching industry has contributed significantly to the recovery of some wildlife populations. Flack (2002, in Bothma, 2002) points out that there is more wildlife in South Africa now, than there has been for 100 years.

In this chapter, it has been shown how the gross value added from the wildlife market to the economy of South Africa is currently estimated at around R1.4 billion (in constant 2000 prices). While the current market serves to illustrate the great value attributed to wildlife, "great value provides economic incentive for exploitation and for 'milking' wildlife for not-so-wild settings" (Czech, 2000). It is thus of utmost importance that wildlife ranching and farming be regulated appropriately and sufficiently, if they are to continue to contribute both to the economy and to wildlife conservation.

[77] Flack (2002, in Bothma, 2002) mentions that, in the late 1950s, land in the Eastern Cape was advertised with one "advantage" being that it contained no wildlife.

CHAPTER 3

ESTIMATING DEMAND FOR WILDLIFE: CROSS-SECTIONAL MODELS

3.1 INTRODUCTION

To date, no economic analysis has focused on the linkages between demand for wildlife species and South African game auction prices. Therefore, the present investigation focuses on modelling the demand for certain wildlife species at auctions across South Africa. The econometric analysis aims *inter alia* to estimate price and income elasticities of demand for various wildlife species. These results will give some indication as to whether buyers of wildlife are in fact *price sensitive* even though the real prices of live wildlife have been increasing steadily over the past decade, at an average of 1.7% per annum (ABSA, 2002).

This chapter focuses on cross-sectional demand models, while the following three chapters focus on the theory and estimation of *panel data models*. Due to the fact that auction data were available only at annual intervals for the past eleven years, it has not been possible to estimate accurate time series models for wildlife species. Cross-sectional models are estimated for those species for which enough data points exist. Due to scope limitations, this chapter presents the results of the cross-sectional demand models using the *Oryx gazella*, commonly known as the oryx or gemsbok, as an example.

Figure 3-13, shows how the average nominal price of an oryx has increased by 113% over the last 11 years, as compared to a 77% increase in producer prices. Rare species, such as the roan antelope and the white rhino, have had nominal price increases of over 400% and 300%, respectively (Eloff, 2000).

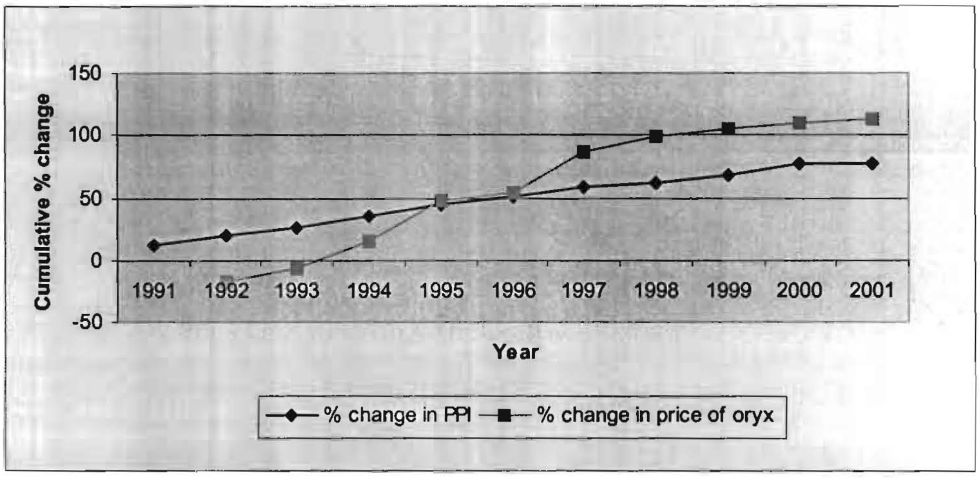

Figure 3-13: **Comparison of the curve of the cumulative % change in producer price index (PPI) from 1991-2001 with the curve of the cumulative % change in the annual average price of an oryx**

3.2 SOME APPLICATIONS OF TIME SERIES AND CROSS-SECTIONAL DEMAND MODELS

3.2.1 Demand models within agricultural economics

3.2.1.1 Demand within livestock production

Recent South African demand studies relating to livestock production include that by Nieuwoudt (1998) who projected demand for livestock products and protein feed for the years 2000, 2010 and 2020. These studies do not estimate income and price elasticities. Rather, as shown in Nieuwoudt (1998), they use past estimations of elasticities from Loubser (1990), Nieuwoudt (1990), Hancock *et al* (1984) and Badurally Adam (1997). These studies will not be further detailed here.

3.2.2 Demand models within wildlife economics

3.2.2.1 Demand for wildlife products

Barnes (1998) points out that the growing realisation that poaching and the ivory trade were endangering elephant existence, led to some econometric analysis on the determinants of international demand for ivory. Demand was apparently found to be generally price inelastic and income elastic in eastern consumer countries. Another study referred to by Barnes (1998) is that of 't Sas-Rolfes (1993, 1995) who discusses and examines the characteristics of demand in the Middle East for rhino horn. It was found that rhino horn demand is price-elastic, although possibly inelastic at higher prices, and has unitary income elasticity. It is mentioned that lack of suitable data was a constraint in there studies.

3.2.2.2 Demand for wildlife-viewing tourism

Day (1996) estimates a recreational demand model of wildlife-viewing visits to game reserves of Kwazulu-Natal. The study concentrates on the provision of overnight accommodation in Hluhluwe, Umfolozi, Mkuzi and Itala. The primary objective of the study was to provide an estimate of the value that the population of KwaZulu-Natal places on the provision of this recreational experience. This study is next further examined.

Day (1996) maintains that in recent years, McFadden's (1974) random utility models (RUMs) have become an increasingly popular approach for estimating the welfare benefits derived by visitors to recreational sites. His paper describes the application of a RUM known as a nested multinominal logit model (NMNL), which distinguishes the three dimensions of choice that characterise break-aways, ie the duration of stay, choice of recreational site and choice of accommodation type. Four costs are mentioned as important in making choices regarding such trips: the cost of travel to the recreational site, the cost of accommodation at the site, the cost of time while traveling, and the cost of time while on-site.

The mathematical details of the models are not presented here, since this model is not applied in this study. Results show average per-trip estimates of the consumer surplus enjoyed by visitors to range from around \$15 for one reserve, to almost \$50 for another. The model is also useful in predicting revenue changes. For instance, increasing entrance fees by R15 at Mkuzi would increase revenue to the KwaZulu-Natal Parks Board by 8%, and would increase by 21% if the same increase were instituted at Hluhluwe.

In conclusion, Day's model is used to answer economic questions, such as valuing the welfare derived from access to the reserves, as well as financial questions, such as predicting the changes in revenues that might result from changing pricing structures.

3.2.2.3 Demand for fishing and hunting licences

Boyle *et al* (2000) use data on sales of Vermont fishing and hunting licenses from 1979 through 1999 to predict license sales revenue for 2000/2001, by estimating demand equations for three different license types.

The demand equation used for fishing licenses is as follows (Boyle *et al*, 2000):

$$\#\text{licenses} = \alpha - \beta_1 \text{ price} + \beta_2 \text{ income} + \beta_3 \text{ population} - \beta_4 \text{ trend}$$

Where:
α, β_1, β_2, β_3 and β_4 are coefficients to be estimated
#licenses = number of licenses sold per year
price = real price of a license
income = real average annual income per capita in Vermont
population = Vermont population
trend = a count variable that equals 0 before 1991, and increases by 1 each year thereafter

The resident hunting and combination (hunting and fishing) license equations include two additional variables to measure the influence of changes in regulations on license sales. The first is a binary variable that is assigned the value of one for each of the years 1979 through 1982, during which time hunters could take antlerless deer. The second variable reflects trends in Vermont deer populations. The coefficient on this variable was expected to be positive; as deer populations increase and presumably hunters' success increases, sales would increase.

1999 predictions were found to deviate less than 11% from the actual sales. These models were estimated so that Vermont's Department of Fish and Wildlife could determine at what license prices they would be able to stabilize license revenue, for different declining sales levels.

3.3 THE THEORETICAL FRAMEWORK FOR WILDLIFE DEMAND

In line with normal microeconomic theory, the following theoretical *cross-sectional model* is proposed for the demand for wildlife species at auctions:

Q_w = function[P_w, P_s, P_c, R, I_r]

Where:

Q_w = Quantity demanded of a certain wildlife species at a specific auction within one year

P_w = Average auction price of a certain wildlife species at a specific auction

P_s = Average price of a substitute for a certain wildlife species within a region

P_c = Average price of a complement for a certain wildlife species within a region

R = Region in which auction was held

I_r = Regional annual average income of the buyers at the auctions.

For model results to be meaningful and reliable, the estimated model should comply with the following "*Full Ideal Principles*". This holds for all models, whether using cross-sectional, time series or panel data. In sum, the model selection process involves the following five criteria (Du Toit, 1999):

1. Consistency

The model should have logically possible signs and magnitudes for the parameters. It should also be consistent with long-run equilibria between the variables.

2. Significance

The estimated model should exhibit the economic and statistical significance of its parameters. It must be consistent with economic theory and be examined through a vast array of statistical indicators.

3. Data adequacy

Various indices of inadequacy should affirm that the chosen model provides an adequate representation of the data. Numerous problem-dependent statistical tests, whicht focus on explanatory power as well as testing the adequacy of the underlying assumptions, are applied.

4. Encompassing

The model should be encompassing of rival models, omitting no information that would have been useful in improving the preferred model. Several econometric methods have been suggested to examine whether any alternative model could be expected to deliver superior results.

5. Sensitivity

The model should not display too much sensitivity to the sample size, the variable menu, or to other equations in the system. Sensitivity tests can be applied to individual equations or to full systems.

A table summarising the methods to test model selection criteria can be found in Appendix 3.1.

3.4 EMPIRICAL CROSS-SECTIONAL WILDLIFE DEMAND FUNCTIONS

3.4.1 Theoretical model

Cross-sectional models are estimated for each of the years from 1999 to 2001. *Logarithmic-linear models* are estimated throughout the study, allowing elasticities of demand to be estimated directly.

The following cross-sectional demand models were estimated using ordinary least squares (OLS) estimation, providing unbiased and consistent parameter estimators (Pindyck & Rubinfeld, 1998):

$$\ln Q_{w\,i} = \alpha + \beta_1 \ln P_{w\,i} + \beta_2 D_{tranv\,i} + \beta_3 D_{kzn\,i} + \beta_4 D_{ecape\,i} + \varepsilon_i \qquad \text{equation 1}$$

Where:

Q_w = Quantity demanded of a certain wildlife species at a specific auction within one year

P_w = Average auction price of a certain wildlife species at a specific auction

$$D_{tranv} = \begin{cases} 1 & \text{if auction was held within the former Transvaal} \\ 0 & \text{otherwise} \end{cases}$$

$$D_{kzn} = \begin{cases} 1 & \text{if auction was held in Kwazulu-Natal} \\ 0 & \text{otherwise} \end{cases}$$

$$D_{ecape} = \begin{cases} 1 & \text{if auction was held in the Eastern Cape} \\ 0 & \text{otherwise} \end{cases}$$

$i = 1,2,3\ldots n$

the number of game auctions held in a year across South Africa

α	= a constant
$\beta_1, \beta_2, \beta_3$ and β_4	= the regression coefficients
ε_i	= error term.

3.4.2 Exposition of data

Data on the average prices and quantities of wildlife species sold at each auction, for the years 1999 to 2001, are available from monthly *Game and Hunt* magazines.

Due to current lack of data on buyers' income[78] by region, as well as price data on regional substitutes and complements, such as the price of beef and price of venison, models have been estimated using only average prices, quantities sold and regional data. The inclusion of regional agricultural gross domestic product (GDP) would not be sound, as regional gross agricultural income[79] is not necessarily correlated to the revenue generated by the wildlife

[78] In addition, no reliable data currently exist at a regional level on the gross income generated by the total wildlife market, nor on total hunting nor wildlife-viewing tourism income.

[79] For instance, gross farming income is the highest in the Western Cape (R7.5 billion in 1996) and the second lowest (R2 billion in 1996) in the Eastern Cape (STATSSA, 2000), while the Western Cape generates only 1% of total foreign hunting income and the Eastern Cape 31%. Gross foreign hunting income is currently the largest contributor to gross income from the wildlife market (see section 2.4.6.1).

ranching sector in the different regions. It is assumed that demand for certain wildlife species differs between regions of South Africa – hence the use of regional dummy variables. For instance, if a particular species is unsuitable to the climate and vegetation in the bushveld, there will be a low demand for that species at the auctions within that region.

3.4.3 Estimation results for the oryx

As already mentioned, the estimation results will be discussed using the *Oryx gazella*, commonly known as the oryx or gemsbok, as an example. The oryx is one of the three[80] most regularly sold species at game auctions, being sold at 28 auctions in 2000 and 29 auctions in both 1999 and 2001. This includes live and catalogue sales seen as separate auctions. The total number of reported game auctions held annually was 38, 49 and 39 for the years 1999, 2000 and 2001, respectively.

Estimating equation 1 for the oryx, for the years 1999 to 2001, provided statistically insignificant parameter estimates. In addition, for each possible combination of regional dummies within the model, the parameter estimates were found to be statistically insignificant. Non-logarithmic models and semi-log models were also estimated; nevertheless, regional dummies remained statistically insignificant.

The empirical cross-sectional model for the oryx is therefore:

ln(quantity demanded) = f [constant, ln(average price)]

Estimation results for the three cross-sectional models (one per year) are tabulated in Table 3-29. Table 3-30 gives the results for White's heteroscedasticity test.

[80] The impala, blue wildebeest and oryx were the three most commonly sold species at game auctions between 1999 and 2001. Models were estimated for each of these species.

Table 3-29: Estimation results for the cross-sectional models for the oryx

Year	No. of auctions	Variable	Coefficient	t-statistic	Prob.
1999	29	Constant	18.63464	2.924528	0.0069
		Price	-1.983765	-2.522816	0.0178
2000	28	Constant	11.20022	2.055795	0.05
		Price	-1.017464	-1.525927	0.1391
2001	29	Constant	14.89115	2.27971	0.0307
		Price	-1.525167	-1.907266	0.0672

Year				
1999	***R-squared***	0.190759	***F-statistic***	6.36
	Adjusted R-squared	0.160787	***Prob(F)***	0.0178
2000	***R-squared***	0.082195	***F-statistic***	4.79
	Adjusted R-squared	0.046895	***Prob(F)***	0.1391
2001	***R-squared***	0.118732	***F-statistic***	3.64
	Adjusted R-squared	0.086092	***Prob(F)***	0.0672

Table 3-30: Diagnostic test results for White's heteroscedasticity test

	Test for:	*Test:*	*F-statistic*	*Prob.*
1999	***Heteroscedasticity***	White	0.055343	0.946272
2000	***Heteroscedasticity***	White	0.31083	0.73563
2001	***Heteroscedasticity***	White	1.233192	0.307846

If the coefficients in Table 3-29 are substituted into the demand model, the following estimated demand equations for the oryx are obtained.

For 1999:

$$\ln(\text{quantity demanded}) = 18.63 - 1.98 \times \ln(\text{average price})$$

For 2000:

$$\ln(\text{quantity demanded}) = 11.20 - 1.02 \times \ln(\text{average price})$$

For 2001:

ln(quantity demanded) = 14.89 - 1.53 $\times$ ln(average price)

Economic evaluation of oryx demand models

For each of the three models, the coefficient of the explanatory variable (average price), which is transformed into logarithmic form, represents the price elasticity of demand. The qualitative influence of the price variable confirms the expected influence (as described in economic theory) - as the quantity demanded of a normal good increases, so the price will decrease. Thus, a negative relationship exists between price and quantity. Empirical interpretation of the quantitative influence of the explanatory variable, average price, on the dependent variable in the 1999 model, has the following result:

Price coefficient:

1) For 1999: a 1% increase in the average price results in a 1.98% decrease in the quantity of oryx demanded at an auction.

Statistical evaluation of oryx demand models

For the year 1999, the price variable and constant are statistically significant at a 5% level (see Table 3-29 for t-statistics and probability values). However, for 2000 and 2001, the price variable and constant are not significant at a 5% level. White's test for heteroscedasticity concludes that there is no heteroscedasticity present in the error terms, and therefore the ordinary least squares estimators of the 1999 model are asymptotically efficient (see Table 3-30). Results therefore show that, at a 5% level, only the 1999 model is statistically significant.

The R-squared is not an absolute measure of the goodness of fit, but it can be used as an indication thereof. The R-squared explains the percentage of the variation in the dependant variable that is explained by the variation of the independent variables. The R-squared coefficients for the three models are 0.19 (1999), 0.08 (2000) and 0.12 (2001). Such poor model fits imply that the average price of oryx is not the only determinant of the quantity demanded of oryx at auctions, and that the model could most likely be improved by the inclusion of further explanatory variables such as regional wildlife ranching income and regional substitutes and complements.

How price responsive are the buyers of oryx at game auctions?

Due to the fact that only one of the three models is statistically significant, one cannot conclusively conclude infer how price responsive buyers are for oryx at game auctions. However, one can conclude that buyers were very price responsive in 1999 due to the price elasticity of demand being -1.98.

3.5 CONCLUSION

With the aim of understanding the drivers of demand for wildlife species at game auctions across South Africa, cross-sectional models were developed in this chapter. The results are shown using the oryx as an example. Results indicate that buyers of oryx were price sensitive in 1999 with regard to quantity demanded at auctions, but results are inconclusive for the years 2000 and 2001. Current data limitations at the regional level are particularly prevalent, and thus no models showing a good fit were estimated. This holds not only for the oryx but also for impala and blue wildebeest, for which models were also estimated.

Due to the inability of the pure cross-sectional models (within the current data constraints) to determine the demand for various wildlife species, *panel data models* will next be investigated. Panel data models have a number of benefits over cross-sectional and time series models, and hopefully some of these will enable the estimation of significant demand models for wildlife species.

CHAPTER 4

APPLICATIONS AND THE THEORY OF PANEL DATA MODELS

4.1 INTRODUCTION

In this chapter, panel data models are introduced, and their potential benefits and limitations are mentioned. Various applications of panel data models are briefly reviewed. This includes applications both within environmental and/or ecological economics and outside of these fields. The theoretical backgrounds to specific panel data models that will be used in Chapter 5 are then provided.

4.1.1 Benefits and limitations of panel data models

As has been stated *panel data* are pooled observations on a cross-section over several time periods. This can also be referred to as "pooled time series". The cross-section would commonly be households, countries, industries or firms. In the present instance, it is the pooling of wildlife species traded at game auctions over the last 11 years. With reference to the work of Hsiao (1985,1986), Klevmarken (1989) and Solon (1989), Baltagi (2001) mentions several benefits of the use of panel data. These include the following:

1. Controlling for individual heterogeneity
2. Providing more informative data, more variability, less collinearity, more degrees of freedom and more efficiency
3. Being better suited to study the dynamics of adjustment
4. Being able to identify and measure effects that are simply not determinable in pure cross-section or time series data
5. Allowing the testing and construction of more complicated behavioural models than in pure cross-section or time series data

6. Usually data are gathered on micro units, eliminating the biases that can result from the aggregation of firms and individuals.

The limitations, as described by Baltagi (2001), include the following:

1. Design and data collection problems
2. Distortion of measurement errors
3. Selectivity problems, including self-selectivity, nonresponse, and attrition
4. Length of time series.

In the next section, various applications of panel data demand models will be reviewed. As will be revealed, there are a number of very useful outcomes from these models.

4.2 BRIEF REVIEW OF PANEL DATA DEMAND MODEL APPLICATIONS

4.2.1 Within environmental and/or ecological economics

Baltagi (2001) mentions two studies involving panel data econometrics within an environmental economics context. Baltagi and Chang (1994, in Baltagi, 2001:166-167) applied an unbalanced *one-way error*[81] model with *random effects* to assess home-owners' WTP for clean air in the Boston area. The dependent variable used was the median value of owner-occupied homes, and regressors included structural, neighbourhood and accessibility variables, and a pollution variable NOX. Mendelsohn *et al* (1992, in Baltagi, 2001:167-168) assessed the damage to housing value associated with proximity to a hazardous waste site. Actual sales data for 780 properties, in the harbour area surrounding New Bradford, Massachusetts, over the period 1969 to 1988 were used. *First differenced* and *fixed effects* estimation methods were used to control for specific individual housing characteristics. Their

[81] See section 4.3 for the explanation of panel data terminology.

results show a significant reduction in housing values, between 7 000 and 10 000 (1989 US$), as a result of houses' proximity to hazardous waste sites.

4.2.1.1 Demand for fishing licenses

Policy-makers may wish to have estimates of the economic value of a day of recreational fishing, for analysing *inter alia* the benefits of improvements in water quality, or for inclusion in a cost-benefit analysis (CBA) of a potential lakeside development. The two existing approaches, contingent valuation (CVM) and travel-cost method (TCM)[82] use detailed micro-data, of respondents' explicit or revealed estimates of WTP for specific sites to develop benefit estimates that are specific to particular bodies of water (Stavins, 1996). Both approaches have the disadvantage of requiring large quantities of geographically specific data, the collection of which can be time and cost intensive. In addition, CVM has generated considerable controversy within economics (Stavins, 1996).

Stavins (1996) therefore sought to develop and apply a new "conceptually distinct, revealed-preference" econometric method for estimating the economic benefits of an environmental amenity. The method fits within the *household production framework*, and is based upon the notion of estimating the derived demand for a privately traded option to utilise a freely-available public good. In particular, the demand (using panel data) for state fishing licenses were used to infer the benefits of recreational fishing. In contrast to the other approaches, this method uses highly aggregated data at the state level.

Data on fishing license sales and prices for 48 US states over a fifteen-year period, from 1975 to 1989, were combined with data on substitute prices and demographic variables. One concern was whether price and quantity were being simultaneously determined, or whether prices were *exogenously* set by states. A set of specifications therefore treated the licence

[82] Freeman (1993, in Stavins, 1996) maintains that other direct, revealed-preference methods that have been used for examining other environmental amenities – hedonic property and wage models – have not been applied to estimating the value of recreational fishing days.

price as endogenous, estimating the relationships with instrumental variable (IV) methods. Fixed effects were employed to control for constant differences among states in the quantity and quality of their recreational fishing resources.

Fixed effect models using OLS, were considered for five categories of permits, and estimated for three functional forms, namely: linear, multiplicative and semilog. IV models used multiplicative and semilog functional-form specifications. Below is a representation of the data used for the multiplicative specification of the demand for resident annual licenses (expressed as sales per capita) as provided in Stavins (1996).

Q_{it} = quantity of sales of resident annual licence in state i in year t;
N_{it} = population of state i in year t;
D_i = dummy variable which equals unity for state i, and zero otherwise;
P_{it} = price of resident annual licence in state i in year t;
P_{it}^{S1} = price of short-term, type 1 (1-3 day) resident licence in state i in year t;
D_{it}^{S1} = dummy variable which equals unity if a short-term resident licence is not offered in state I in year t, and otherwise equals zero;
F_{it} = area of fishable waters (acres) in state i in year t;
$_{it}$ = an independent, but not necessarily homoscedastic error term.

The results of this model can be found in Appendix 4.1.

The econometric results led to estimates of the benefits of a fishing licence, and subsequently to the expected benefits of a recreational fishing day (per state). These estimates were then compared with results from previous studies using both CVM and TCM (see Appendix 4.2). As Stavins (1996) avers, an added advantage of this method, for estimating the benefits of environmental amenities, is that it facilitates the development of a set of mutually consistent estimates that can be effectively compared with one another over time and space.

4.2.1.2 Estimating elasticities of residential energy demand from panel data

In another study, Garcia-Cerrutti (2000) used a dynamic linear model for each county i of the N counties:

$$y_i = x_i\beta_i + u_i$$

Here the y_i and u_i, (T × 1) vectors consist of T yearly observations on the dependent variable and the error term; the x_i (T × k) matrix includes observations on k – 1 independent explanatory variables including a 1-year lagged dependent variable; and the β_i (k × 1) vector contains the parameters to estimate.

The data in the study consist of annual residential electricity and natural gas per capita consumption and average real prices for 44 California counties from 1983 to 1997. A logarithmic-linear relationship relates demand for electricity (natural gas) to 1-year lagged electricity (natural gas) demand, personal income, price of electricity, price of natural gas, and heating and cooling degree-days. The specification treats residential electricity (natural gas) as a substitute or complement of residential natural gas (electricity). Negative and insignificant cross price effects are yielded. The author concludes that this implies natural gas as a complement for electricity, and vice versa.

Included in Appendix 4.3 are a number of panel data studies reviewed for the present study, although they all find their applications outside of environmental and ecological economics.

4.3 THEORY OF PANEL DATA MODELS

4.3.1 Brief overview of panel data models applied

In section 4.3.2, the theory of the panel data models applied in this study is briefly explored. The pooled model is first examined, followed by one-way error component models using fixed effects. Fixed effects models are used in this study, rather than random effects models, due to the type of data and the length of available data series. The seemingly unrelated regression (SUR) model is then also explored in a panel context (with a one-way error component).

4.3.2 One-way error models

4.3.2.1 Pooled models

The pooled model is the earliest panel specification. It pools cross-sectional data over time, allowing the *joint estimation of all coefficients*. A major benefit of pooled data, over pure time or cross-sectional data, is the increase in the size of the data set and thus also the degrees of freedom. Pooling potentially also lowers the standard errors on the coefficients.

The pooled model is appropriate in situations where the cross-sections are thought to behave similarly. For instance, it may be appropriate to pool data (over time) on developing countries if one is aiming to model demand for foreign aid. The rationale would be that the countries would have a similar demand structure.

A *pooled* logarithmic-linear demand model has the following *general* specification:

Logarithmic-linear model: $ln\_y_{it} = \alpha + \beta_1 ln\_x_{1,it} + \beta_2 ln\_x_{2,it} + \beta_3 ln\_x_{3,it} + \beta_4 ln\_x_{4,it} + \nu_{it}$

For i = 1,2, … N (number of cross-sections)
and t = 1, 2, … T (number of time periods)

Where:

y_{it}	= total quantity demanded
$x_{1,it}$	= gross income

$x_{2,it}$	= own price
$x_{3,it}$ and $x_{4,it}$	= the prices of substitutes and complements respectively
α and β	= scalar coefficients common across all cross-sections and time
ν_{it}	= assumed to be a "well-behaved" disturbance or error term ~ $IID(0,\sigma^2_\nu)$.

OLS is performed on the above model. The benefit of the logarithmic-linear demand model is that it estimates elasticities directly. The results of the pooled demand model will thus provide elasticities of demand for prices, income and cross-prices.

It is appropriate next to introduce one-way error component models.

One-way error component models

One-way error component models allow heterogeneity in the error term across one dimension, either across the cross-sections or the time series.

The error term is specified as follows (if heterogeneity is allowed across the cross-section):

$$u_{it} = \mu_i + \nu_{it}$$

The error becomes the sum of μ_i, the unobservable individual specific effect, which is time-invariant, and ν_{it}, a "well-behaved" disturbance, which varies across cross-sections and time.

4.3.2.2 Fixed effect: Least squares dummy variable (LSDV) model

Fixed effects model

In this approach, the fixed effects, μ_i, are assumed to be fixed parameters to be estimated, and the remainder disturbances stochastic with ν_{it} independent and identically distributed IID (0, σ^2_ν). In addition, the X_{it} are assumed independent of the error term, ν_{it} (Baltagi, 2001).

The fixed effects model is an appropriate specification if one is focusing on a specific set of, for instance, N firms, or N countries. However, the inferences made are then restricted to the behaviour of only those N firms or countries. In contrast, if a cross-section of, for instance, N firms, is randomly selected from the population of all firms, inferences can be made for the population. This example would use the random effects approach.

Cross-section or time series fixed effects models can be estimated with either the LSDV approach, also known as fixed effects (FE) least squares, or using "within" estimation.

Fixed effects LSDV model

The LSDV model can be written as follows[83]:

$$y = Z\delta + Z_\mu \mu + \nu$$

Where:

Z is NT × (K+1)

Z_μ is the matrix of *individual dummies* with a dimension of NT × N

K is the number of explanatory variables

N is the number of cross-sections

T is the number of time periods.

---

[83] See Baltagi (2001) for further explanation.

Ordinary least squares (OLS) is then performed on this model to obtain estimates for α, βs and μ_i's. The number of parameters to be estimated are (K + 1)×(N – 1). A fixed parameter (μ_i) is estimated for each cross-section, whereas, in the pooled model, only one intercept is estimated.

4.3.2.3 Fixed effect: The "within" model

Using the "within" estimation, one can still assume individual effects, although they would then no longer be directly estimated. In this approach, one demeans the data, "wiping out the individual effects" to estimate only βs. Although demeaning the data will not change the estimates for βs, the variance of the estimates will be smaller (McCoskey, 2002). The benefit of this approach, over the LSDV approach, is that it does not require the estimation of (N-1) dummies and therefore gains in degrees of freedom.

The OLS estimator is defined as follows:

$$\tilde{\beta} = (X`QX)^{-1}X`Qy$$

$$\mathrm{Var}\,\tilde{\beta} = \sigma_\nu^2(X`QX)^{-1}$$

The derivation will not be discussed here and it can be found in Baltagi (2001). The "within" model now becomes a fairly simple regression. The general model in section 4.3.2.1 now has the following regression specification:

$$(y_{it} - \bar{y}_{i\bullet}) = \beta_{1i}(x_{1,it} - \bar{x}_{1,i\bullet}) + \beta_{2i}(x_{2,it} - \bar{x}_{2,i\bullet}) + \beta_{3i}(x_{3,it} - \bar{x}_{3,i\bullet}) + \beta_{4i}(x_{4,it} - \bar{x}_{4,i\bullet}) + (\nu_{it} - \bar{\nu}_{i\bullet})$$

for i = 1,2, …N

t = 1,2, … T

Where:

y_{it}	=	total quantity demanded
$x_{1,it}$	=	gross income
$x_{2,it}$	=	own price
$x_{3,it}$ and $x_{4,it}$	=	the prices of substitutes and complements, respectively
βs	=	scalar coefficients
ν_{it}	=	disturbance or error term ~ IID($0,\sigma^2_{\nu}$)
$\bar{y}_{i\bullet}, \bar{x}_{1,i\bullet}, \bar{x}_{3,i\bullet}, \bar{x}_{4,i\bullet}, \bar{\nu}_{i\bullet}$	=	the averages of a cross-section over time.

There are K×N parameters now estimated by the model. As is evident in the above specification, the coefficients of the explanatory variables may be cross-section specific. However, if all the coefficients of explanatory variables are specified as cross-section specific, then the model is actually estimating individual models for each cross-section. It would therefore no longer be a panel model. An important disadvantage of the "within" model is that explanatory variables, which are themselves dummy variables, cannot be used, due to the demeaning of the data.

Individual effects can be solved under the assumption:

$$\sum_{i=1}^{N} \mu_i = 0$$

For a general demand model with two explanatory variables one solves for:

$$\bar{\alpha} = y_{..} - \bar{\beta}_{1i} x_{1,..} - \bar{\beta}_{2i} x_{2,..}$$

$$\bar{\mu}_{i.} = y_{i.} - \bar{\alpha} - \bar{\beta}_{1i} x_{1,i.} - \bar{\beta}_{2i} x_{2,i.}$$

4.3.2.4 Seemingly unrelated regression (SUR) model

The next model, the seemingly unrelated regression (SUR) model, is useful for estimating demand as a system of equations. If cross-equation correlation exists among the errors, the individual cross-section specific models will have inefficient estimators. In this case, Zellner's (1962) SUR approach is popular, since it captures the efficiency due to the correlation of the disturbances across equations (Baltagi, 2001).

The SUR approach improves on the efficiency of OLS, by writing the equation system as one combined equation, and then estimating the equation using generalized least squares (GLS) estimation (Pindyck & Rubinfeld, 1998). Baltagi (2001) mentions that Avery (1977) appears to have been the first to consider the SUR model with error component disturbances (in a panel context).

The one-way SUR model can be generalized by writing the system of M equations as follows (adapted from Pindyck & Rubinfeld, 1998):

$$y_j = X_j\beta_j + u_j \qquad j = 1,2,\ldots, M$$

Where:
- y_j = NT × 1 vector
- X_j = NT × K_j vector
- β_j = K_j × T vector
- u_j = NT × 1 vector
- N = number of observations per time period (T)

The generalised least squares (GLS) estimators have the following form:

$$\hat{\beta} = (X'\Omega^{-1}X)^{-1}(X'\Omega^{-1}Y) \qquad \text{with } \Omega \text{ needing to be estimated}$$

The derivations for this model are not dealt with here – these can be found *inter alia* in Pindyck and Rubinfeld (1998) and Baltagi (2001).

4.4. CONCLUSION

As shown in this chapter, there are numerous panel data models available for the estimation of one-way error demand models for the sale of wildlife at auctions across South Africa. These can also be extended to two-way error demand models, which would allow for heterogeneity across two dimensions (either a second cross-section component or across time). In the next chapter, various one-way error demand models are estimated and their results discussed.

CHAPTER 5

ESTIMATING DEMAND FOR WILDLIFE: ONE-WAY ERROR MODELS

5.1 INTRODUCTION

In this chapter, a number of one-way error models are estimated using panel data on the sale of wildlife species at game auctions across South Africa. Since the previous cross-sectional models did not reveal much, the purpose is still to attain a better understanding of the demand for various wildlife species sold at game auctions across South Africa, including ascertaining whether buyers are price responsive.

5.2 AN EXPOSITION OF THE DATA USED

Annual data on game auctions for the years 1991 to 2001 were obtained from Prof T Eloff, Centre of Wildlife Economics, at the University of Potchefstroom. This included graphical representations of data on average prices and total quantities sold of various wildlife species, as well as gross annual income from game auctions. Data on annual meat prices from cattle and sheep, for the same years, were obtained from the Agricultural Department at the University of Pretoria. Unfortunately, no annual data are available for these years on the gross income of the wildlife industry or wildlife ranches, nor from hunting or wildlife-viewing tourism[84] (or ecotourism). Similarly, no annual data are available on the average annual prices of trophies or game hunted for venison.

[84] Growth in the various wildlife uses (such as wildlife-viewing tourism) and market structure could influence the elasticities of demand for the various species. Unfortunately, there are currently no annual data on the gross income from these uses (except for game auctions) over the time period. Similarly, no annual data exist on the market structure for wildlife, for instance, on how many ranches provided trophy hunting from 1991 to 2001.

As mentioned in Chapter 2, ABSA (2002) estimates the growth of the wildlife industry over the last decade to have been at an average of 25% per annum. This annual growth rate is, however, not certain. For instance, the increase in the number of hectares under wildlife ranching, has grown on average at 6.71% per annum. It is therefore possible that the ABSA (2002) estimate is slightly inflated. In the light of the above discussion, it can be concluded that an average annual real growth rate of 18.83%, in gross game auction income, could actually be quite a feasible estimate for the growth rate per annum in the wildlife industry. It is for this reason that gross auction income can be used as a proxy for the gross income of the wildlife industry in South Africa. In turn, growth in the income of the buyers at game auctions would have some correlation with the growth in the wildlife industry, as the buyers of wildlife at auctions are themselves suppliers of wildlife ranching. Hence, due to lack of any other option, gross annual auction income is used in the models as a proxy for the gross income of buyers.

Where appropriate, data have been transformed, with 2000 as base year. All price data have been deflated using the production price index (PPI), consistent with supply and production behaviour over the period of estimation. Data have been transformed into the logarithmic form to enable log-linear models to be estimated. In logarithmic models, the estimated coefficients are the elasticities (partial multipliers), rather than coefficients of marginal effects. This is an advantage, since the purpose of this study includes determining elasticities of demand. Graphical representations of the data in their non-logarithmic and logarithmic form can be found in Appendix 5.1.

The focus in this chapter is only on the logarithmic-linear model, where total quantity demanded of a species is a function of the explanatory variables: own average price and gross game auction income (from all species). Many more models were run with different specifications but none of these passed all the economic and statistical significance tests. This included models with the constant prices of beef or mutton. These exact series thus did not have a significant effect on the quantity sold of wildlife at auctions. Perhaps if another variable such as gross income in the cattle ranching industry, a substitute industry for wildlife ranching, were to have been included, it may have had a negative relationship with the quantity of wildlife sold. This would have been due to the fact that, as income and profits in cattle ranching are decreasing, many farmers are changing over to wildlife ranching - there is thus an increase in the demand for wildlife. The addition of a trend variable into various models also proved insignificant. In all cases, the models failed to fulfil some of the most basic model selection criteria.

Lastly, all data series are assumed to be *stationary* for the purpose of this study, as the time period (T) in this chapter is 11 years, which is a relatively short time series. As T increases, the problem of non-stationarity worsens. In addition, the tests for stationarity with 11 years of data are so low in power that they may not be able to reveal much (S McCoskey, panel data expert, Department of Economics, US Naval Academy, *pers comm* 2002).

Section 5.3 and section 5.4 provide the model specifications and estimation results for the various models, while the discussion on the results will only be included in section 5.5.

5.3 DEMAND ESTIMATION FOR 14 WILDLIFE SPECIES

5.3.1 Individual models for the 14 wildlife species

The first type of model that is estimated, is the "individual model" of the species. Individual models are not panel models, as the species are not yet pooled. These models are estimated because the residual sum of squares (RSS) of the individual models are needed for the F-test for poolability in section 5.3.2.2.

For each of the 14 species, the model has the following specification:

$$ln\_y_t = \alpha + \beta_1 ln\_x_{1t} + \beta_2 ln\_x_{2t} + u_t$$

For $t = 1,2, \ldots 11$ (number of years, seen as independent time periods)

Where:

y_t	= total quantity of species sold in year t
x_{1t}	= gross income in year t
x_{2t}	= average price of species in year t
u_t	= disturbance or error term, independent and identically distributed IID $(0, \sigma^2_u)$
α and βs	= scalar coefficients.

5.3.1.1 Estimation results

The estimation results of the individual species models for the four species which pass all economic and statistical significance tests can be found in section 5.4.1. The remainder outputs of the individual models are in Appendix 5.2. These models either have the price coefficient, being statistically insignificant or with the incorrect sign (they have no economic significance). In section 5.4, a closer look is taken at the four species for which significant individual logarithmic models were found. The aim is to obtain additional information as to the price and income elasticities of demand for these wildlife species.

In the next section, a *pooled model* will be estimated for the 14 species with the aim of determining whether the various wildlife species do have common demand structures, and thus common slope and intercept coefficients. It is hypothesized that the buyers would have different elasticities of demand for the various wildlife species. These would depend on the species uses and scarcity.

5.3.2 Pooled models

Pooled models pool cross-sectional data over time, allowing the joint estimation of all coefficients. Here, data on the 14 species are pooled to estimate a single demand model. All species are therefore assumed to have similar demand structures.

The model estimated has the following specification:

$$\ln\_y_{it} = \alpha + \beta_1 \ln\_x_{1,it} + \beta_2 \ln\_x_{2,it} + \nu_{it}$$

For i = 1,2, … 14 (number of species in cross-section)
and t = 1, 2, … 11 (number of years in time series)

Where:

y_{it}	= quantity demanded of species i in year t
$x_{1,it}$	= gross income in year t
$x_{2,it}$	= average price of species i in year t
α and β's	= scalar coefficients common across all species and time
ν_{it}	= disturbance or error term assumed to be IID $(0, \sigma^2_\nu)$.

OLS is performed on the above model.

5.3.2.1 Estimation results

Model P.5.3.1[85], shown below in Table 5-31, illustrates how average price has a negative effect on the quantity of species demanded at game auctions and how gross income has a

[85] Models are numbered firstly according to their type, secondly by the section in which they are, and thirdly by their specification. Appendix 5.3 furnishes the different specifications used in Chapter 5. This includes a list of the models that use each specification.

positive effect on the quantity of species demanded. These results accord with economic demand theory. In addition, the elasticities of demand are both statistically significant, and the model has an adjusted R-squared of 75%, showing a good fit. The model directly estimates the price and income elasticities, which are common across all 14 species. The elasticities can be interpreted as follows:

Income elasticity: a 1% increase in the gross income results in a 0.88% increase in the quantity demanded of any of the 14 species.

Price elasticity: a 1% increase in the average price of a species results in a 0.69% decrease in the quantity demanded of that species.

The substituted equation for the buffalo is as follows (for all 13 other species the equation will appear the same and have the same coefficients):

$$\ln Q\_buffalo = -3.776221 - 0.691759 \times \ln P\_buffalo + 0.883797 \times \ln I\_buffalo$$

Below, the result of the F-test for poolability on model P.5.3.1 is included. This test is used to assess whether it is in fact appropriate to pool all the wildlife species finding common price and income coefficients.

5.3.2.2 Hypothesis Testing: F-test for poolability on model P.5.3.1

The result of the test[86] is that the null hypothesis of all coefficients being equal across the species ($\delta = \delta_i$ for all i), is rejected in favour of the alternative. This means that there are differences in the demand for the various wildlife species, and that the species should ideally not be pooled across all coefficients. This means that the species most likely have different price and income elasticities, although only one estimate for each elasticity was obtained with the pooled model 5.3.1.

[86] For the mathematical formulae of the test see Appendix 5.4.

Table 5-31: **Pooled logarithmic model 5.3.1**

SUMMARY OUTPUT: Pooled model 5.3.1 showing price and income elasticities

Regression Statistics	
Multiple R	0.86590603
R Square	0.74979325
Adjusted R Square	0.74647925
Standard Error	0.68276612
Observations	154

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	2	210.942152	105.4711	226.25045	3.72886E-46
Residual	151	70.39160586	0.46617		
Total	153	281.3337579			

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Intercept	-3.7762212	1.716690509	-2.19971	0.0293475
ln_Income	0.88379727	0.099741413	8.860886	2.046E-15
ln_Price	-0.6917591	0.034278155	-20.1808	1.013E-44

The next models estimated are the fixed effects models, which still allow for common elasticities. This is to determine whether the demand for the wildlife species depends on some *unobservable species-specific effects*.

5.3.3 Fixed effects: Least squares dummy variable (LSDV) model

In this section, the LSDV approach for the estimation of cross-section fixed effects will be used. See section 4.3.2.2 for more details regarding the LSDV approach.

The model estimated has the following specification:

$$\ln\_y_{it} = \alpha + \beta_1 \ln\_x_{1,it} + \beta_2 \ln\_x_{2,it} + u_{it}$$

For i = 1,2, … 14 (number of species in cross-section) and t = 1, 2, … 11 (number of years in time series)

Where:

y_{it}	= quantity demanded of species i in year t
$x_{1,it}$	= gross income in year t
$x_{2,it}$	= average price of species i in year t
α and β's	= scalar coefficients common across all species and time
u_{it}	= disturbance or error term.

Recall from section 4.3.2.1 that the error term is specified as follows:

$$u_{it} = \mu_i + \nu_{it}$$

Where the error u_{it} becomes:

the sum of μ_i, the unobservable individual specific effect, which is time-invariant; and ν_{it}, a "well-behaved" disturbance, which varies with species and time.

OLS is performed on the above model.

5.3.3.1 Estimation results

As can be seen in Table 5-32, the price coefficient is no longer statistically significant (t-statistic < 2 in absolute value) nor does it have economic significance (since it has a positive sign). The income coefficient and all the individual species effects are, however, statistically significant. Although the model does not provide a significant price elasticity, the model does most probably indicate that the inclusion of fixed effects for the species is an improvement on the pooled model (due to the statistically significant individual effects). This means that the demand models for the 14 species would have different intercept terms.

The estimate of the price elasticity (from the LSDV model) does not change when the "within" model is estimated (as would have been expected), and therefore the "within" model is not included here.

Table 5-32: LSDV model 5.3.1

SUMMARY OUTPUT: LSDV model 5.3.1

Regression Statistics	
Multiple R	0.950692013
R Square	0.903815304
Adjusted R Square	0.893360445
Standard Error	0.442817086
Observations	154

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	15	254.2737558	16.95158	86.44931168	2.40991E-62
Residual	138	27.06000208	0.196087		
Total	153	281.3337579			

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Intercept	-9.09233617	1.422221028	-6.393054	2.33474E-09
ln_Income	0.605624081	0.080679074	7.506582	6.79642E-12
Ln_Price	0.195625154	0.155400009	1.258849	0.210210205
Blue Wildebeest	4.036392268	0.577063412	6.994712	1.04803E-10
Eland	3.195860557	0.47112859	6.783415	3.16348E-10
Oryx	3.236606678	0.531757911	6.086617	1.07531E-08
Giraffe	1.500732649	0.33496653	4.480247	1.55123E-05
Kudu	3.590194125	0.578272257	6.208484	5.8852E-09
Nyala	2.849358923	0.487657235	5.842954	3.52016E-08
Impala	5.434659404	0.770844387	7.050268	7.81879E-11
Zebra	3.293548133	0.549608702	5.992533	1.70508E-08
Springbok	4.363091426	0.811053755	5.379534	3.11263E-07
Sable antelope	0.412575569	0.212247595	1.943841	0.053949441
Ostrich	2.870444312	0.663284584	4.327621	2.87451E-05
Waterbuck	2.78616933	0.482123932	5.778948	4.78557E-08
White rhino	0.550608871	0.193525927	2.845143	0.005115911

5.3.4 Discussion

The panel models estimated thus far have not provided much information as to how the elasticities of demand compare among the 14 species. The pooled model found common price and income elasticities for the species; however, through the F-test for poolability, it is shown that the species do have varying coefficients (and thus probably elasticities). This is as would have been expected, since the species range in scarcity from abundant to rare (see Appendix 2.2).

In the next section, the focus is on only four wildlife species. The four selected species range in scarcity from being relatively uncommon to rare. The very common species, such as the impala, have therefore been excluded from this selection.

5.4 DEMAND ESTIMATION FOR FOUR WILDLIFE SPECIES

The models here, as in section 5.3, are estimated using annual data for the years 1991 to 2001. However, as mentioned above the models in this section include data on only four wildlife species, namely the *eland, nyala, sable antelope* and *white rhino*. Again, individual models are first discussed, and then panel data models are used with the objective of finding reliable *estimates for elasticities of demand* for these species.

5.4.1 Individual models for the four wildlife species

The first models estimated are the individual species models. In addition to the model outputs, data plots of the actual, fitted and residual values obtained from each of the regressions are provided. Diagnostic tests are performed and the results are displayed.

Model specifications for the four individual species models are furnished in section 5.3.1 and will thus not be repeated here.

5.4.1.1 Eland

The estimated equation for the eland with the substituted coefficients, is as follows:

$$\ln Q\_eland = 1.56 - 0.64 \times \ln P\_eland + 0.58 \times \ln I\_eland$$

Table 5-33: Individual log-linear model for eland

Log-linear individual model: eland				
Dependent Variable: lnQ_eland Method: Least Squares Sample: 1991 2001				
	Coefficient	*Std. Error*	*t-Statistic*	*Prob.*
constant	1.560749	1.878746	0.83074	0.4302
lnP_eland	-0.641542	0.305378	-2.100815	0.0689
lnI_eland	0.575717	0.099906	5.762612	0.0004
R-squared	0.8293	Mean dependent var		6.186037
Adjusted R-squared	0.786625	S.D. dependent var		0.295202
S.E. of regression	0.136361	Akaike info criterion		-0.920018
Sum squared resid	0.148755	Schwarz criterion		-0.811501
Log likelihood	8.060097	Durbin-Watson stat		1.447902

A data plot of the actual, fitted and residual values obtained from the regression is given below (in Figure 5-14).

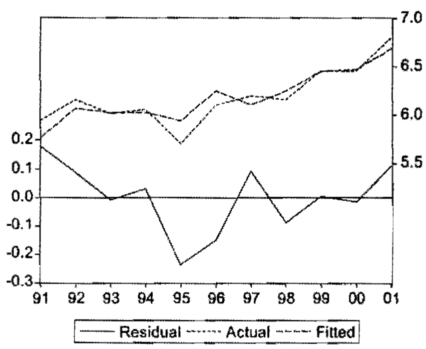

Figure 5-14: Actual, fitted and residual values of lnQ_eland

As can be seen in Table 5-33, the model exhibits statistical and economic significance (bearing in mind the short time series). The parameters have the expected signs and reasonable magnitudes. They are all statistically significant, passing the t-test. Moreover, the adjusted R-squared is 78.66%, which means that the model has a good fit.

Diagnostic tests are then performed to ensure that the model complies with the "Full Ideal Principles" mentioned in section 3.3.

Table 5-34: Diagnostic tests: Total quantity of eland sold (lnQ_eland)

Test for:	*Test*	*Test-statistic*	*Prob*
Normality	Jarque-Bera	0.604269	0.739239
Serial correlation	Breuch-Godfrey LM	0.074847	0.928738
Heteroscedasticity	ARCH	0.062275	0.809224
Heteroscedasticity	White	2.726811	0.131189
Parameter stability	Cusum	stable	
	Cusum square	stable	

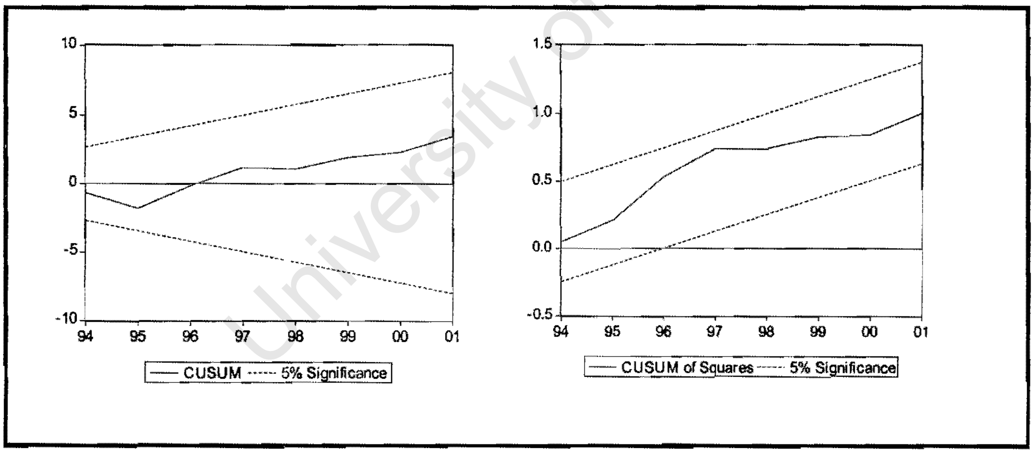

Figure 5-15: Parameter stability tests performed on the log-linear model of the eland

From these diagnostic tests it can be concluded that:

1. The error term is normally distributed
2. The error terms are not serially correlated - therefore the assumption can be made that all explanatory variables are included in the equation

3. There is no heteroscedasticity in the error terms - thus the ordinary least squares estimators are asymptotically efficient
4. The parameters are stable, as depicted in Figure 5-15

It is evident from the above results that the eland model has also passed all the statistical diagnostic tests. Thus, one can be confident that this model form is the most appropriate for the demand for eland.

5.4.1.2 Nyala

For the remaining species, including the nyala, all individual model results are not discussed, unless there is an important difference from the results of the eland requiring elucidation.

The estimated equation for the nyala with the substituted coefficients (as found in Table 5-35) is as follows:

$$\ln Q\_nyala = -14.33 - 0.85 \times \ln P\_nyala + 1.57 \times \ln I\_nyala$$

Table 5-35: Individual log-linear model for nyala

Log-linear individual models: nyala				
Dependent Variable: LnQ_nyala Method: Least Squares Sample: 1991 2001				
	Coefficient	*Std. Error*	*t-Statistic*	*Prob.*
constant	-14.33233	2.176656	-6.584565	0.0002
lnP_nyala	-0.847127	0.379549	-2.231928	0.0561
lnI_nyala	1.567883	0.217949	7.193797	0.0001

R-squared	0.91954	Mean dependent var	5.816897
Adjusted R-squared	0.899426	S.D. dependent var	0.730718
S.E. of regression	0.231736	Akaike info criterion	0.140565
Sum squared resid	0.429613	Schwarz criterion	0.249082
Log likelihood	2.226892	Durbin-Watson stat	2.614481

A data plot of the actual and fitted values obtained from the regression is given below, in Figure 5-16.

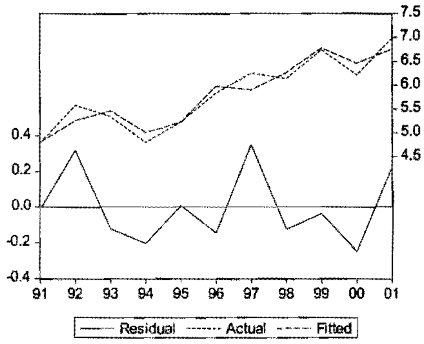

Figure 5-16: Actual, fitted and residual values of lnQ_nyala

Table 5-36: Diagnostic tests: Total quantity of nyala sold (lnQ_nyala)

Test for:	*Test*	*Test-statistic*	*Prob*
Normality	Jarque-Bera	1.182455	0.553647
Serial correlation	Breuch-Godfrey LM	0.861182	0.469032
Heteroscedasticity	ARCH	1.521433	0.252412
Heteroscedasticity	White	0.316059	0.857687
Parameter stability	Cusum	stable	
	Cusum square	stable	

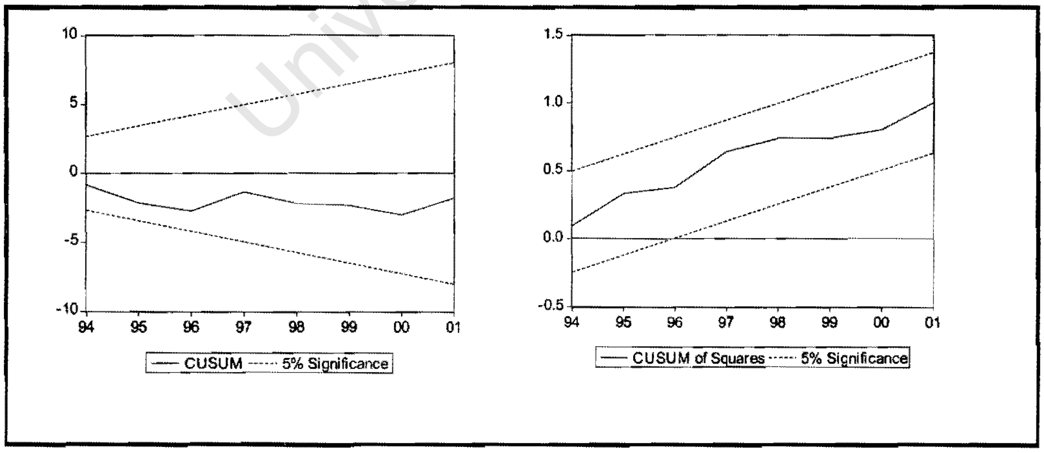

Figure 5-17: Parameter stability tests performed on the log-linear model of the nyala

The model exhibits both economic and statistical significance, passing all necessary tests. The adjusted R-squared coefficient (90%) shows a particularly good fit.

5.4.1.3 Sable antelope

Table 5-37: Individual log-linear model for sable antelope

Log-linear individual models: sable				
Dependent Variable: lnQ_sable Method: Least Squares Sample: 1991 2001				
	Coefficient	*Std. Error*	*t-Statistic*	*Prob.*
constant	-13.35035	4.004086	-3.334181	0.0103
lnP_sable	-0.935578	0.420055	-2.227273	0.0565
lnI_sable	1.559805	0.292812	5.326983	0.0007
R-squared	0.794848	Mean dependent var		3.824087
Adjusted R-squared	0.74356	S.D. dependent var		0.809358
S.E. of regression	0.409858	Akaike info criterion		1.280991
Sum squared resid	1.343871	Schwarz criterion		1.389508
Log likelihood	-4.045449	Durbin-Watson stat		2.958643

The estimated equation for the sable with the substituted coefficients, is as follows:

$$\ln Q\_sable = -13.35 - 0.94 \times \ln P\_sable + 1.56 \times \ln I\_sable$$

A data plot of the actual and fitted values obtained from the regression is given below, in Figure 5-18.

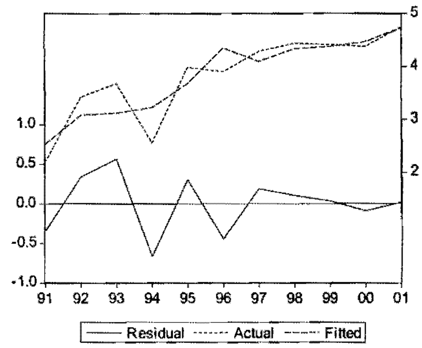

Figure 5-18: Actual, fitted and residual values of lnQ_sable

Table 5-38: Diagnostic tests: Total quantity of sable sold (lnQ_sable)

Test for:	*Test*	*Test-statistic*	*Prob*
Normality	Jarque-Bera	0.482046	0.785824
Serial correlation	Breuch-Godfrey LM	1.719842	0.256792
Heteroscedasticity	ARCH	2.152213	0.180543
Heteroscedasticity	White	3.029763	0.109174
Parameter stability	Cusum	stable	
	Cusum square	not stable around 1996	

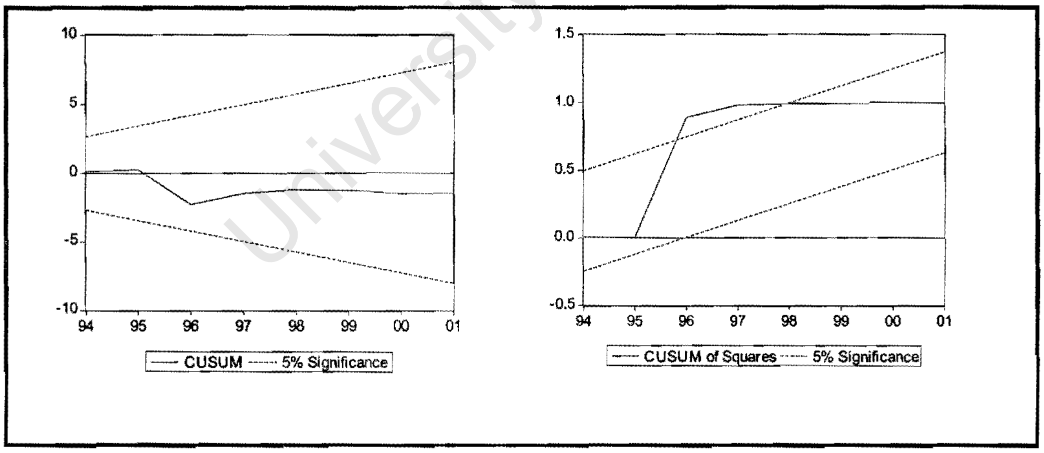

Figure 5-19: Parameter stability tests performed on the log-linear model of the sable

All economic and statistical tests are passed, except for the cusum square test, which shows some instability over the 1996 period. This could be attributed to the low number of observations (due to game auction data being only from 1991 onwards).

5.4.1.4 White rhino

Table 5-39: Individual log-linear model for white rhino

Log-linear individual models: white rhino				
Dependent Variable: lnQ_white rhino Method: Least Squares Sample: 1991 2001				
	Coefficient	*Std. Error*	*t-Statistic*	*Prob.*
constant	-7.577651	3.016334	-2.512206	0.0362
lnP_white rhino	-1.163218	0.405124	-2.871261	0.0208
lnI_white rhino	1.441647	0.35407	4.07164	0.0036
R-squared	0.701525	Mean dependent var		4.13756
Adjusted R-squared	0.626906	S.D. dependent var		0.507629
S.E. of regression	0.310067	Akaike info criterion		0.722944
Sum squared resid	0.769132	Schwarz criterion		0.831461
Log likelihood	-0.97619	Durbin-Watson stat		1.670471

The estimated equation for the white rhino with the substituted coefficients, is as follows:

lnQ_white rhino= - 7.58 - 1.16 × lnP_white rhino + 1.44 × lnI_white rhino

A data plot of the actual and fitted values obtained from the regression is given below, in Figure 5-20.

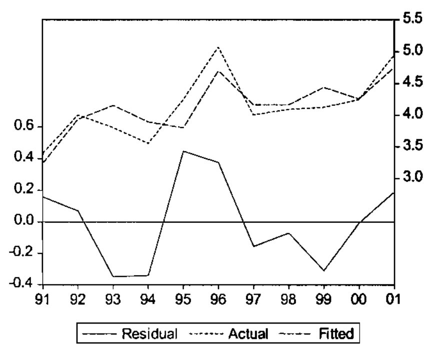

Figure 5-20: Actual, fitted and residual values of lnQ_white rhino

Table 5-40: **Diagnostic tests: Total quantity of white rhino sold (lnQ_white rhino)**

Test for:	*Test*	*Test-statistic*	*Prob*
Normality	Jarque-Bera	0.66129	0.71846
Serial correlation	Breuch-Godfrey LM	3.64694	0.091939
Heteroscedasticity	ARCH	1.137967	0.317214
Heteroscedasticity	White	0.903712	0.517103
Parameter stability	Cusum	stable	
	Cusum square	stable	

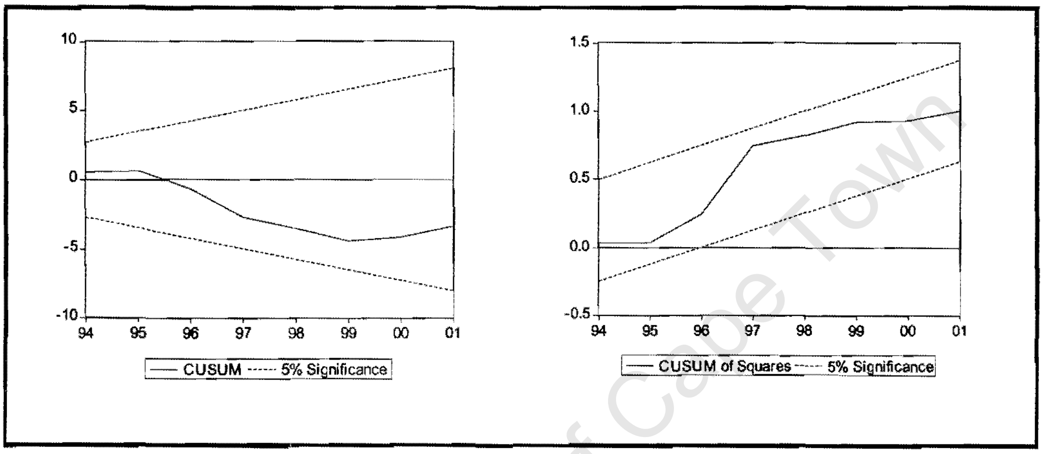

Figure 5-21: Parameter stability tests performed on the log-linear model of the white rhino

The model for the white rhino has an adjusted R-squared of 62.69%, which means that 62.69% of the variation in the dependent variable is explained by the variation in the explanatory variables. This demonstrates that there may be an omitted variable in the model. This is confirmed by the fact that there is some serial correlation in the error terms at a 5% significance level. In conclusion, this model could be improved by the addition of a significant explanatory variable, such as gross income from wildlife-viewing tourism or ecotourism in South Africa.

The individual models are next concluded with a table showing the price and income elasticities of demand for each of the four wildlife species:

Table 5-41: Price and income elasticities of demand from the four individual species models

Species	*Price elasticity*	*Income elasticity*
Eland	-0.641542	0.575717
Nyala	-0.847127	1.567883
Sable antelope	-0.935578	1.559805
White rhino	-1.163218	1.441647

Next, panel data models are utilised with the purpose of obtaining improved estimates for elasticities. The simplest panel specification, the pooled model, is first investigated.

5.4.2 Pooled model

As is stated in section 5.3.2, the pooled model involves pooling cross-sections over time, allowing the joint estimation of all coefficients. The pooled model below has the same specification as model P.5.3.1 discussed in section 5.3.2. The difference here is that only four species are pooled (rather than 14). The pooled model may indeed be a more appropriate model for the four species than it was for the 14 wildlife species. This is due to the four wildlife species being less varied in respect of scarcity and price range (see Appendix 2.2 for their 11-year price means).

The model has the following specification:

$$\ln\_y_{it} = \alpha + \beta_1 \ln\_x_{1,it} + \beta_2 \ln\_x_{2,it} + \nu_{it}$$

For i = 1,2, … 4 (number of species)
and t = 1, 2, … 11 (number of years)

Where:

y_{it} = quantity demanded of species i in year t

$x_{1,it}$	= gross income in year t
$x_{2,it}$	= average price of species i in year t
α and β's	= scalar coefficients common across all species and time
v_{it}	= disturbance or error term assumed to be IID (0, σ^2_v).

5.4.2.1 Estimation results

The results of model P.5.4.1 are reflected in Table 5-42 below. The estimated equation for the eland with the substituted coefficients, is as follows:

$$\ln Q\_eland = -8.29 - 0.70 \times \ln P\_eland + 1.16 \times \ln I\_eland$$

The above equation is exactly the same for the other three species, since all coefficients are common across the species. The estimated price and income elasticities for each species are significant. These are -0.70 and 1.16, respectively. The adjusted R-squared is 81.38%, which shows a good model fit.

Table 5-42: Pooled model 5.4.1

Pooled model 5.4.1				
Dependent Variable: lnQ Method: Pooled Least Squares Total panel (balanced) observations: 44				
Variable	Coefficient	Std. Error	t-Statistic	Prob.
C	-8.291141	2.425158	-3.418804	0.0014
lnP	-0.700839	0.05619	-12.47265	0
lnI	1.156964	0.142799	8.102046	0
R-squared	0.822473	Mean dependent var		4.991145
Adjusted R-squared	0.813813	S.D. dependent var		1.196325
S.E. of regression	0.516207	Sum squared resid		10.92525
Log likelihood	-31.78482	F-statistic		94.97529
Durbin-Watson stat	0.698995	Prob(F-statistic)		0

5.4.2.2 Hypothesis testing

Below, the assumption that all coefficients are identical across species, as well as the assumption that the error terms are homoscedastic (have a common variance), are tested. This is important as the pooled model assumes that all coefficients are common across species, when this may not be correct.

5.4.2.2.1 F-test for poolability on model P.5.4.1

The F-statistic[87] is greater than the critical value (CV), and thus the null hypothesis of common coefficients across the species, is not rejected in favour of the alternative. Therefore, according to the results, the different species appear to have similar demand structures and common elasticities of demand.

5.4.2.2.2 Testing for heteroscedasticity

The standard error component model assumes that the errors are homoscedastic, with the same variance across time and individual species. This, however, may be a restrictive assumption where cross-sectional units may be of varying size. Estimating heteroscedastic errors with the assumption of homoscedasticity will yield consistent but inefficient estimators. The standard errors will also be biased (McCoskey, 2002). One can therefore test[88] the assumption of homoscedasticity across the errors.

The Lagrange multiplier (LM) is found to be less than the 5% critical value, and therefore the null hypothesis of homoscedastic error terms across the individual species and time is not

[87] For the mathematical formulae of the test see Appendix 5.4.
[88] The approach described in McCoskey (2002) is taken from Greene (2000) and can be found in Appendix 5.5.

rejected in favour of the alternative. Therefore the OLS estimates for the pooled model 5.4.1 reflected in Table 5-42 are consistent and efficient.

In the next section, fixed effects models are estimated in an attempt to improve demand models for the four wildlife species by capturing unobserved species-specific effects.

5.4.3 "Within" models for the four species

5.4.3.1 Pooled "within" model 5.4.1

The first model estimated is that of the pooled "within" model. The pooled "within" model does not allow for the coefficients of the explanatory variables to vary across the species. It merely allows for cross-section fixed effects. These would need to be calculated, since they are not estimated directly with the "within" model as they would be with the LSDV model.

Model 5.4.1 has the following regression specification:

$$(y_{it} - \bar{y}_{i\bullet}) = \beta_1(x_{1,it} - \bar{x}_{1,i\bullet}) + \beta_2(x_{2,it} - \bar{x}_{2,i\bullet}) + (\nu_{it} - \bar{\nu}_{i\bullet})$$

for i = 1,2, …4

t = 1,2, …11

Where:

y_{it}	=	total quantity demanded
$x_{1,it}$	=	gross income
$x_{2,it}$	=	own price
βs	=	scalar coefficients common across species
ν_{it}	=	a "well-behaved" disturbance or error term
$\bar{y}_{i\bullet}, \bar{x}_{1,i\bullet}, \bar{x}_{2,i\bullet}, \bar{\nu}_{i\bullet}$	=	the averages of a cross-section over time.

5.4.3.1.1 Estimation results

The estimates for the pooled price and income elasticities exhibit economic and statistical significance, and are estimated at -0.76 and 1.18 (see Table 5-43). It is notable that, in model P.5.4.1, they were -0.70 and 1.16. The elasticities of demand have not changed much between the pooled model and the "within" model (which has allowed for fixed effects), and estimates thus seem to be rather stable.

Table 5-43: Pooled "within" model 5.4.1

Pooled "within" model 5.4.1				
Dependent Variable: lnQ				
Method: Pooled Least Squares				
Total panel (balanced) observations: 44				
Variable	Coefficient	Std. Error	t-Statistic	Prob.
lnP	-0.756269	0.216486	-3.493386	0.0012
lnI	1.183422	0.140571	8.418658	0
R-squared	0.924036	Mean dependent var		4.991145
Adjusted R-squared	0.914041	S.D. dependent var		1.196325
S.E. of regression	0.350747	Sum squared resid		4.674899
Log likelihood	-13.10969	F-statistic		92.44793
Durbin-Watson stat	1.614865	Prob(F-statistic)		0

5.4.3.2 "Within" model with a slope coefficient varying across the species

This model, unlike the pooled "within" model, allows both the intercept and a slope coefficient to be species-specific. Model 6.4.2 is estimated by allowing only the income coefficients to vary across the species.

Model 6.4.2 has the following regression specification:

$$(y_{it} - \bar{y}_{i\bullet}) = \beta_{1i}(x_{1,it} - \bar{x}_{1,i\bullet}) + \beta_2(x_{2,it} - \bar{x}_{2,i\bullet}) + (\nu_{it} - \bar{\nu}_{i\bullet})$$

for i = 1,2, …4

t = 1,2, …11

Where:

y_{it}	=	total quantity demanded
$x_{1,it}$	=	gross income
$x_{2,it}$	=	own price
βs	=	scalar coefficients with income not common across species
ν_{it}	=	a "well-behaved" disturbance or error term
$\bar{y}_{i\bullet}, \bar{x}_{1,i\bullet}, \bar{x}_{2,i\bullet}, \bar{\nu}_{i\bullet}$	=	the averages of a cross-section over time.

5.4.3.2.1 Estimation results

Table 5-44 below reflects the estimation output for this model. All price and income coefficient estimates have economic and statistical significance. This time, the income elasticities differ by species, unlike the income elasticity of 1.18 in model W.5.4.1, which was common across the four species. The pooled price elasticity changed from -0.76 (in model W.6.4.1) to -0.96. Fixed effects for each of the species are not directly estimated with the "within" model. However, they can be calculated, as shown in section 4.3.2.3.

Table 5-44: "Within" model 5.4.2 with income coefficients varying across species

"Within" model 5.4.2 with income coefficients varying across species				
Dependent Variable: lnQ Method: Pooled Least Squares Total panel (balanced) observations: 44				
Variable	Coefficient	Std. Error	t-Statistic	Prob.
lnP	-0.955363	0.192657	-4.958878	0
lnI_eland	0.644579	0.157841	4.083711	0.0002
lnI_sable	1.568758	0.175293	8.949352	0
lnI_nyala	1.618611	0.176861	9.151876	0
lnI_white rhino	1.281874	0.212267	6.038957	0
R-squared	0.95546	Mean dependent var		4.991145
Adjusted R-squared	0.945279	S.D. dependent var		1.196325
S.E. of regression	0.27985	Sum squared resid		2.741055
Log likelihood	-1.36467	F-statistic		93.85112
Durbin-Watson stat	2.587909	Prob(F-statistic)		0

Next, the validity of using fixed effects models is tested.

5.4.3.3 Testing the joint validity of fixed effects

The F-statistic[89] is established as being greater than the 5% CV - thus the null hypothesis of no cross-section heterogeneity, is rejected in favour of the alternative. The "within" models estimated above are therefore valid, and demand models for these species are correct in capturing the unobservable species fixed effects. It can be concluded that the fixed effects models are therefore an improvement on the pooled model.

Seemingly unrelated regression (SUR) models with error component disturbances (thus in a panel context) are next estimated. As mentioned in section 4.3.2.4, the SUR approach is

[89] See Appendix 5.6 for the mathematical formulae used in this hypothesis testing.

popular since it captures the efficiency due to the correlation of the disturbances across equations (Baltagi, 2001).

5.4.4 Seemingly unrelated regression (SUR) models

The one-way SUR model can be written as a system of equations, as follows (adapted from Pindyck & Rubinfeld, 1998):

$y_j = X_j\beta_j + u_j$ $\quad$ j = 1,2,…, 4 $\quad$ (number of species)

Where:

y_j = 11 × 1 vector

X_j = 11 × 2 vector

β_j = 2 × 11 vector

u_j = 11 × 1 vector

If:

T = 11 (the number of time periods)

N = 1 (the number of observations per time period).

This model is then estimated using *generalized least squares* (GLS).

5.4.4.1 Estimation results for the SUR model with a common price coefficient for all four species

The SUR model 5.4.2 is estimated with both the intercept and income parameters being species-specific, while the price parameter is common across the four wildlife species. Table 5-45 below provides the estimation results for model S.5.4.2.

The estimated demand equations with substituted coefficients, are as follows:

$$lnQ\_eland = 2.17 - 0.78 \times lnP\_eland + 0.61 \times lnI\_eland$$

$$lnQ\_sable = - 13.77 - 0.78 \times lnP\_sable + 1.49 \times lnI\_sable$$

$$lnQ\_nyala = - 14.34 - 0.78 \times lnP\_nyala + 1.54 \times lnI\_nyala$$

$$lnQ\_w\ rhino = - 6.83 - 0.78 \times lnP\_w\ rhino + 1.15 \times lnI\_w\ rhino$$

Table 5-45: SUR model 5.4.2 with common price coefficients across the species

SUR model 5.4.2 with common price coefficients across the species				
Dependent Variable: lnQ Method: Seemingly Unrelated Regression Total panel (balanced) observations: 44				
Variable	Coefficient	Std. Error	t-Statistic	Prob.
lnP	-0.777599	0.11294	-6.885059	0
Eland constant	2.17319	1.268349	1.7134	0.0955
Sable constant	-13.76506	3.296474	-4.175691	0.0002
Nyala constant	-14.33855	1.86547	-7.686297	0
W rhino constant	-6.82923	2.533432	-2.695644	0.0107
lnI_eland	0.605572	0.071657	8.450973	0
lnI_sable	1.488315	0.196721	7.56562	0
lnI_nyala	1.535296	0.120218	12.77097	0
lnI_w rhino	1.145232	0.169894	6.740876	0
Weighted Statistics				
Log likelihood	12.17107			
Unweighted Statistics				
R-squared	0.954376	Mean dependent var		4.991145
Adjusted R-squared	0.943948	S.D. dependent var		1.196325
S.E. of regression	0.283233	Sum squared resid		2.807731
Durbin-Watson stat	2.603023			

The model has an adjusted R-squared of 94.39%, which shows a very good model fit. The pooled price elasticity, -0.78, is lower than in model W.5.4.2, where it is -0.96. The income elasticities for each of the species have changed only slightly from model W.5.4.2, and for all practical purposes can be assumed to be the same. Both model W.5.4.2 and S.5.4.2 have standard regression errors of 0.28, Durbin-Watson (DW) statistics of 2.6, and show statistical and economic significance. It can therefore be argued that neither of these models is preferred over the other.

5.4.4.2 Estimation results for SUR model 5.4.3, which allows all coefficients to vary by species

The final model reported on is the SUR model 5.4.3, which allows all parameter coefficients to vary across the species. Estimation outputs are provided in Table 5-46 below.

The model has an adjusted R-squared of 93.89%, again showing a very good fit. All estimated elasticities, provided in Table 5-46, have the correct signs and magnitudes, and are statistically significant. Their interpretation can therefore be relied upon. As mentioned earlier in this section, if cross-equation correlation exists among the errors, the individual species models will have *inefficient* estimators. It is further shown in Darnell (1994) that if the cross-equation covariances are zero, then the GLS estimator is simply the OLS estimator from the regression of y_j on the regressors X_j. This is clearly not the case here, as the estimates found are different to the OLS estimates on the individual models. Therefore, it can be argued that the SUR estimates are more efficient than those of the individual models, since the SUR approach captures the efficiency due to the correlation of the disturbances across equations.

Table 5-46: SUR model 5.4.3 with no common coefficients

SUR model 5.4.3 with no common coefficients				
Dependent Variable: lnQ Method: Seemingly unrelated regression Total panel (balanced) observations: 44				
Variable	Coefficient	Std. Error	t-Statistic	Prob.
Eland constant	1.553117	1.38142	1.12429	0.2693
Sable constant	-14.14698	3.365713	-4.203263	0.0002
Nyala constant	-14.32856	1.856135	-7.719565	0
W rhino constant	-7.25193	2.556318	-2.836865	0.0078
lnI_eland	0.575345	0.075457	7.62485	0
lnI_sable	1.422477	0.229098	6.209046	0
lnI_nyala	1.58767	0.148331	10.70356	0
lnI_w rhino	1.312644	0.279799	4.691373	0
lnP_eland	-0.639847	0.187917	-3.404945	0.0018
lnP_sable	-0.63211	0.283068	-2.233066	0.0327
lnP_nyala	-0.889346	0.218311	-4.073765	0.0003
lnP_w rhino	-0.995392	0.312335	-3.18694	0.0032
Weighted Statistics				
Log likelihood	13.22874			
Unweighted Statistics				
R-squared	0.954564	Mean dependent var		4.991145
Adjusted R-squared	0.938945	S.D. dependent var		1.196325
S.E. of regression	0.295604	Sum squared resid		2.79621
Durbin-Watson stat	2.598801			

Table 5-47 summarises the price and income elasticities for the four species as estimated with model S.5.4.3.

Table 5-47: Price and income elasticities of demand for the four species from SUR model 5.4.3

Species	*Price elasticity*	*Income elasticity*
Eland	-0.639847	0.575345
Nyala	-0.889346	1.58767
Sable antelope	-0.63211	1.422477
White rhino	-0.995392	1.312644

5.5 DISCUSSION

Included below are Tables 5-48 and 5-49, which provide a summary of the four-species models (employed in section 5.4), and which allow for comparisons of the elasticity estimates. All the panel data models are classified into one of the following:

Pooled, where both price and income elasticities are common across the species;
Mixed, where income elasticity is species-specific and price elasticity is pooled; and
Species-specific, where both the price and income elasticities vary across the species.

As already mentioned, the inclusion of fixed effects is an improvement on the pooled model, P.5.4.1. Models W.5.4.2 and S.5.4.2 also improved on model W.5.4.1 due to a lower regression standard error and an improvement in the DW statistic. The three models, W.5.4.2, S.5.4.2 and S.5.4.3, appear to be "equally" statistically significant. However, as is evident from the estimated elasticities in Table 5-48, the individual species certainly play a unique role in the final magnitude of the elasticities, whereas the price and income elasticities where species are pooled fall somewhere in the middle of the range of the species-specific price and income elasticities. It is thus submitted that the preferred model is S.5.4.3, since it provides efficient species-specific elasticity estimates. As stated in section 5.4.4.2, the SUR model S.5.4.3 also improves on the individual models (due to the existence of cross-equation correlation). Accordingly, results are interpreted using the elasticity estimates from model S.5.4.3.

Table 5-48: Comparison of price elasticities by model type and specification

Model classification	Non-panel	Pooled		Mixed		Species-specific
Model type	**Individual model**	Pooled model	Pooled "Within" model	**"Within" model**	**SUR model**	**SUR model**
Model number	I.5.4	P.5.4.1	W.5.4.1	W.5.4.2	S.5.4.2	S.5.4.3
Price elasticity estimates (in absolute values)						
Eland	0.641542	0.700839	0.756269	0.955363	0.777599	0.639847
Nyala	0.847127	0.700839	0.756269	0.955363	0.777599	0.889346
Sable	0.935578	0.700839	0.756269	0.955363	0.777599	0.63211
W rhino	1.163218	0.700839	0.756269	0.955363	0.777599	0.995392

The SUR model's *price elasticity* (ε_P) estimate for each species is:

ε_P(eland) = - 0.64

ε_P(nyala) = - 0.89

ε_P(sable antelope) = -0.63

ε_P(white rhino) = -1.00

The price elasticities above show that demand for each of the wildlife species is generally inelastic to price changes (ε_P<1), in other words not very responsive to price changes. This could be due to a number of factors. One can speculate, for instance, that this could mean that there has been competitive buying for these species, and generally limited availability of these species at the auctions. This might then have forced the buyers, who may need to stock their ranches with a certain species, to buy even when reserve prices are a bit high. Buyers, under these circumstances, who need the wildlife for a specific wildlife use, might not be able to be price responsive. Indeed, buyer profile, ie the characteristics and circumstances pertaining to a particular buyer, will impact significantly upon price elasticity. Due to the fact that wildlife species are not generally substitutable, constant lack of substitutability[90] will most likely render buyers more price sensitive.

Demand for the sable and eland is the most unresponsive to price changes, while demand for the white rhino is the most price responsive. For the sable, this could mean that, due to its scarcity and limited availability at auctions, buyers need to bid competitively without being able to be too price sensitive. The eland is a very popular hunting species; therefore, although it is not a rare species, buyers might still be buying competitively. Demand for the white rhino has unitary elasticity, which means that for a 1% increase/decrease in price, there is a 1% decrease/increase in quantity demanded. The reason for this could be related to the fact that white rhino prices are very high in comparison to all other species, and thus a 1% change in their price is felt more strongly than a 1% change in the price of the less expensive eland, for instance.

The SUR model's *income elasticity* (ε_{I}) estimate for each species is:

ε_{I}(eland) = 0.58

ε_{I}(nyala) = 1.59

ε_{I}(sable antelope) = 1.42

ε_{I}(white rhino) = 1.31

The four species can be classified as "normal goods", in that, as the income available in the wildlife market increases, so the quantity demanded of species increases. The results show that the income elasticities for the nyala, sable antelope, and white rhino are greater than one,

[90] Substitutability is where a buyer can substitute the good sought with a relatively equivalent good.

while the income elasticity for the eland is much lower than one. If demand for a good increases by a greater proportion than income, it is said to be a "luxury good" (Varian, 1996). These three species, with income elasticities greater than one, can therefore be termed "luxury goods". One of the reasons why these would be classified as "luxury goods", while the eland is not, would be due to their *scarcity*. Either absolute scarcity, in the sense of their total population levels, or relative scarcity, ie their being uncommon in a particular region of the country. In addition, scarcity plays a major role in the price of the wildlife. Therefore, as income in the wildlife industry increases, so more money is circulating with which these more luxury species can be bought.

Table 5-49: Comparison of income elasticities by model type and specification

Model classification	**Non-panel**	Pooled		**Mixed**		**Species-specific**
Model type	**Individual model**	Pooled model	Pooled "Within" model	**"Within" model**	**SUR model**	**SUR model**
Model number	I.5.4	P.5.4.1	W.5.4.1	W.5.4.2	S.5.4.2	S.5.4.3
Income elasticity estimates (in absolute values)						
Eland	0.575717	1.156964	1.183422	0.644579	0.605572	0.575345
Nyala	1.567883	1.156964	1.183422	1.618611	1.535296	1.58767
Sable	1.559805	1.156964	1.183422	1.568758	1.488315	1.422477
W rhino	1.441647	1.156964	1.183422	1.281874	1.145232	1.312644

From Table 5-49 it is clear that demand for the various species certainly does have different sensitivities to income changes. For the more common hunting species, such as the eland, demand is not that sensitive to income changes. On the other side of the spectrum, however, buyers of the more rare and popular tourism-viewing species are very responsive to income changes. These species include the sable antelope, nyala and white rhino.

As mentioned earlier, an additional, and potentially very beneficial, use of panel data demand models (as shown in section 4.2.1.1) is that they allow for comparable estimates of the *economic value* of market or non-market environmental goods and services. This approach was applied by Stavins (1996) in an attempt to develop a new "conceptually distinct, revealed preference" econometric technique, in estimating benefits of a recreational fishing day for different states in the US.

5.5.1 Limitations of 11-year panel data models

Firstly, data series have been assumed stationary (data series ideally need to be tested for stationarity[91]). If the data series are in fact non-stationary, spurious results may be found. However, the problem of stationarity increases as the time dimension (T) increases. In this study, a relatively short time series is used. Assuming stationarity is therefore not as potentially problematic as it would be with a longer time series. In addition, the power of the stationarity tests is so low that they may not be able to reveal much.

The most serious limitation of these models is the present lack of data related to the wildlife industry in South Africa. Currently, no South African Government Department survey or census adequately allows for the determination of gross income and expenses from wildlife ranching activities. Furthermore, no research studies have collected time series data on *annual* gross estimates for income, at a national or regional level, from either the total wildlife industry, or hunting-related activities, or wildlife-viewing tourism. Due to this lack of data, gross annual game auction income is thus herein used as a proxy for the growth over time in gross annual income from all wildlife-use activities. In addition, annual auction data are available only from 1991 and therefore earlier trends in the growth of the industry cannot be captured. Considering the current lack of economic data, it is here suggested that Statistics

[91] Stationarity tests in a panel context are relatively new and many prior studies have not included tests for stationarity.

South Africa (STATSSA) take on the task of refining their agricultural surveys and censuses, thus enabling the collection of reliable statistics concerning wildlife ranches.

Finally, some of the 11-year panel data demand models may be able to be improved upon by the inclusion of various dummy variables relating to *inter alia* drought, sanctions and the change of Government in 1994. These factors all play a role in the changeover from cattle ranching to wildlife ranching (see section 2.4.1.2.1), and thus the increase in demand for wildlife species. The US$/Rand exchange rate could also more recently have had a positive effect on the continued growth of the wildlife ranching sector in South Africa.

5.6 CONCLUSION

Although the models exhibit economic and statistical significance, and efficient stable estimates are found, the models may not be of much value as forecasting models. This is due to potentially omitted variables, and the past (1991-2001) not necessarily according with the future. It is thus submitted that these models need first to be improved upon by the addition of new economic data on the wildlife industry, before valid and reliable forecasts can be made. Moreover, it is submitted that these models can, in further research, be extended into two-way panel data models. For instance, a second cross-sectional dimension, such as wildlife regions, could be included, with the aim of assessing any regional differences in the demand for wildlife species at auctions.

CHAPTER 6

CONCLUSION

6.1 SYNOPTIC OVERVIEW

Chapter 1 reveals that the wildlife ranching industry in South Africa has been increasing steadily and rapidly. Some estimates place it at an average of 25% per annum since the early 1990's. This growth has taken place at such a rate that research within the wildlife market has not kept up with it. The vast lack of reliable economic data can potentially allow wildlife to be mismanaged, and inadequate or inappropriate regulations and policies to be instated. Accordingly, this study firstly reviews all available economic information regarding the wildlife markets in South Africa, culminating in an estimate of the gross value of the whole wildlife market, and, secondly, attempts to determine the demand for wildlife species traded at game auctions in South Africa, using a variety of econometric modelling techniques.

Part 1 of the study (chapter 2) begins by examining some of the theoretical aspects of wildlife economics, including the importance of assessing the total economic value (TEV) of the wildlife market in South Africa. It is noted that the trend towards wildlife ranching has been due to a number of factors including *inter alia* the deregulation of the agricultural sector, agricultural subsidies effectively disappearing after 1994, and bush encroachment. The extent of wildlife areas under both private and public ownership in South Africa is estimated at around 14% (17 million ha) of South Africa's surface area. This highlights the need for wildlife ranching to be economically viable in the long term if much of South Africa's wildlife and many of its ecosystems are to be conserved.

The main wildlife markets in South Africa are then examined, focusing on their contribution to both the economy and conservation. Annual gross incomes (in constant 2000 prices) from these markets are estimated at R999 million (for gross hunting income), R444 million (for wildlife-viewing tourism) and R180 (for live game sales). Total gross income (in 2000 prices) from wildlife utilisation is indicated as being just under R2 billion, or R112 per hectare (averaged across all ecological regions). The chapter concludes with a gross value estimate of R1.4 billion for the wildlife market. This is, in fact, a conservative estimate for direct use values only – the total economic value (TEV) of the wildlife market would be significantly larger.

Part 2 of the study (chapters 3 to 5) focuses on modelling the demand for different wildlife species at game auctions across South Africa. Chapter 3 employs cross-sectional data to estimate the demand for wildlife species for each of the years from 1999 to 2001. Models are estimated for three species: impala, blue wildebeest and oryx, each being sold at a sufficient number of auctions each year. A statistically significant model is established only for the oryx (in 1999). A price elasticity of demand of -1.98 is revealed, demonstrating that buyers of oryx were very price sensitive in 1999. Due to the models lacking regional data and model results being poor, the investigation progresses to the use of panel data models, which are beneficial in that they control for individual heterogeneity, provide more data, more degrees of freedom and more efficiency.

Chapter 4 introduces panel data econometrics and includes varied applications of the panel data demand models reviewed for this study. A brief theoretical overview of the various model options that are suitable for determining demand for wildlife species, is then presented.

Chapter 5 includes the specification and estimation of one-way error demand models, using data on various wildlife species traded at auctions between 1991 and 2001, where heterogeneity is allowed across the various wildlife species. Due to data limitations, only the explanatory variables, annual average price and gross auction income (a proxy for gross income from the wildlife industry) are used. The first section estimates demand for 14 wildlife species, while the second section estimates demand for four wildlife species. Individual models, pooled models, LSDV and "within" models are estimated in both cases.

From the model results in chapters 3 and 5, a number of conclusions can be drawn. Results show that price has a negative effect, and gross income in the wildlife industry has a positive effect, on demand. This is accords with economic theory in respect of "normal goods". Furthermore, some of the more scarce or rare species, such as the white rhino and sable antelope, can be classified as "luxury goods". It can be argued that, as the availability of a wildlife species at auctions increases (often due to population levels increasing), buyers will become more price responsive, and the real prices will stabilise. Buyers of the oryx, which is currently classified as an abundant wildlife species, were found to be price responsive in 1999. Demand for those species that are not regularly sold (due to rarity or regional scarcity), such as the sable antelope and nyala, was found not to be price responsive. Availability at game auctions is, however, not the only factor affecting price responsiveness. In general, one can speculate that price responsiveness is a function of buyer profile, competitive buying, auction availability, rarity, price levels and lack of substitutability. Further research would, however, be needed more fully to assess these interrelationships.

6.2 CONCLUSION

The South African wildlife market is indubitably flourishing, and is contributing significantly to both the economy and conservation. Much land has been restored (from cattle ranching) to its "natural' state as wildlife ranching begins to show economic efficiency, and cattle farmers begin to face a variety of problems, including the lack of agricultural subsidies. Regrettably, for an industry of its magnitude, the wildlife market seriously lacks reliable economic data, particularly time series data.

Annual gross income from wildlife utilisation is currently estimated to be around R2 billion (in constant 2000 prices). No economic data presently exist to estimate annual figures even for the past decade. Furthermore, this lack of data is an obvious limitation to determining the demand for live wildlife species at game auctions. It is hoped that these economic demand models may in the near future be able to be improved with additional reliable statistics in respect of both cross-sectional and time series data. Only then will they really be useful for forecasting and policy evaluation.

It is necessary that an extensive analysis of the wildlife sector of South Africa be undertaken so as to reveal where the major regulation and enforcement needs are, and also where appropriate economic incentives can be applied. It is hoped that this work constitutes a small step in that direction.

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APPENDIX 2.1: Comparison between trophy and live-auction prices for game in South Africa

	Live game prices[1] (1999 current prices in Rands)	Trophy prices[2] (Eastern Cape prices in Rands at 10/99)	Trophy/live price
BlueWildebeest	2 250	5 544	2.46
Buffalo	76 700	12 320	0.16
Eland	4 182	9 856	2.36
Giraffe	10 875	12 320	1.13
Impala	563	1 848	3.28
Kudu	1 841	5 544	3.01
Nyala	3 250	9 240	2.84
Oryx	3 107	4 312	1.39
Ostrich	744	7 392	9.94
Sable Antelope	48 000	18 480	0.39
Springbuck	422	1 232	2.92
Waterbuck	3 611	9 240	2.56
White Rhino	127 000	154 000	1.21
Zebra	2 188	7 392	3.38

Data sources:

[1] Game auction data for 1999 from Prof T. Eloff

[2] Bezuidenhout (2000) US$ converted to Rands with R:US$ exchange rate of R6.16:1

APPENDIX 2.2: 11-year species means of quantity sold and averages prices at auctions

The table shows how the average prices of various wildlife species compare to one another. A general trend that is seen is that, as the scarcity of a wildlife species increases, so its price also increases. There are, however, many further determinants of the prices for live wildlife sold at auctions. These are mentioned in section 2.4.4.1.

11-year species means of quantity sold and averages prices at game auctions (in constant 2000 prices)

	Quantity	*Average Price*
Species means (over 11 years):		
Springbuck	1 071	411
Impala	3 206	536
Ostrich	293	1 098
Kudu	695	1 916
BlueWildebeest	1 159	2 001
Zebra	528	2 331
Oryx	486	2 677
Waterbuck	351	3 729
Nyala	425	3 783
Eland	507	4 055
Giraffe	126	10 860
Sable Antelope	58	36 873
Buffalo	50	71 109
White Rhino	71	94 355

APPENDIX 2.3: Game auction questionnaire

Wildlife Auction Questionnaire: Pretoria catalogue sale, 6 April 2002

This questionnaire is for the purpose of academic research for a master's thesis in environmental economics. The focus of the thesis is on showing how economic development and conservation can coexist and that they can mutually benefit one another. In light of the above, this questionnaire asks a few questions related to wildlife ranching and game auctions, with the aim of showing the extent to which the wildlife industry succeeds in showcasing how the natural environment can provide a platform for economic prosperity.

All information provided by you, will be held strictly confidential and data will only be used to estimate averages over all respondents. In addition, the questionnaires and results will not be shown or discussed with any other participant or organiser of this auction, thus not influencing the auction process.

Thank you very much for your willingness to participate in this research.

Hilary Anderson, UCT master's student
Tel: 012 841 3131, Fax: 012 841 2689
Email: handerson@csir.co.za

Wildlife farm/ranch details

1. Your position as related to the farm (owner, manager, etc): ____________
2. Province: ____________
3. Postal code: ____________
4. Approximate size of farm, in hectares: ____________
5. Year in which farm was purchased: ____________
6. Year of 1st exemption permit: ____________
7. Does the farm stock only game? Yes [ ] No [ ]
8. If no, what % of the farm area is used for: Domestic livestock [ ] Game [ ]
9. What % of the farm income is generated by: Domestic livestock [ ] Game [ ]
10. In what year did wildlife ranching begin on your farm? ____

11. *Why wildlife farming/ranching?*	% for each
Personal reasons (leisure & aesthetics, pass on to children)	
Contribution to conservation	
Financial and economic reasons	
TOTAL	100%

12. What are the main sources of income? 1) ____________
(From highest to lowest, including non-agricultural related incomes) 2) ____________
3) ____________

13. Number of full time employees on farm for: Domestic livestock [ ] Game [ ]

14. *What % of game-related income comes from the following:*	% for each
Wildlife-viewing tourism (including: activities and accommodation)	
All hunting (trophy and safari) related income	
Live sales	
TOTAL GAME-RELATED TURNOVER	100%

15. % of income from	*Local market*	*Foreign market*	TOTAL
Wildlife-viewing tourism			100%
Trophy hunting			100%
Other hunting			100%

16.1 Estimate the quantity (Q) of the following species on the farm;
16.2 What would you estimate last years average prices (P) were for the following species?

	Q	P		Q	P		Q	P
White rhino			Wild dog			Wildebeest		
Roan antelope			Giraffe			Waterbuck		
Black rhino			Tsessebe			Springbok		
Lion			Gemsbok			Zebra		
Elephant			Eland			Reedbuck		
Buffalo			Blesbok			Impala		
Sable antelope			Kudu			Hartebeest		
Hippopotamus			Nyala			Warthog		

Auction related questions

17. Main purpose of attending auction: *(tick appropriate blocks)*

To buy only	
To sell only	
To buy and sell	

18. List 3 species you most hope to buy and/or sell at this auction

BUY	SELL

Willingness to pay

	Yes	No	Rands
19.1 Would you consider buying a ***lion*** for the farm?			
19.2 Have you bought any lion for the farm in the last 3 years?			
19.3 What do you believe is a fair price for a lion?			
19.4 What is the most you would pay for a lion?			
20.1 Would you consider buying an ***impala*** for the farm?			
20.2 Have you bought any impala for the farm in the last 3 years?			
20.3 What do you believe is a fair price for an impala?			
20.4 What is the most you would pay for an impala?			
21.1 Would you consider buying a ***wildebeest*** for the farm?			
21.2 Have you bought any wildebeest for the farm in the last 3 years?			
21.3 What do you believe is a fair price for a wildebeest?			
21.4 What is the most you would pay for a wildebeest?			
22.1 Would you consider buying a ***white rhino*** for the farm?			
22.2 Have you bought any white rhino for the farm in the last 3 years?			
22.3 What do you believe is a fair price for a white rhino?			
22.4 What is the most you would pay for a white rhino?			
23.1 Would you consider buying a ***buffalo*** for the farm?			
23.2 Have you bought any buffalo for the farm in the last 3 years?			
23.3 What do you believe is a fair price for a buffalo?			
23.4 What is the most you would pay for a buffalo?			

24.1 Would you consider buying a ***kudu*** for the farm?
24.2 Have you bought any kudu for the farm in the last 3 years?
24.3 What do you believe is a fair price for a kudu?
24.4 What is the most you would pay for a kudu?

25.1 Would you consider buying an ***elephant*** for the farm?
25.2 Have you bought any elephant for the farm in the last 3 years?
25.3 What do you believe is a fair price for an elephant?
25.4 What is the most you would pay for an elephant?

26.1 Would you consider buying a ***roan antelope*** for the farm?
26.2 Have you bought any roan antelope for the farm in the last 3 years?
26.3 What do you believe is a fair price for a roan antelope?
26.4 What is the most you would pay for a roan antelope?

27.1 Would you consider buying a ***black rhino*** for the farm?
27.2 Have you bought any black rhino for the farm in the last 3 years?
27.3 What do you believe is a fair price for a black rhino?
27.4 What is the most you would pay for a black rhino?

28. For the following species, would you say recent auction prices have been correct/too high/too low?
(tick appropriate block)

	Recent auction prices	***Correct (+-)***	***Too high***	***Too low***
Impala	*R 650*			
Wildebeest	*R 2 300*			
Lion	*R 15 000*			
Black rhino	*R 550 000*			
White rhino	*R 170 000*			
Roan antelope	*R 105 000*			
Elephant	*R 10 000*			
Nyala	*R 7 100*			
Kudu	*R 2 300*			
Buffalo	*R 81 000*			

29. Estimate of annual gross farm income (cattle and wildlife ranching): ____________________

Personal details: For the purpose of possible further communication
NB: Details will not be revealed to anybody else. (Optional to fill in)
Name: ______________________________
Farm name: ______________________________
Contact no: ______________________________
Email address: ______________________

Thank you very much

APPENDIX 2.4: Summary of game auction questionnaire - wildlife ranches by province, size and income type

Respondent	Year game Ranching began	Province	Size (ha)	Main income source	Wildlife ranch income	% Hunting income	% Ecotourism income	% Live sales income
1	1995	Mpumalanga	1 200	Irrigated agriculture	500 000	60	40	0
2	1992	Kwazulu-Natal	2 000	Plantations	500 000	100	0	0
3	1987	Free State	1 500	Insurance brokerage	250 000	30	20	50
4	1991	Northern Cape	2 300	Game veterinarian	2 000 000	5	5	90
5	2000	Limpopo	300	Businessman	not disclosed	100	0	0
6	1992	Gauteng	1 200	Businessman	for leisure only			
7	2000	Limpopo	512	Retired	still setting up ranch			
8	1986	North West	200	Game	100 000	100	0	0

APPENDIX 2.5: Gross output and value added from the wildlife sector in Botswana

Wildlife utilisation	*Gross output*	*Gross value added*[1]
Wildlife viewing	67 856	20 176
Safari hunting	15 795	8 317
Licensed and subsistence hunting	21 403	17 336
Commercial wildlife production	2 632	1 365
Secondary trade and processing	13 194	5 274
Government (excluding revenue)	3 625	521
Totals	**124 506**	**52 989**

Data source: Adjusted from Barnes (1998)
Notes: 1986 data adjusted to 1991 prices, pula '000
At time of study US$0.47 = P1
[1] Gross value added is calculated by taking gross output less intermediate consumption

APPENDIX 3.1: Model selection criteria

Category	Property	Examples/Methods
Consistency	a) Inadmissibility	Signs, magnitude of parameters & predictions
	b) Poor operating characteristics	No flow equilibria Inconsistent with long-run parameters
Significance	a) Economic	Quantitative impact of unacceptable magnitude
	b) Statistical i) Nominal significance levels ii) Optimised significance levels	 F-tests, t-tests, Wald tests, etc Various information criteria
Indices of inadequacy	a) For the conditional mean	RESET, LM-test for serial correlation
	b) For the conditional variance	Various heteroscedasticity tests
	c) For normality	Jarque-Bera test
Encompassing	a) The mean	F-test
	b) The variance	t-test
Sensitivity	a) To sample size	Chow-test
	b) To variable menu	Extreme bound analysis
	c) To equations of model	Simulation

Source: Adrian Pagan's course Economics 517 Model Selection and Evaluation, taught at the University of Rochester in the Fall 1987 (De Wet, 1998)

APPENDIX 4.1: Demand for recreational fishing licenses: Multiplicative specification (fixed effects)

	Type of license				
	Resident annual	Resident Annual Combination	Resident Short-term (Type 1)	Nonresident Annual	Nonresident Short-term (Type 1)
Dependent variable	***log (sales per capita)***	***log (sales per capita)***	***log (sales per capita)***	***log (sales)***	***log (sales)***
log (price)	-0.1793 (-7.3)[1]	-0.2657 (-3.17)	-0.6552 (-5.01)	-0.7123 (-8.72)	-0.9655 (-9.76)
log (price of resident short-term license)	0.0497 (2.54)				
Dummy/no resident short-term license	0.1031 (2.98)				
log (price of resident annual license)			0.9671 (3.87)		
log (price of nonresident short-term license)				0.0876 (2.12)	
Dummy/no nonresident short-term license				0.1006 (1.54)	
log (price of nonresident annual license)					0.6176 (3.56)
log (acres of fishable waters)	0.0815 (2.01)	0.1801 (1.56)	0.1891 (0.87)	0.1821 (2.14)	0.2314 (1.52)
Number of observations	720	481	203	709	378
R-squared	0.121	0.255	0.154	0.273	0.21
Adjusted R-squared	0.055	0.154	0.056	0.223	0.141
Correlation coefficient squared	0.941	0.875	0.808	0.906	0.791

Source: Stavins (1996)

Note: [1] Numbers in parentheses are t-statistics

APPENDIX 4.2: Estimates of the value of a freshwater recreational fishing day

State	Previous valuation methods (non-panel methods)				Valuation from Stavins (1996) (1989 $)
	Estimation method	Types of fishing	Study	Valuation (1989 $)	
Alabama	TC[1]	Trout	King & Hof (1985)	17.69	14.51
Arizona	TC	All	Miller & Hay (1980)	52.67	31.41
Colorado	CV[2]	Cold water	Walsh et al (1980)	15.85	40.22
Connecticut	CV	All	Hyatt (1984)	8.68-12.29	4.9
Florida	TC, CV	All	Bell (1979)	13.1	15.37
		Warm water	Gibbs (1970)	34.55	
Georgia	TC	Warm water	Ziemer et al (1980)	19.91	15.7
Idaho	TC	All	Miller & Hay (1980)	40.63	35.72
		Cold/warm water	Loomis & Sorg (1986)	32.83/33.87	
Maine	TC	All	Miller & Hay (1980)	34.61	32.6
Minnesota	TC	All	Miller & Hay (1980)	43.64	34.27
Missouri	TC	Trout	Haas & Weithman (1982)	20.14	16.27
New Jersey	TC	All	Wiley & Leeworthy (1991)	16.18	10.23
New York	TC	All	Violette (1985)	35.72	22.84
		Streams	Mullen & Menz (1985)	28.88	
		Lakes		27.61	
Ohio	TC	Cold water	Dutta (1984)	6.29	13.56
		Perch/walleye	Hushak et al (1988)	3.30/4.03	
Oregon	TC	Salmon	Brown & Shalloof (1984)	26.26	29.03
		Steelhead		35.8	
Wisconsin	TC	All	Kealy & Bishop (1986)	37.16	27.37

Source: Stavins (1996)

Notes:

[1] TC refers to travel-cost method

[2] CV refers to contingent valuation method

APPENDIX 4.3: Non-environmental and/or ecological economic panel data studies

Estimating UK telephone access demand using pseudo-panel data

Gassner (1998) uses UK Family Expenditure Survey (FES) data, spanning the period fro m 1985 to 1996, where the FES is a continuous household survey that generates random samples of the populations every year. It is concluded that significant independent variables include: connection and rental charges (price variables), household income, and various socio-demographic variables.

Shortrun demand and supply elasticities in the West European market for secondary aluminium

Blomberg and Hellmer (2000) explore the supply-demand relationships in the market for secondary aluminium alloys. A standard microeconomic model is employed, where the determinants of supply and demand are identified, and an econometric model using data from Germany, France, Italy and the UK for the time period 1983-97, is estimated. Possible demand determinants are seen to be the price of secondary aluminium alloy (P_s), the price of magnesium (a potential substitute), auto production (AP), and gross domestic product (GDP). The final model specification to be estimated for demand is:

$$\ln QD_{it} = \alpha_0 + \beta_1 \ln P_{s,it} + \beta_2 \ln AP_{it} + u_{it}$$

Here i denotes country, and t time.

Blomberg and Hellmer (2000) use this model to assess the relative importance of the factors determining the supply and demand for the European secondary aluminium industry. Since price and quantity are determined simultaneously, ordinary least square (OLS) estimates would be biased and inconsistent. Hence, the authors apply the two-stage least square (2SLS) regression technique to account for the simultaneous equation bias in the estimation procedure.

The results demonstrate that both the supply and the derived demand for secondary aluminium are own-price inelastic. On the demand side, the level of auto-production (a primary user of secondary aluminium) revealed as having a substantial impact on the level of secondary aluminium alloy demand.

An analysis of housing expenditure using semiparametric models and panel data

Charlier, Melenberg, and Van Soest (2001) model expenditure on housing for owners and lessees by means of endogenous switching regression models for panel data. The share of housing in total expenditure is explained using a household specific effect, family characteristics, constant-quality price, and total expenditure, where the latter is allowed to be endogenous. Unbalanced panel data from the waves 1987-1989 of the Dutch Socio-Economic Panel (SEO) are used. The panel data models that the authors consider allow for household specific effects, which are either assumed to be independent of the explanatory variables (random effects), or allowed to be correlated with the explanatory variables (fixed effects).

Estimates for the random effects model are compared with estimates for the linear panel data model, in which selection only enters through the fixed effects, and with estimates allowing for fixed effects and a more general type of selectivity. Differences appear to be substantial. Moreover, the models lead to different conclusions regarding aggregate elasticities of housing expenditure in respect of total expenditure and prices.

Do health changes affect smoking?

Clark and Etile (2002) use seven waves of British Household Panel Survey (BHPS) data to examine the link between health developments while smoking (both one's own and those of other smokers in the same household) and future cigarette consumption. Becker *et al* (1994) in Clark and Etile (2002) demonstrate that the demand equation for an addictive good can be written as follows:

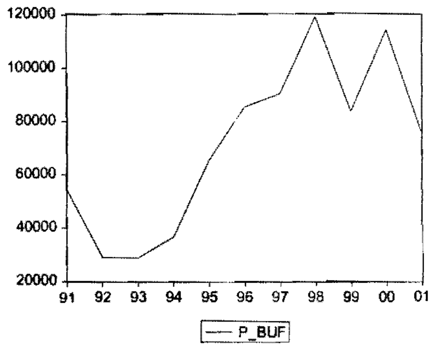
$$x_{it} = a_0 + a_1 p_t + a_2 x_{it-1} + a_3 x_{it+1}$$

Here p_t is the market price, and x_{it-1} and x_{it+1} are, respectively, past and future consumption. The demand equation used in this study, is a modified version of the equation above, and includes socio-demographic variables (Y_{it}), past health developments, an individual fixed effect (δ_i) and an error term, ε_{it}, which is likely heteroscedastic and autocorrelated. A standard way of dealing with the unobserved heterogeneity (in the δ_i term) is used - this is to estimate cigarette consumption in first differences.

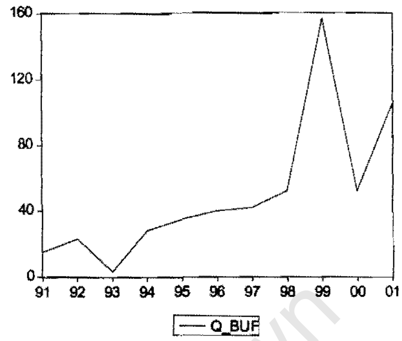
Since the error term is autocorrelated, the estimation of addiction models requires that future and past consumption, which are endogenous, be *instrumented*. This is typically carried out using prices as instruments. In particular, regional price variation, due to state cigarette taxes, is the key instrument for consumption in existing literature. The authors employ an alternative approach whereby lagged consumption instruments current consumption. The study concludes that those whose health worsens when smoking, smoke less in the future, and are more likely to quit.

APPENDIX 5.1: Graphical representation of the data

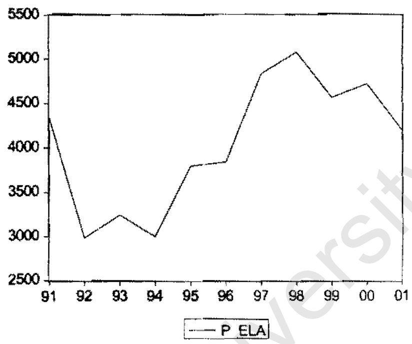
Real average price of buffalo



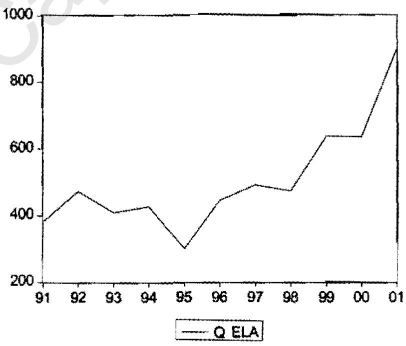
Quantity sold of buffalo



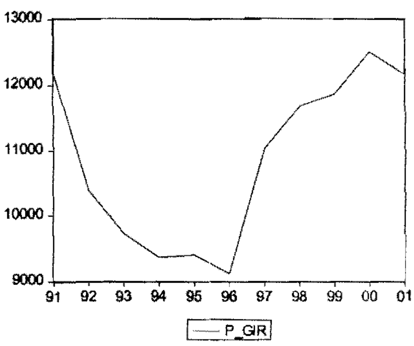
Real average price of eland



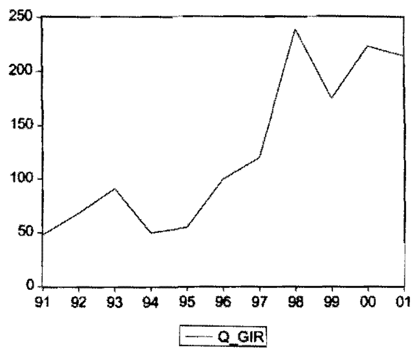
Quantity sold of eland



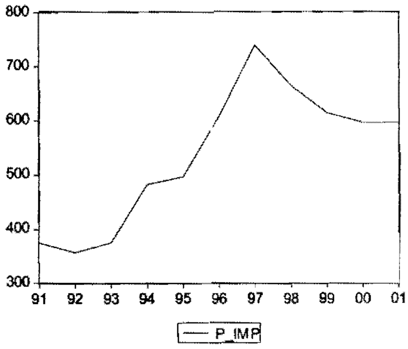
Real average price of giraffe



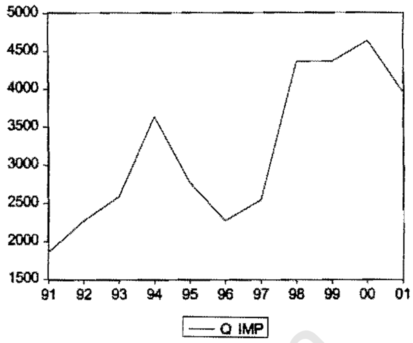
Quantity sold of giraffe



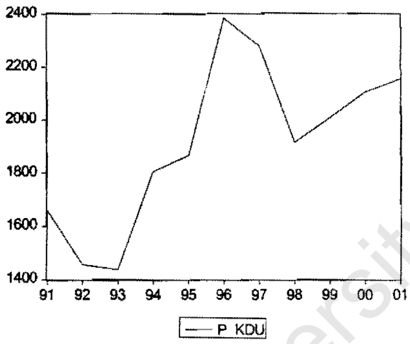
Real average price of impala



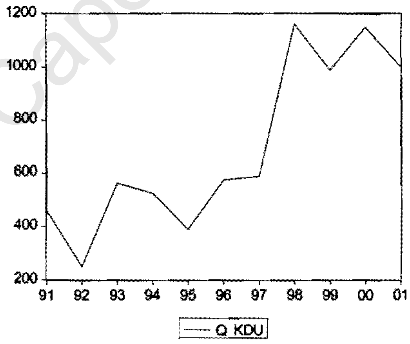
Quantity sold of impala



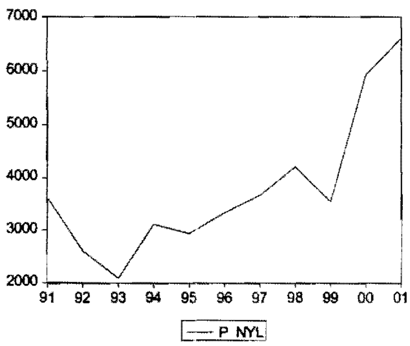
Real average price of kudu



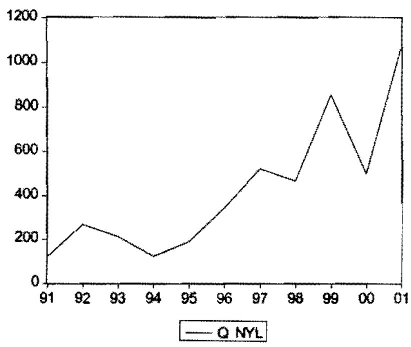
Quantity sold of kudu



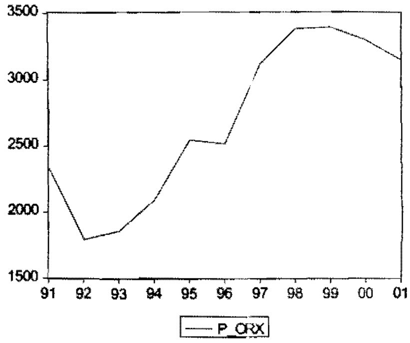
Real average price of nyala



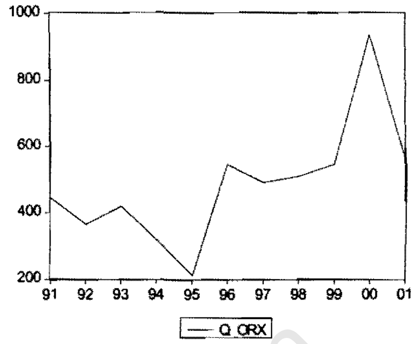
Quantity sold of nyala



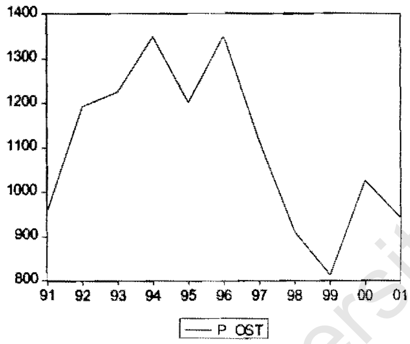
Real average price of oryx



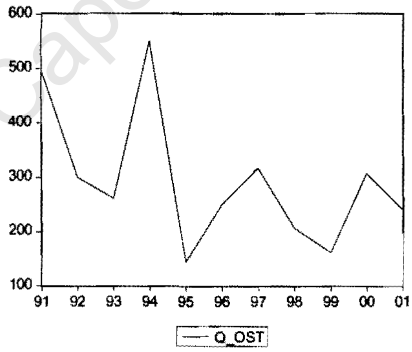
Quantity sold of oryx



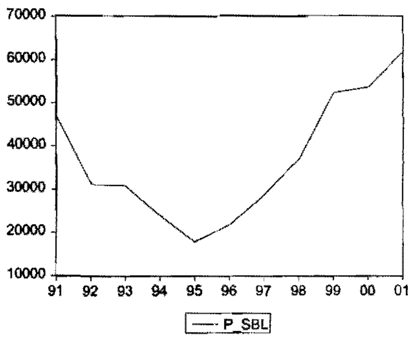
Real average price of ostrich



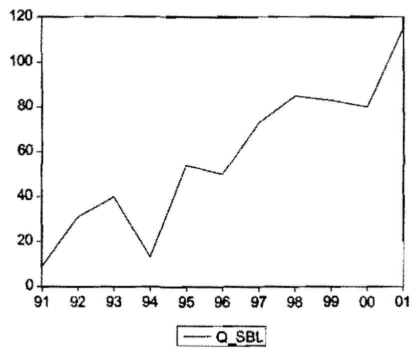
Quantity sold of ostrich



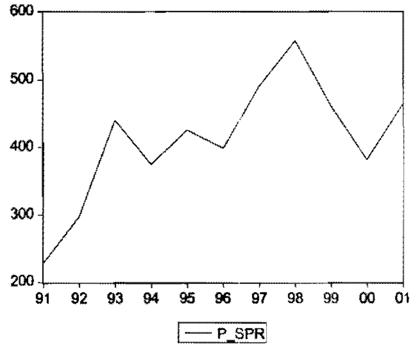
Real average price of sable



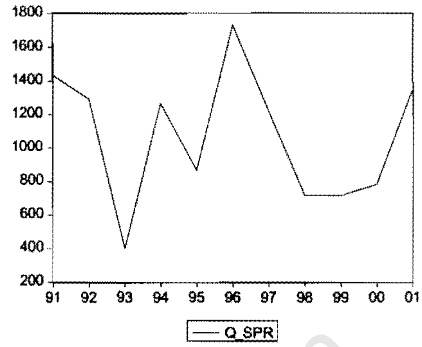
Quantity sold of sable



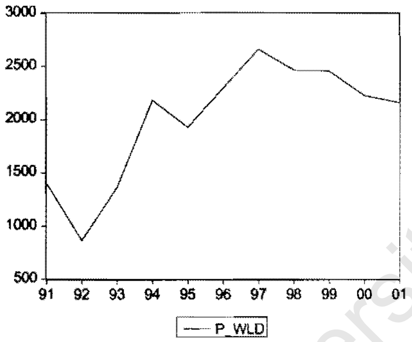
Real average price of springbok



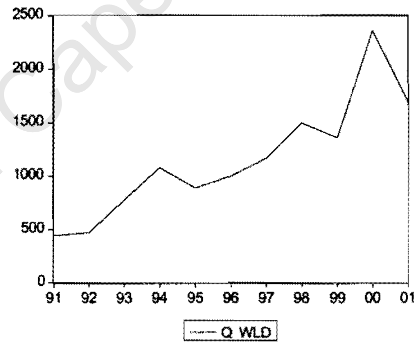
Quantity sold of springbok



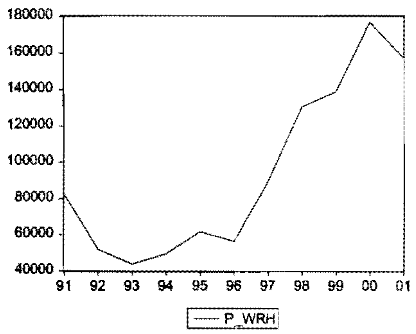
Real average price of wildebeest



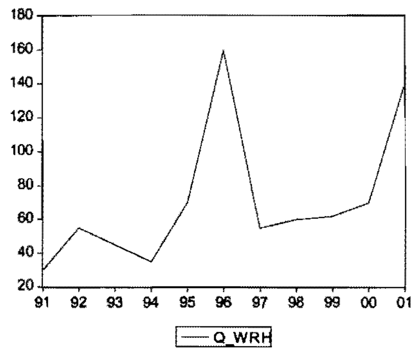
Quantity sold of wildebeest



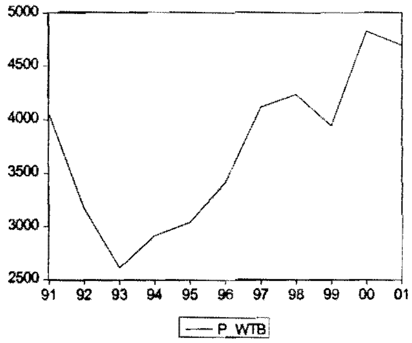
Real average price of white rhino



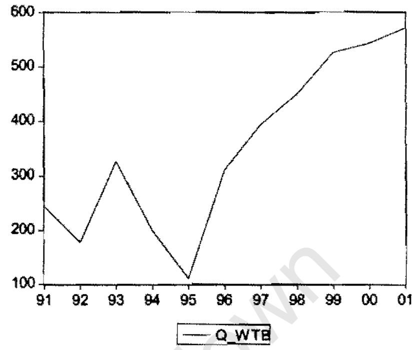
Quantity sold of white rhino



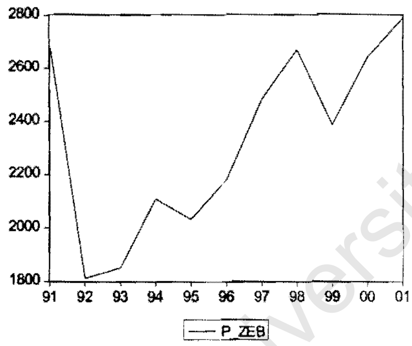
Real average price of waterbuck



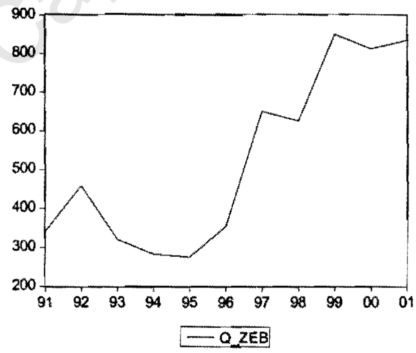
Quantity sold of waterbuck



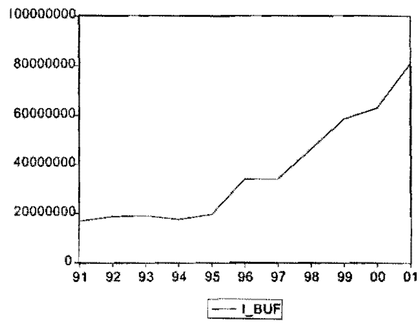
Real average price of zebra



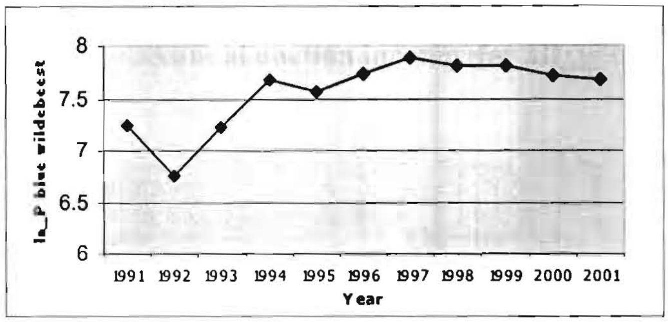
Quantity sold of zebra

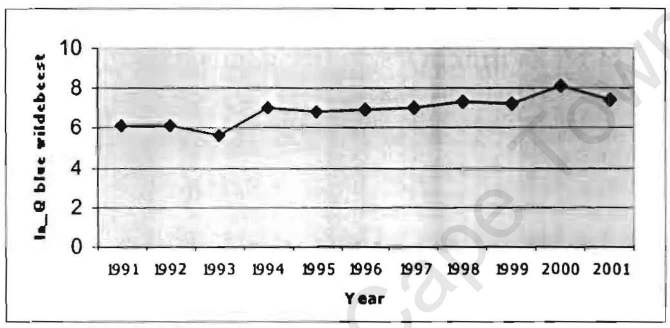


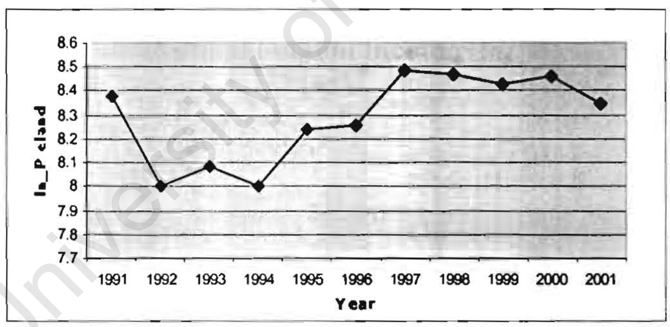
Annual auction income (for all species)

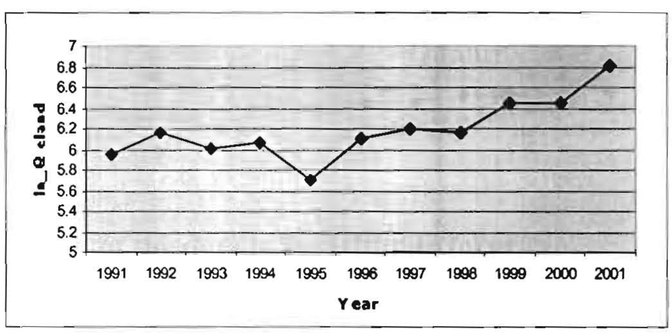


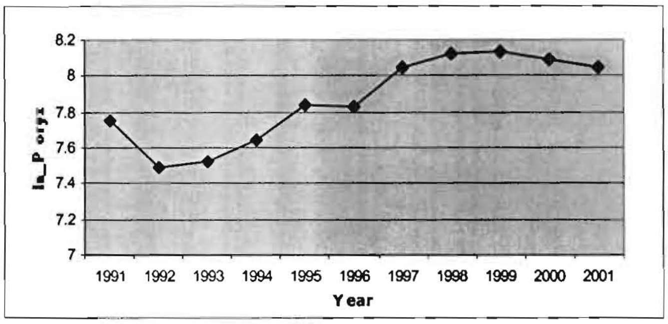
Graphical representation of the data in logarithmic form










8.2
8
7.8
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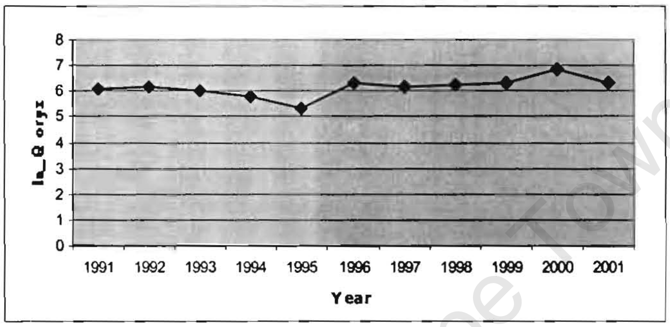
7.4
7.2
7
ln_P oryx
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year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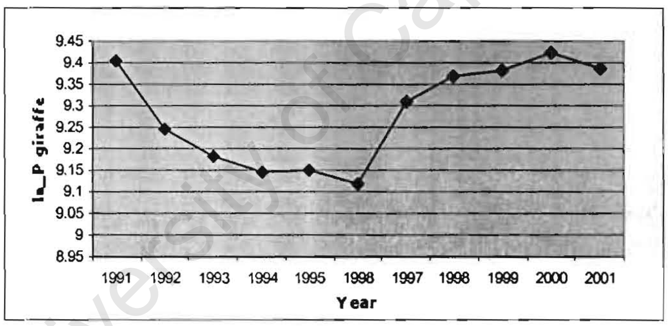
ln_Q oryx
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year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9.4
9.35
9.3
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9.2
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9
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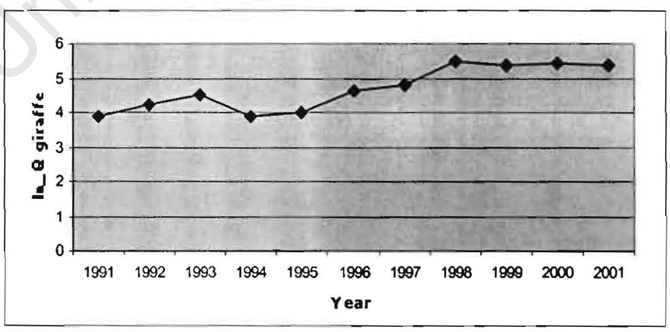
ln_P giraffe
1991 1992 1993 1994 1995 1998 1997 1998 1999 2000 2001
Year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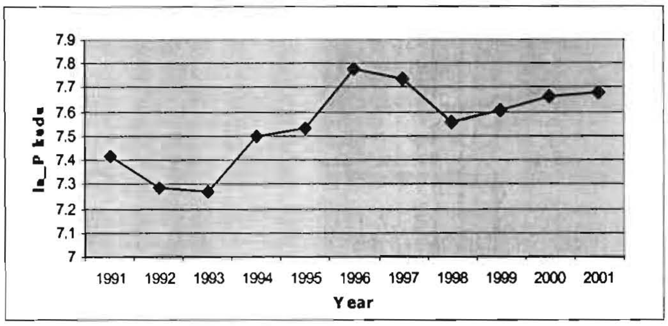
ln_Q giraffe
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year



ln_P kedu
7.9
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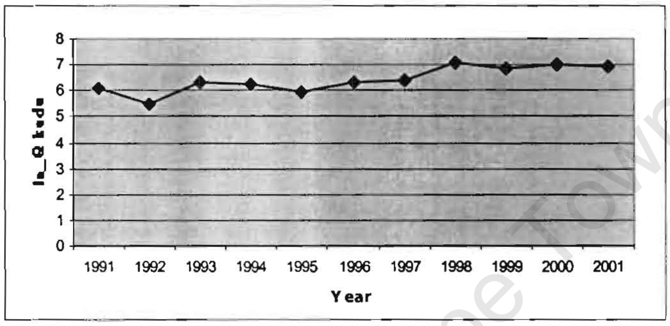
Year



ln_Q kedu
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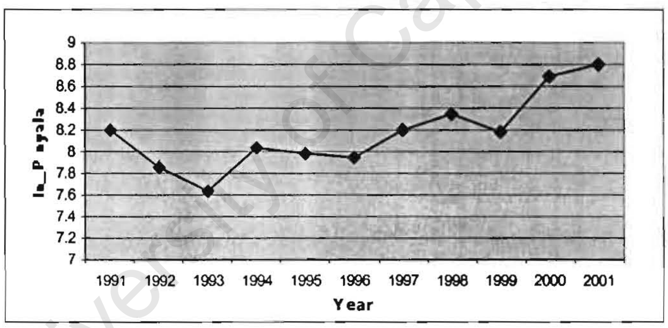
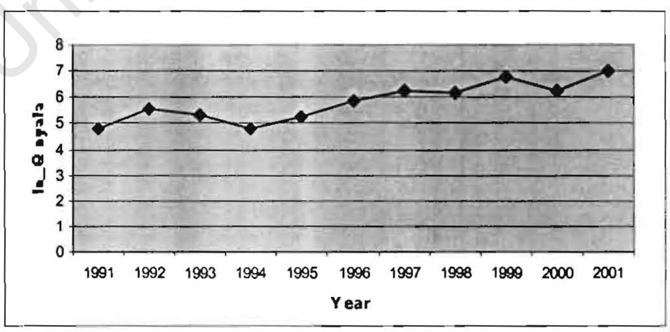
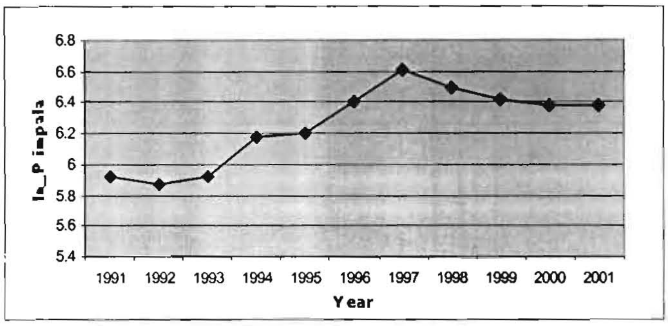
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

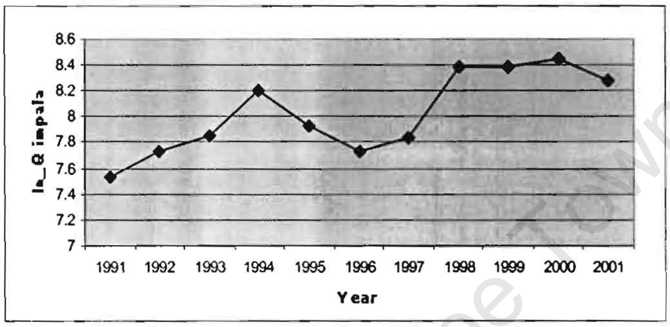


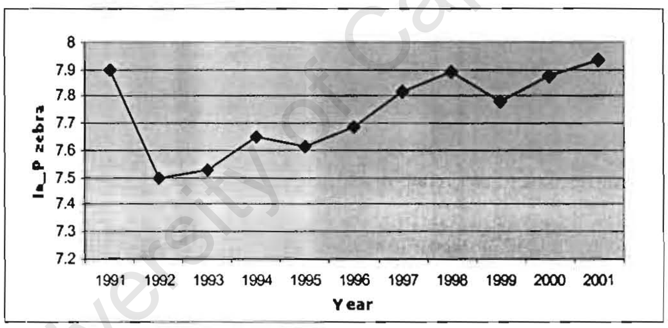
ln_P ayala
9
8.8
8.6
8.4
8.2
8
7.8
7.6
7.4
7.2
7
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

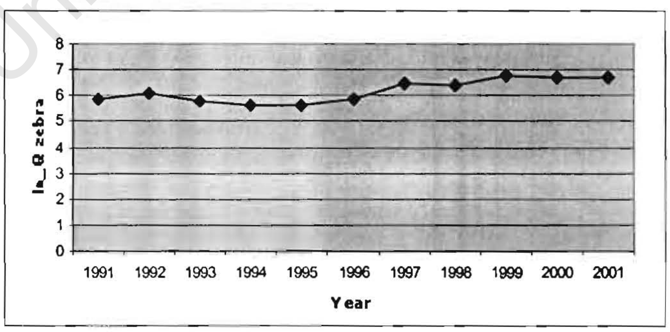


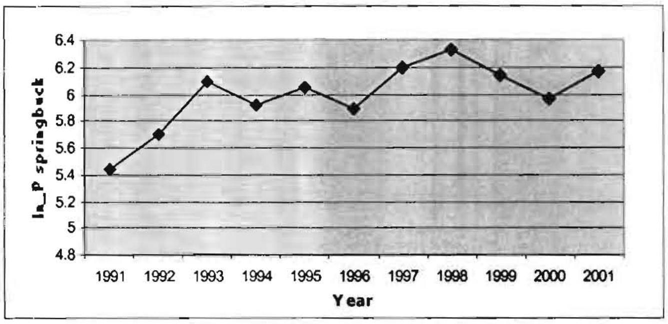
ln_Q ayala
8
7
6
5
4
3
2
1
0
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

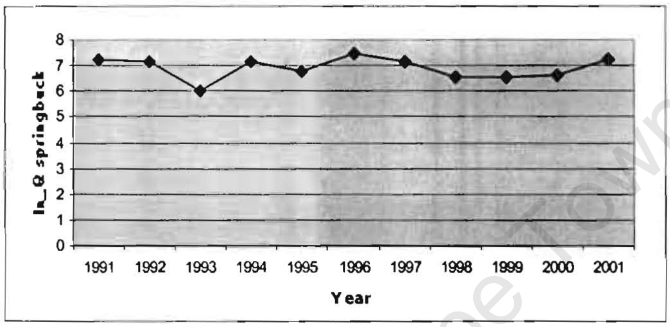

6.8
6.6
6.4
6.2
6
5.8
5.6
5.4
ln_P impala
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

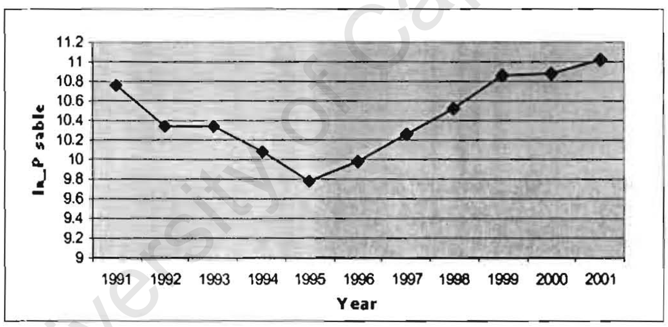

8.6
8.4
8.2
8
7.8
7.6
7.4
7.2
7
ln_Q impala
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

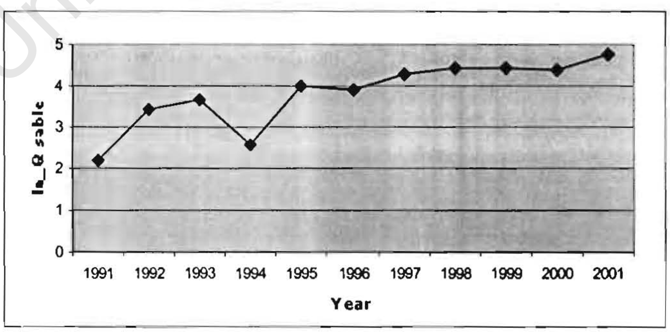

8
7.9
7.8
7.7
7.6
7.5
7.4
7.3
7.2
ln_P zebra
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

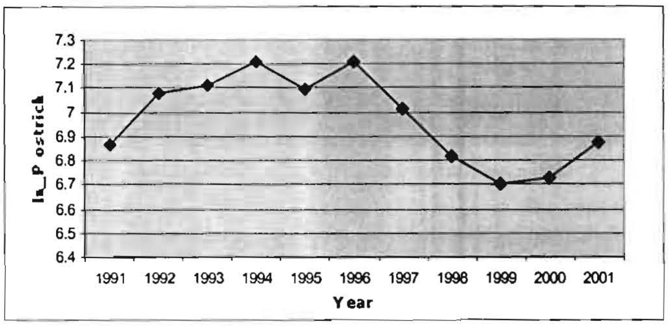

8
7
6
5
4
3
2
1
0
ln_Q zebra
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

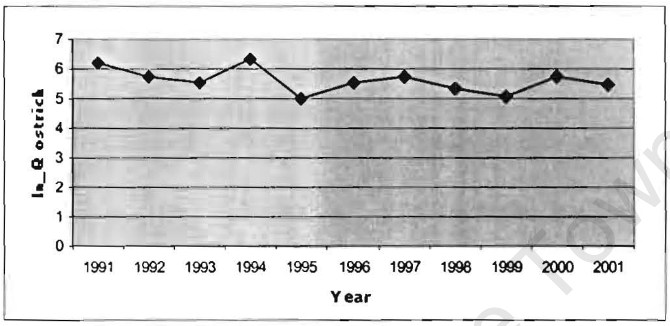

ln_P springbuck
Year
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001

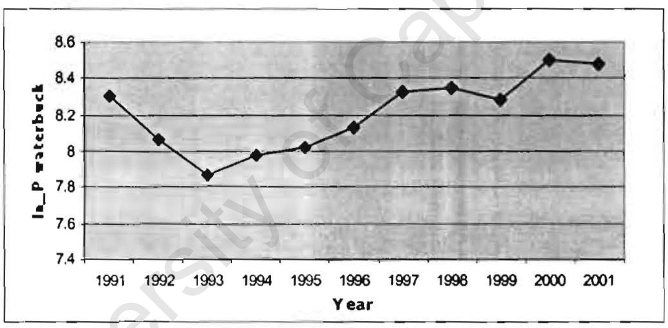

ln_Q springbuck
Year
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001

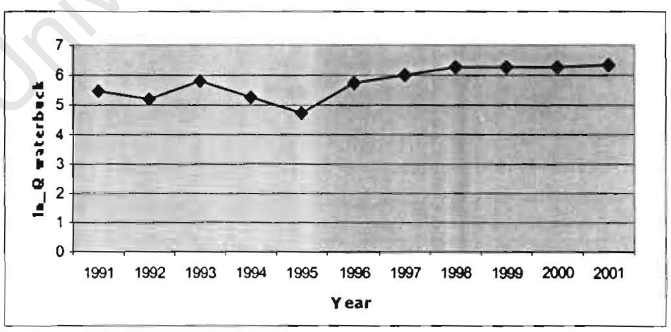

ln_P sable
Year
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001

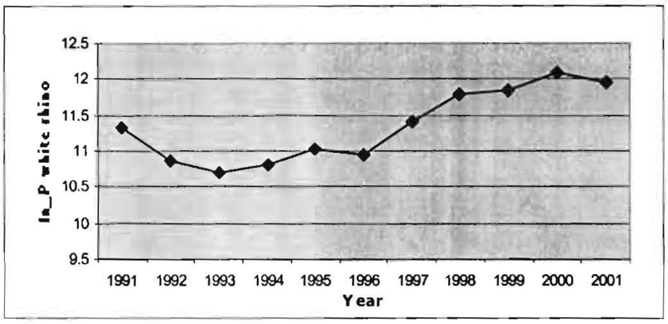

ln_Q sable
Year
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001

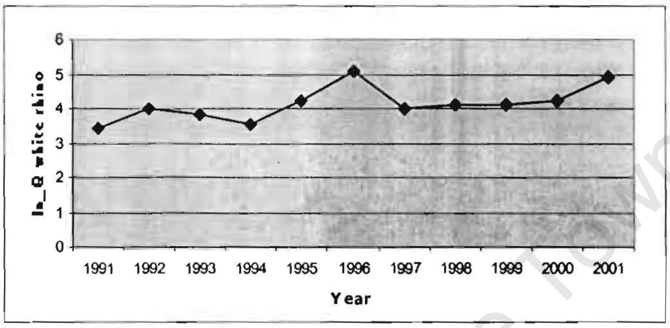

ln_P ostrich
7.3
7.2
7.1
7
6.9
6.8
6.7
6.6
6.5
6.4
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

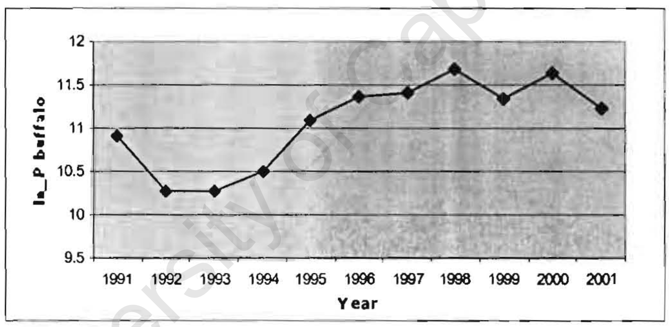

ln_Q ostrich
7
6
5
4
3
2
1
0
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

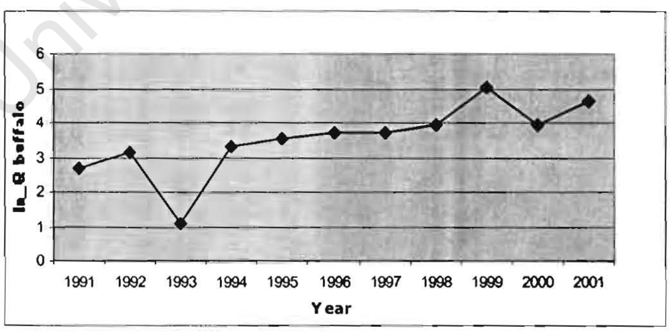

ln_P waterbuck
8.6
8.4
8.2
8
7.8
7.6
7.4
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year

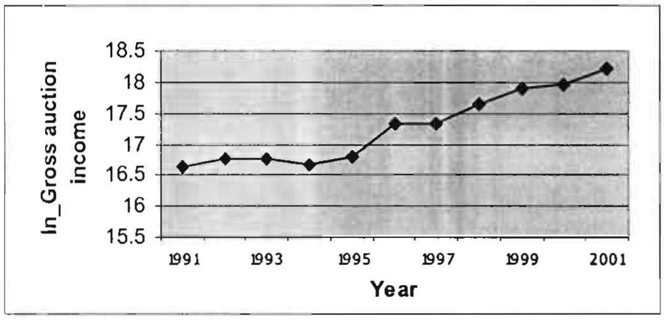

ln_Q waterbuck
7
6
5
4
3
2
1
0
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year


ln_P white rhino
12.5
12
11.5
11
10.5
10
9.5
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year


ln_Q white rhino
6
5
4
3
2
1
0
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year


ln_P buffalo
12
11.5
11
10.5
10
9.5
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year


ln_Q buffalo
6
5
4
3
2
1
0
1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001
Year



ln_Gross auction income
18.5
18
17.5
17
16.5
16
15.5
1991
1993
1995
1997
1999
2001
Year

APPENDIX 5.2: Individual log-linear models of 10 of the species

SUMMARY OUTPUT: **Blue Wildebeest**

Regression Statistics	
Multiple R	0.860825655
R Square	0.741020809
Adjusted R Square	0.676276011
Standard Error	0.398144002
Observations	11

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	2	3.62857309	1.814286545	11.44525637	0.004498415
Residual	8	1.26814917	0.158518646		
Total	10	4.89672226			

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Intercept	-10.8644704	3.796149712	-2.861971004	0.021087441
Auction Inc	0.662692797	0.272103054	2.435447845	0.040853571
Price	0.833942774	0.463939482	1.797524908	0.109968177

SUMMARY OUTPUT: **Oryx**

Regression Statistics	
Multiple R	0.730770196
R Square	0.534025079
Adjusted R Square	0.417531349
Standard Error	0.284015599
Observations	11

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	2	0.739560121	0.36978006	4.584152966	0.047146578
Residual	8	0.645318886	0.080664861		
Total	10	1.384879007			

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Intercept	-0.191287019	2.981432362	-0.064159436	0.950417484
Auction Inc	0.678450809	0.286483471	2.368202279	0.04537356
Price	-0.684475637	0.69938688	-0.978679549	0.356402517

SUMMARY OUTPUT: Giraffe

Regression Statistics	
Multiple R	0.889453338
R Square	0.791127241
Adjusted R Square	0.738909051
Standard Error	0.327676079
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	3.25344896	1.626724	15.15041	0.001903387
Residual	8	0.8589729	0.107372		
Total	10	4.11242186			

	Coefficients	Standard Error	t Stat	P-value
Intercept	3.44191615	1.038076299	3.315668	0.010608
Auction Inc	2.46471E-08	6.0912E-09	4.046348	0.003703
Price	3.03529E-05	0.000107397	0.282622	0.784641

SUMMARY OUTPUT: Kudu

Regression Statistics	
Multiple R	0.823198059
R Square	0.677655044
Adjusted R Square	0.597068805
Standard Error	0.314050143
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	1.658730292	0.829365146	8.409066528	0.010796513
Residual	8	0.78901994	0.098627493		
Total	10	2.447750233			

	Coefficients	Standard Error	t Stat	P-value
Intercept	5.574092519	0.656659767	8.488554957	2.84312E-05
Auction Inc	1.74215E-08	5.56897E-09	3.128318725	0.014050926
Price	0.00011179	0.000390903	0.285978184	0.782160303

SUMMARY OUTPUT: Impala

Regression Statistics	
Multiple R	0.73073086
R Square	0.53396759
Adjusted R Square	0.417459488
Standard Error	0.243068454
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	0.541559231	0.270779616	4.583094046	0.047169848
Residual	8	0.472658188	0.059082273		
Total	10	1.014217419			

	Coefficients	Standard Error	t Stat	P-value
Intercept	7.530030463	0.339658937	22.16938711	1.81109E-08
Auction Inc	9.52165E-09	4.32539E-09	2.201338509	0.058870898
Price	0.000268573	0.000749573	0.35830169	0.729388225

SUMMARY OUTPUT: Zebra

Regression Statistics	
Multiple R	0.897592986
R Square	0.805673169
Adjusted R Square	0.757091462
Standard Error	0.222056493
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	1.635471846	0.817736	16.58388	0.001426038
Residual	8	0.394472688	0.049309		
Total	10	2.029944534			

	Coefficients	Standard Error	t Stat	P-value
Intercept	-6.534685078	3.542407302	-1.844702	0.102305
Auction Inc	0.667313985	0.161563278	4.130357	0.003297
Price	0.153041728	0.609488802	0.251099	0.808067

SUMMARY OUTPUT: **Springbok**

Regression Statistics	
Multiple R	0.531455397
R Square	0.282444839
Adjusted R Square	0.103056049
Standard Error	0.405682851
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	0.518252777	0.259126389	1.574484329	0.265106984
Residual	8	1.316628607	0.164578576		
Total	10	1.834881385			

	Coefficients	Standard Error	t Stat	P-value
Intercept	9.506555704	3.966715901	2.396580935	0.04340772
Auction Inc	0.228902935	0.267217619	0.856616176	0.416576924
Price	-1.094884284	0.619902086	-1.766221325	0.11534847

SUMMARY OUTPUT: **Ostrich**

Regression Statistics	
Multiple R	0.415010973
R Square	0.172234108
Adjusted R Square	-0.034707366
Standard Error	0.416690202
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	0.289020199	0.1445101	0.832284147	0.469494069
Residual	8	1.389045797	0.173630725		
Total	10	1.678065996			

	Coefficients	Standard Error	t Stat	P-value
Intercept	11.91037398	11.02114386	1.080684014	0.311336294
Auction Inc	-0.316488345	0.306325928	-1.033175178	0.331741508
Price	-0.120450915	0.966516334	-0.124623776	0.903895996

SUMMARY OUTPUT: Waterbuck

Regression Statistics	
Multiple R	0.850138175
R Square	0.722734917
Adjusted R Square	0.653418646
Standard Error	0.318313651
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	2.112926403	1.056463	10.42663	0.005909908
Residual	8	0.810588641	0.101324		
Total	10	2.923515044			

	Coefficients	Standard Error	t Stat	P-value
Intercept	-8.708733164	3.955725969	-2.201551	0.058851
Auction Inc	0.712783993	0.284250099	2.507595	0.036507
Price	0.262802886	0.790129553	0.332607	0.747979

SUMMARY OUTPUT: Buffalo

Regression Statistics	
Multiple R	0.776522905
R Square	0.602987821
Adjusted R Square	0.503734777
Standard Error	0.732045944
Observations	11

ANOVA

	df	SS	MS	F	Significance F
Regression	2	6.511355039	3.255678	6.075258	0.024843645
Residual	8	4.287130116	0.535891		
Total	10	10.79848516			

	Coefficients	Standard Error	t Stat	P-value
Intercept	-19.34783368	6.880234666	-2.812089	0.02277
Auction Inc	0.925620149	0.603487488	1.533785	0.163626
Price	0.622938862	0.684579733	0.909958	0.389437

APPENDIX 5.3: Different panel data model specifications used in Chapter 5 with corresponding models using that particular specification

Number	Specification	Models
1	lnQ = fn(lnP, lnI) with P & I pooled	P.5.3.1 P.5.4.1 L.5.3.1 W.5.4.1
2	lnQ = fn(lnP, lnI) with I coefficient varying by species and P pooled	W.5.4.2 S.5.4.2
3	lnQ = fn(lnP, lnI) with P & I coefficients varying by species	S.5.4.3

Note: Q = quantity, P = price, I = income, and

P = pooled model, L = LSDV model, W = "within" model, and S = SUR model

Models are numbered:

- firstly by type of model: P, L, W, S
- secondly by chapter and section: 5.3 and 5.4
- thirdly by model specification: 1, 2 and 3

APPENDIX 5.4: F-test for poolability

H_0: $\delta = \delta_i$ for all i

H_A: not all equal to δ

The F-statistic is then constructed as follows (this is an extension of the Chow test in Baltagi (2001)):

$$F = \frac{(RRSS - URSS) / (N - 1)(K + 1)}{(URSS) / N(T - (K + 1))}$$

under $H_0 \sim F\,((N - 1)(K + 1), N(T - (K + 1)))$

with the strict assumption that $u \sim (0, \sigma^2 I_{NT})$

Where: RRSS (restricted residual sum of squares) is from the pooled OLS, and URSS (restricted residual sum of squares) is the sum of the RSS of the individual models

T is the number of time periods

N is the number of cross-sections

K is the number of explanatory variables

Determine critical value at 5% significance level

If F > CV, the null hypothesis is rejected in favour of the alternative that the coefficients are not all equal across the cross-section.

APPENDIX 5.5: Testing for heteroscedasticity

H_0: $\sigma_i^2 = \sigma^2$ for all i

H_A: not all equal to i

$$\text{LM} = \frac{T}{2}\sum_{i=1}^{N}\left(\frac{\sigma_i^2}{\sigma^2} - 1\right)^2 \sim \chi^2(\text{N-1})$$

Where:

Estimated $\sigma^2 = (\frac{1}{NT}) \times (e'e)$ from OLS pooled model

Estimated $\sigma_i^2 = (\frac{1}{T}) \times (e_i'e)$ from individual regressions

If LM > CV (at 5% level), then H_0 is rejected in favour of the alternative.

APPENDIX 5.6: Testing the joint validity of fixed effects

H_0: $\mu_1 = \mu_2 = \ldots = \mu_{N-1} = 0$

H_A: not all equal to 0

The F-statistic is then constructed as follows (this is another application of the Chow test, from Baltagi (2001)):

$$F = \frac{(RRSS - URSS) / (N - 1)}{(URSS) / (NT - N - K)}$$

under $H_0 \sim F\ (N - 1),(NT - N - K)$

Where: RRSS is from the pooled OLS model, and
URSS is from the LSDV model
T is the number of time periods
N is the number of cross-sections
K is the number of explanatory variables

Determine critical value at 5% significance level

If F > CV, the null hypothesis that all fixed effects are equal to zero is rejected in favour of the alternative.