Conservation and sustainable use of wildlife — an evolving concept

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The proposition that wildlife conservation can sometimes be enhanced through allowing and even promoting the harvesting of wildlife is a sensitive issue. For the last 30 years, conservation has tended to focus on protecting rather than using wildlife. Yet conservation through sustainable use (CSU) is now a mainstream conservation strategy, and research on sustaining rather than stopping uses is commonplace. This paper discusses some of the fundamental and confusing elements of the CSU concept. Two case histories are discussed: Saltwater Crocodiles *Crocodylus porosus* in the Northern Territory of Australia, and Hawksbill Turtles *Eretmochelys imbricata* in Cuba. That wildlife populations are themselves highly dynamic entities, capable of adapting to harvest reductions, is well established, but often not appreciated. To advance conservation, research at the dynamic population level of resolution needs to take precedence over research on individual population dynamics.

INTRODUCTION

ALL people have an inherent interest in wildlife conservation, in the sense that few would actively promote the eradication of all species and most would mourn the loss of at least some species. But there are vast differences in the capacity and motivation of different individuals, communities, peoples and even nations to pursue conservation. Many arguments justifying conservation are small comfort to people living in poverty.

That the worst environmental threat is poverty (Brundtland *et al.* 1987) is obvious but chilling. Obvious, because we all know that if we lived in poverty ourselves, we would kill critically endangered animals to feed our children. Chilling, because the time scales required to solve the problems of poverty are an order of magnitude greater than those needed for at least short-term solutions to many of today's wildlife conservation problems.

However, conservation efforts over the last 30 years have sometimes increased hardship and poverty, particularly amongst rural people, which is intuitively counterproductive. Rural peoples have often been relocated, dispossessed of lands, restricted from water, wetlands, forests, wildlife and marine environments, not to mention loss of traditional incomes. Yet rural people tend to be the ones who ultimately determine whether habitats and wildlife species are retained or lost, regardless of well-meaning laws.

These problems can be dismissed as unavoidable, or "collateral damage", but an increasing body of conservationists believe local people should not be treated as the enemy of conservation (Hutton and Dickson 2000). They should be active partners, at the frontline. To achieve and sustain this, they need to receive tangible, sustainable benefits for their efforts. In most cases, the only sustainable way of providing those benefits is through using wildlife for economic gain. That is, conservation through sustainable use (CSU).

CSU is not a conservation or economic panacea. It is simply a conservation tool with a long history of success in game management (McCullough and Barrett 1992; Hudson 1993; Caughley and Sinclair 1994), that can be applied in some circumstances. There are no definitive, prescriptive guidelines to follow, because CSU programmes need to be tailored to local circumstances. However, important elements of successful programmes are identified in the recent IUCN (World Conservation Union) policy statement on sustainable use (Appendix I). CSU is germane to this conference, because it often involves new or increased interactions between people and wildlife, in highly diverse contexts. It also requires compromises between conservation goals on the one hand, and animal health and welfare considerations on the other.

GETTING CONSERVATION AND SUSTAINABLE USE IN FOCUS

To solve any problem efficiently, the level of resolution of the problem needs to be matched to the level of resolution of the solution. We first need to identify the key variables associated with the problem, and then address them to the level needed to solve the problem, rather than to the level technically possible. To construct a building, we do not measure bricks to ten decimal places, even though we have the technology to do so: nor we do measure the bricks but ignore the size of the foundations.

The key variables causing conservation problems are typically social, cultural, economic and biological ones. All four need to be addressed to a similar level of resolution when formulating conservation solutions. That

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biological variables are often dealt with in "great detail, but socioeconomic variables ignored, is commonplace. It reflects the fact that biologists engaged in conservation work are often trying to solve the lack of biological information as a problem in its own right.

With CSU, we need initially to examine the high level of resolution variables involved — get the big picture in focus — before looking at any of the operating variables (e.g., biological variables) in more detail.

Unwilling partners in conservation

Of the 23 species of crocodilians in the world, one is really critically endangered, and may disappear from the wild in the next few years unless urgent action is taken. An already remnant population of Chinese Alligators *Alligator sinensis* has declined from 1 000 individuals (Webb and Vernon 1992) to 150 individuals in seven years (Thorbjarnarson *et al.* 2000a, 2000b). They exist today in 10 small agricultural ponds, within a 600 square kilometre Chinese Alligator Reserve — shared by two million people pursuing intensive agriculture (Thorbjarnarson *et al.* 2000a).

When a conservation team visited one of these sites recently, the local people threw stones at-them! They hate the alligators, which eat their ducks and burrow into and drain their rice fields, causing economic hardship. They do not consider they have benefitted in any tangible way from the efforts to conserve the alligators to date, and have no love of the people who put the welfare of the alligators before that of themselves. They are clearly, unwilling partners.

The same situation exists all over the world, be it: elephants trampling crops and destroying houses; tigers, lions and crocodiles eating people and livestock; parrots, cockatoos and flying foxes raiding seeds and fruit; the 30 000 people each year killed by snake bite in India; wheat crops being eaten by emus and kangaroos; etc.

Conventional conservation is constrained when animals are pests to local people, and it is unrealistic to expect them to be willing partners in efforts to increase their abundance.

What is conservation about?

It is hard to expect people to agree on conservation priorities, or on the strengths and weaknesses of different approaches to conservation, if they do not share a common vision about what conservation is and what its goals are or should be. Yet the critical elements of conservation (Webb 1995, 1997a), can be embodied in a simple definition:

> *Conservation is the sum total of actions taken to preserve and maintain items to which we attribute a positive value.*

This definition applies to conservation in its broadest sense, and is not restricted to wildlife conservation. It accepts that people have had a long history of conserving things (paintings, buildings, stories, legends, religious icons, etc.), and that the principles are always the same. We never expend resources conserving things we consider useless, valueless, or to have a negative value. When applied to wildlife, this definition is not controversial, because value can be expressed as intrinsic value, instrumental value (use values), or mixtures of both.

With the Chinese Alligator example, conservation can theoretically be pursued with or without the people. But if you want them involved, or given past failures, need to have them involved, mechanisms for generating sustainable benefits need to be found. A conservation programme needs to be tailored to the exact local situation, on site, in China, and it should not be unduly constrained by what people in Sydney or London think. Its success or failure should be judged primarily on *what* happens, and *not on how it happens.* In this case, conventional harvest is clearly out of the question, but restocking (from successful captive-bred stocks), and modest payment for new clutches of eggs laid, may be all that is needed to tip the balance.

What is sustainable use?

Some countries allow great freedom in the range of uses acceptable for driving conservation, but others are more conservative. Regardless, decisions about the acceptability of different "uses" initially need to be made on the basis of culture, tradition, convention, ethics, morality and politics. The Australian public may agree to use koalas for ecotourism, but not for meat and skins. We need to resolve problems of acceptability openly and honestly before proceeding to issues about sustaining uses. These are a completely different type of problem, requiring different types of evidence and expertise to evaluate.

Sustainability itself is really a process. To sustain uses of anything from televisions, to cars, to marriages, to wildlife, requires the same procedures'. monitoring, assessment and adjustment. By way of a practical definition, with wide application:

> *Sustainable use is a use of something associated with a process aimed at ensuring that the use can continue and that its impacts are maintained within acceptable or defined limits.*

It is impossible to sustain consumptive or non-consumptive uses of a wild population, or even gain rewards from simply knowing a wild population exists, unless the population itself is conserved. If one takes the widest definition of use possible, it can be argued that *conservation is sustainable use.*

This being so, whether a particular use is sustained or not, can only be measured in hindsight: *was the use kept going?* With foresight one can debate whether the probability of sustaining uses appears high or low on the basis of current information. But the distinction between predicted sustainability — *what I think might happen* — and measured sustainability — *what happened* — can only be determined by measurement. Risk and uncertainty in all variables is possible, and the population dynamics of harvested populations change when density is reduced. Management for sustainability is adaptive management — management which can adjust rapidly to changed insights and events.

Emphasis on habitats

The central aim of most CSU programmes is to create incentives for habitat conservation, particularly on lands outside of protected areas. On these lands there is an expectation that landowners will produce food and generate wealth, typically from farming, grazing, aquaculture or forestry. Natural habitats and the wildlife they contain are usually a casualty.

Rapid large-scale losses of habitat through dams and forestry catch public attention, but most habitat loss involves long series of minor land use decisions, made daily, all over the world, invariably for economic reasons. Eventually it is realized that a wildlife species or its habitat has disappeared or become restricted to well-separated remnants.

The general concept of CSU can be applied to large tracts of habitat, and game ranching is a good example. However, a cattle station owner engaged in some use of crocodiles for commercial purposes, does not need to manage his property like a national park in order for "conservation benefits" to be accrued. Relatively small economic returns can change the perception of a wildlife species or a patch of habitat from a liability to an asset. If they generate no economic return, and for example harbour crocodiles that eat livestock, their future is far less secure.

Biocentric and anthropocentric approaches

That wildlife conservation and sustainable use of wildlife *are* intimately linked in many cases is well established. However, other factors dictate the extent to which this linkage has been used in the past, and will be used to advance conservation in the future. That is, CSU is still the focus of conflict within the conservation arena.

People value wildlife, and put resources into its conservation, for both biocentric and anthropocentric reasons: for the primary benefit of animals or for the primary benefit of people respectively. Over the last 50 years, with television bringing images of wildlife into homes throughout the world, biocentric goals and philosophies have increased in popularity. Many people now hold the intrinsic value of wildlife highly, and it motivates them to support conservation through donations or other means.

Yet we live in a world that is economically driven and decidedly anthropocentric. Thus, even if we value wildlife for its intrinsic value, and hold strongly biocentric views, we should also be able to embrace instrumental values as practical and pragmatic mechanisms for ensuring the wildlife we value is conserved. Arguments such as: *"I would rather see wildlife go extinct than be utilized"* indicate that defending biocentric philosophies has become more important than ensuring wildlife is conserved. CSU attempts to bridge the gap between biocentric and anthropocentric approaches, but it requires compromises.

Constraints on conservation through sustainable use

Proponents of conservation, animal rights and animal welfare are not always part of the same, happy, green family. Consumptive use of wildlife is not acceptable within animal rights philosophies, regardless of whether it leads to conservation benefits or not. Perhaps more important, actions taken to advance animal rights can cause obvious and significant conservation problems. For example:

1. An endemic species of ground boa (snake) on Round Island, off Mauritius, was critically endangered by habitat loss caused by feral goats and rabbits (King 1988). Efforts to eradicate the feral animals with poisons were delayed two years, because the rights and welfare of the feral animals were likely to be compromised. In the interim, the snake went extinct — a conservation tragedy.
2. Releasing farmed mink into the wild in England, in order to advance perceived animal rights goals, appears to have been done with complete disregard for the conservation costs of deliberately releasing new alien predators into the environment, and nor for the welfare of the mink themselves, which had been captive-bred for many generations.

The major constraint on our ability to expand the extent to which sustainable use is used as a conservation tool today, or even test it in some countries, is that science, technology, research, logic, experience and even economic rationklization are very much secondary to political opportunism. Salzman's (1995) words about environmentalism are not an idle prediction:

> *Science is not going to be the deciding factor, or even a major player in the debate but rather the values, opinions, and politics of the players. Scientists will increasingly find that the issues will not be argued on their merits, and that the introduction of scientific evidence will simply be ignored.*

Animal rights and conservation

The concept that animals have rights, in the sense of rights and responsibilities with which people are familiar, is highly biocentric. It is a moral, ethical and philosophical issue, in which the role of people as beneficiaries is couched in vague long-term predictions about fundamental changes in society. It invariably constrains options for pursuing conservation.

The most obvious practical problem with the philosophy is that rights assigned to animals are invariably rights taken away from people, which increase rather than decrease poverty: rights of access, rights of tenure, rights of use, rights of security. Actions transferring rights to animals' tend to alienate people from conservation and foster "unwilling partners". But there are more serious potential problems.

Most people accept a sliding scale of importance with animals, with dogs, cats, whales and dolphins at one end, and snakes, crocodiles, spiders, mosquitos and lice at the other. If we were to accept that animals should be equal to people and have the same rights as people, with which group of animals should we equate people? Or do we put some with the whales and some with the cockroaches? Cockburn (1996) drew attention to the fact that despite many other well-known problems, Germany in the 1930s imposed very strict animal rights legislation on its people. Cockburn's concern about animal rights was not the consequences of people treating some animals as though they were people. It was the extent to which such philosophies, if adopted by Governments, would ultimately justify actions that treated people like animals.

So on the one hand animal rights does lead to more people valuing wildlife, but on the other, it establishes a narrow range of protocols for conservation, which take rights away from people.

Animal welfare and conservation

Animal welfare is a completely different issue, and regardless of its actions being biocentric, it can be justified and defended for purely anthropocentric reasons. Communities of people have little to gain from encouraging unnecessary pain and suffering in animals. Animal welfare needs to be context-specific and not species-specific for equally obvious reasons. An Aboriginal immobilizing a kangaroo with a spear, and using whatever is available to kill the kangaroo before it stumbles off, cannot be expected to follow the codes of practice designed for people caring for an orphaned kangaroo in an urban backyard in Canberra. That is, what constitutes "unnecessary" in a particular animal:human interaction, is directly linked to context. Animal welfare codes, if sensitive to culture, tradition and socioeconomic constraints, benefit both people and animals, and are rarely at odds with conservation.

When whole classes of interactions between people and animals are challenged on biocentric animal welfare grounds, without considering other anthropocentric factors, the situation becomes messy. For example, traditional Aboriginal hunting could be considered "cruel" relative to hunting with modern armaments, or to killing kangaroos raised on closed-cycle farms by injection. But is this a sufficiently strong motivation to ban traditional weapons or ban hunting altogether? How do we balance "relative cruelty" with issues such as maintenance of culture, tradition or incentives to conserve habitats? Extreme biocentric animal welfare issues, which are often intertwined with animal rights philosophies, can result in serious conflict within the community.

Gamebird hunting was banned in New South Wales (Australia) specifically because it was considered "cruel". That hunters owned and managed private wetlands, had strong interests in the status of public wetlands, and mobilized themselves in various capacities to assist waterbird conservation, was ignored. By the year 2000, 5 years later, waterbird numbers in New South Wales had declined rather than increased, rates of wetland habitat loss had increased, there were no public champions for the cause of wild duck conservation and the main value of wild ducks had become a negative one: "pests" on rice crops (Kingsford *et al.* 2000). There were no conservation advantages and significant conservation costs.

CASE HISTORIES

The issues discussed above place CSU as I see it, within the broader context of what mainstream conservation is all about. It highlights a series of theoretical issues that support the view that CSU is a sound approach to some conservation problems, despite philosophical opposition to using wildlife by some segments of the community.

The two case histories presented below highlight some of the practical issues associated with implementing programmes. There is a heavy bias towards the biological variables, because even though it can be argued that they are not the most important ones, they are fundamental to assessing whether or not harvests have been sustained.

Saltwater Crocodiles in the Northern Territory of Australia

In 1971, after three decades of unregulated hunting, the wild population of Saltwater Crocodiles *Crocodylus porosus* in the Northern Territory of Australia had been depleted by at least 95% (Webb *et al.* 1999, 2000): adults in the population had become extremely rare (Webb *et al.* 1986). By 2001, 30 years later, the wild population is back to near pristine levels (around 75 000 individuals) and they occupy their complete historical range.

From a biological perspective, the recovery of Saltwater Crocodiles went through various stages. In the first few years after protection, the small nucleus of remaining adults nested, and despite perhaps 70% mortality of eggs, mainly due to wet-season flooding, hatchlings and juveniles showed high survival rates. The recovering population became strongly biased towards juveniles in the first few years after protection (Fig. 1).

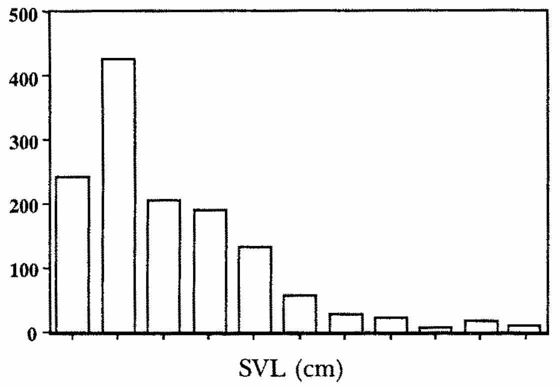


Fig. 1. Size distribution (SVL; snout-vent length = half total length) for 1 354 *C. porosus* caught in the Northern Territory between 1973 and 1976 (after Webb and Messel 1978). Crocodiles less than 75 cm SVL had hatched at or near the time of protection (1971). There were few adults (120cm SVL for females; 155 cm SVL for males) in the population.

The population continued to expand numerically until around 20 years after protection (Fig. 2). The size structure of the population changed dramatically over this time, in the direction of there being more large crocodiles (Fig. 3; Table 1). This was achieved through a series of density-dependent adjustments.

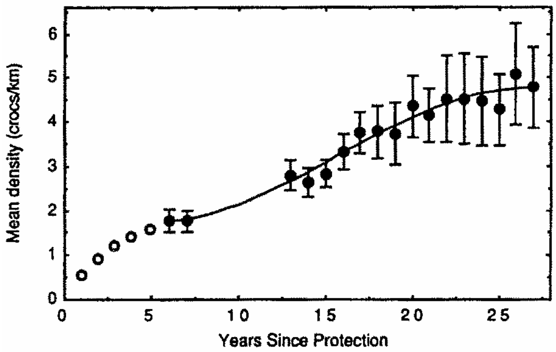


Fig. 2. Mean (mean of means) non-hatchling density (crocs/km) for standardized spotlight counts of *C. porosus* from 11 major rivers in the Northern Territory. Vertical bars represent one standard error from the mean. The line is the third order polynomial regression of best fit ($r^2 = 0.917$, $p = 0.0001$). Years Since Protection: 1 = 1972; 27 = 1998. Open dots reflect the general trends between 1972 and 1976.

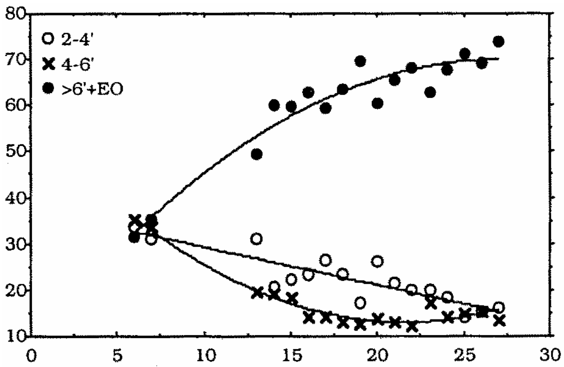


Fig. 3. Percentage contribution of different size classes (total length) of C. *porosus* (2-4'[1], 4-6' and >6'+EO) to the total number of non-hatchling crocodiles sighted in spotlight surveys in 11 Northern Territory rivers since protection (1 = 1971; 27 = 1998).

Table 1. Mean percentage of *C. porosus* in different size classes sighted in 1975-76 (37 rivers) and 1997 (11 rivers). <2' = hatchlings. EO = crocodiles sighted as "eyes only" (no size estimate) which are mainly >6' long (Webb *et al.* 1986).

Size Class (feet)	Mean Percentage 1975–76	1997
<2'	22.8	25.0
2–3'	23.3	7.3
3–4'	17.1	4.5
4–5'	13.9	5.4
5–6'	5.9	5.9
6–7'	2.3	7.5
>7'	3.3	19.6
EO	11.4	24.7

Juvenile survival rates proved to be highly density-dependent, but in complex and unpredictable ways (Messel *et al.* 1981; Webb and Manolis 1992). The percentage of adult females which nested declined as population size increased. The survival rate of hatchlings depended on the numbers of hatchlings produced. In years with lots of hatchlings\ few survived to 1-year of age, whereas in years with few hatchlings, a high proportion survived. Survival from 1-2 years was determined by the numbers of large crocodiles in the area (cannibalism?). Survival from 2-3 years and 3-4 years, was dependent on the total number of crocodiles occupying the area, with new recruits being forced to move, and in the process, suffering higher mortality rates. These mechanisms promoted rapid recovery in the population when it was greatly depleted, but slowed recovery rates as densities increased.

Between 1971 and 1979 management involved strict protection, and it worked: populations increased. But there was a long way to go before new adults were recruited into the population (12-16 years to reach maturity). By the late 1970s, 9 years after protection, crocodiles were abundant and calls for culling started. The political pressure for culling mounted in 1979-80, when a series of crocodile attacks on people occurred. Management adjusted (Webb *et al.* 1987a) by: upgrading a pubic education programme; introducing a formal problem crocodile programme to remove crocodiles, attacking livestock or taking up residence in Darwin Harbour; and, crocodile farms were started with a view to making crocodiles a commercial asset to the Northern Territory. Tourism based on showing and feeding wild crocodiles was promoted in the early 1980s as a further means of increasing the value of wild crocodiles.

In 1983, with landowners starting to lose livestock, management changed again. This time it provided for a ranching programme, in which landowners were paid for wild eggs collected on their lands. In 1995 a further adjustment was made, which allowed landowners permission to undertake limited wild harvests, and sell the skins and meat.

Despite continual opposition from some organizations, the combination of CSU and adaptive management achieved its goals. The wild population did recover, and when one considers the type of predator Saltwater Crocodiles are, public support for that recovery was a remarkable achievement. With the benefit of hindsight, we can now examine the validity of some early predictions and claims:

Ecological role

In the immediate post-protection phase, justification for public support was based on the claim that crocodiles, as a "top-end" predator, were ecologically important. For example, it was suggested that they ate catfish which ate barramundi: the implication being that the commercially important barramundi may be less abundant when crocodile numbers are down. In reality, these types of arguments proved to be more in the category of myths and legends than science (Gorzula 1987). The recovery of Saltwater Crocodiles was not associated with any obvious change in the ecology. They are now most abundant in the same areas where barramundi are abundant, because food is plentiful. Recreational fishing has become more hazardous, and crocodiles can readily be observed eating barramundi rather than catfish! There is little doubt that the recovered crocodile populations now eat many more barramundi and other vertebrates (including threatened species) than they did at the time of protection when densities were low.

Pristine abundance

In the early 1970s, cursory examination of historical trade statistics and reports from early explorers were interpreted as indicating that the original population in the Northern Territory was immense everywhere. Detailed checking of the historical records and interviews with 60 people involved with the crocodile industry in 1945-71 indicated that this was a serious error (Webb rf *al.* 1984). What was interpreted as a 1-2% recovery by the early 1980s, suggesting the conservation programme was an expensive failure, can now be reinterpreted as a 40-50% recovery. Crocodile densities are now highly patchy, with 20 per km in some rivers, but less than 1 per km in most. This was the situation originally, and is unlikely to change in the future.

Nest numbers would be a good index of population trends

In the mid-1970s, Magnusson (1980) predicted that there were potentially 933 suitable nest sites in the Liverpool and Tomkinson Rivers, and that the wild population could increase by a factor of 18. In reality the adult population in this area increased by a factor of 0.5 and is now stable (Webb *et al.* 1999, 2000): nest numbers increased by less. These rivers never supported high densities of crocodiles, and as demonstrated in American Alligators *Alligator mississippiensis* (Mines and Abercrombie 1987), we now know a bottleneck occurs in adult-sized females, and thus social structure rather than habitat limits the number that nest. In some highly depleted rivers, nest numbers did increase with the increasing population, offset by around 10 years (Fig. 4), but in other areas nest numbers remained stable over time (Fig. 4) and have only increased recently. Nest numbers are not necessarily a good index of population status.

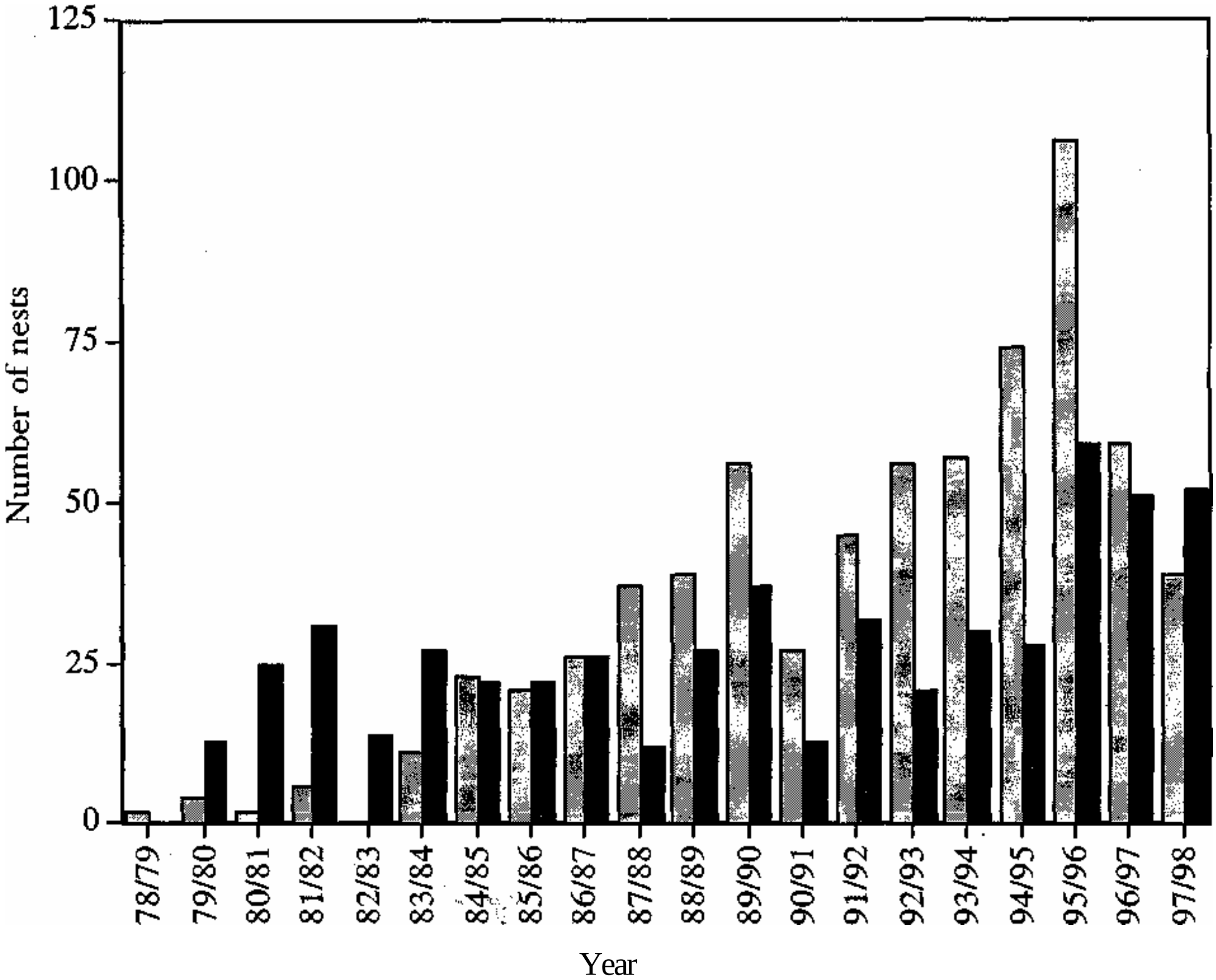


Fig. 4. Numbers of nests located in the Adelaide River system (shaded bars) and Melacca Swamp (solid bars). The nesting season for *C. porosus* typically spans October-May.

Harvesting would compromise the recovery of wild populations

Throughout the period of recovery, the populations were harvested at varying levels [Aboriginal subsistence (150/year); problem crocodiles (200/year); incidental catch in fishing operations (500/year); ranching of eggs (200 000 eggs between 1983 and 2001); landowner harvests (250/year)]. Yet trends in the population size and size structure in harvested rivers were no different from those in river systems which were not harvested (Webb *et al.* 1999, 2000). That is, the total harvest impact was well below any maximum harvest level (Fig. 2), and may have been biologically negligible.

Consumptive use for commercial gain would not assist conservation

There is no doubt that the economic value of crocodiles through tourism, farming, harvesting and the attraction of media attention to the Northern Territory (tourism is the second largest industry), gave the people and Government of the Northern Territory sound reasons for wanting to conserve crocodiles, maintain responsible management, and tolerate the return of crocodiles in the wild. Since the egg harvest programme was initiated, we know of no patches of swamp used for nesting on private lands that have been lost. Some remote Aboriginal communities have made more income from harvesting and selling crocodile eggs and hatchlings, than they have from any other commercial activities. Although some landowners may well take problem crocodiles into their own hands from time to time, the majority see crocodiles as a modest asset, and now have avenues through which they can get some economic return from them. The commercial value has added to other values — not detracted from them.

Hawksbill Turtles in Cuba

The situation with Hawksbill Turtles *Eretmochelys imbricata* in Cuba has many parallels with Saltwater Crocodiles in the Northern Territory. Both are widely distributed aquatic reptiles, with similar life histories, and both are highly mobile and cross international borders (Webb 1997b, 2000). Both are used by local people for food and trade, and the status of both varies greatly from country to country. International trade was a factor contributing to the unsustainable use of both species in various parts of their range. They were both listed on Appendix I of CITES, which prevented international trade in products derived from the wild, or even derived from wild-harvested eggs.

Australia was successful in obtaining an Appendix II listing of Saltwater Crocodiles (in 1995), which allowed the Northern Territory to export crocodile skins, and generate the economic benefits required to drive the CSU programme. Cuba was not successful in obtaining an Appendix II listing (in 1997 and 2000), and so they have a programme based on CSU, but no economic benefits to drive it.

Cuba is the largest island in the Caribbean, and contains an estimated 32% of all shallow water coral reef habitats in the Caribbean (ROC 2000). It supports extensive sea turtle populations, and historically these were harvested for food through a Government fisheries programme. The meat was used to feed people in Cuba, and the only producf exported was the shell plates of Hawksbills. In 1990, changes in the eastern European bloc countries, which were Cuba's main trading partners, resulted in fuel shortages and a severe economic downturn. The Government shifted fishing effort away from the largely domestic turtle harvest, despite turtles still being abundant, so that fishing resources could be concentrated on more lucrative export fisheries. Cuba joined CITES in 1990, and protected sea turtles throughout all Cuban waters as a goodwill gesture. The only exception was two remote local communities, where turtle fishing was a traditional activity.

A detailed assessment of the historical harvest (Carrillo *et al.* 1999) of 5000 Hawksbills per year, demonstrated that from 1983-90, abundance and the mean size of turtle caught had remained constant in some harvest areas, but had been declining gradually in others. The historical harvest (Fig. 5) had clearly demonstrated a high degree of sustainability before it was phased down, although reductions in the harvest in some areas may have been warranted. The question today is not whether the historical harvest was sustainable, but rather whether the present traditional harvest of up to 500 Hawksbills per year is sustainable.

The monitoring data are unequivocal about this (ROC 2000). Catch-per-unit effort is increasing (Fig. 6), and the mean size of turtle caught is increasing (Fig. 7). Juvenile Hawksbills are abundant in the main foraging (growing) areas (Table 2), and standardised nesting surveys indicate an increase of more than 100% in the

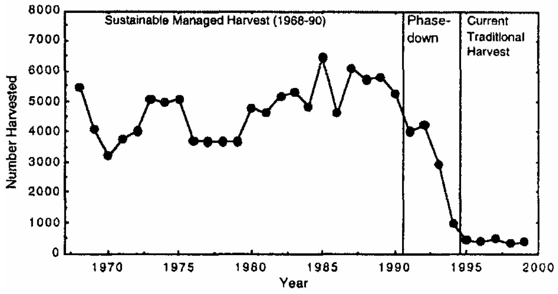


Fig. 5. Harvest data for *E. imbricata* in Cuba. Cuba sustained a regulated harvest of around 5 000 Hawksbills each year, before voluntarily phasing the harvest down to no more than 500 per year, and restricting the harvest area to two local communities and less than 0.5% of the area occupied by Hawksbills in Cuba (ROC 2000).

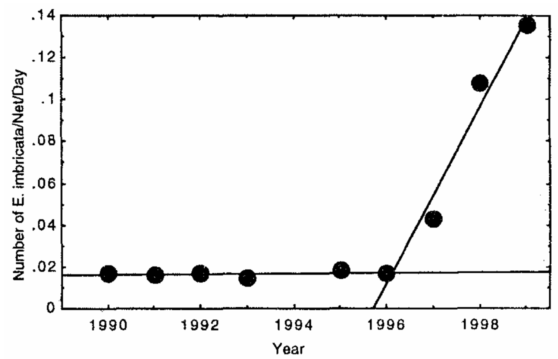


Fig. 6. Catch per unit effort (Jan-Apr, Aug-Dec) for *E. imbricata* at the Isle of Pines, Cuba. Lines are regressions for periods 1990-96 (CPUE remained stable; $r^2 = 0.08$, $p = 0.58$), and 1996-99 (significant increase; $r^2 = 0.97$, $p = 0.016$). Data from some months in 1994 are missing; CPUE for 1999 has been extrapolated from harvest data to 31 October 1999 (ROC 2000).

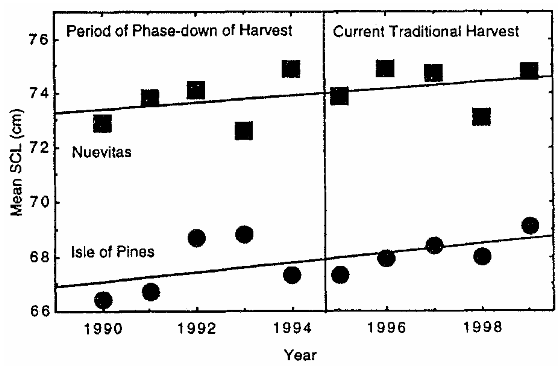


Fig. 7. Mean size (SCL; straight carapace length) of *E. imbricata* landed at the Isle of Pines (circles) and Nuevitas (squares) since 1990. Lines indicate regression relationships. Since the phase-down of the harvest at the Isle of Pines, the increase in the mean size of *E. imbricata* caught reaches significance (1995-99). Data for 1999 are to 18 October (Isle of Pines) and 30 September (Nuevitas)(ROC 2000).

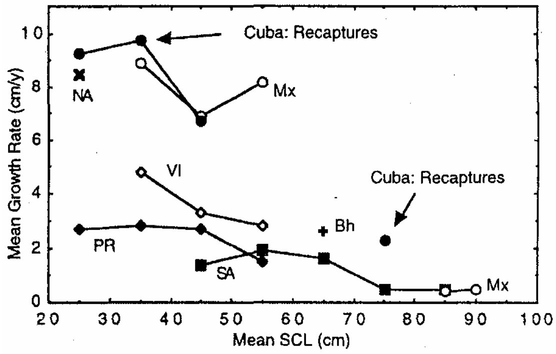
Table 2. Estimates of the density of wild juvenile, adult and subadult *E. imbricata* in Cuba and other countries (ROC 2000).

Country	Density (N/km^2)	Notes
Australia	81	Intertidal reef on north-west coast of Australia (Fog Bay, Northern Territory) (Whiting and Guinea 1998).
Australia	3	Heron Island Reef, southeastern Australia (Limpus 1992).
Cuba	280	Juveniles in 3 km long by <20 m wide swim transect in Doce Leguas coral reef habitat.
Cuba	122	Juveniles in a 300 m long by 300 m wide area in Doce Leguas coral reef habitat.
Cuba	59	4.3 km long by <20 m wide swim transects in mixed habitat (coral reef, sand, seagrass and rocks) at the Isle of Pines.
Dominican Republic	6–97	Various habitats (Leon and Diez 1999).
Mexico	3–41	Coral reef, Rio Lagartos, Yucatan Peninsula (Maldonado and Garduño 1999).

last three years. These results are all consistent with a sustainable harvest being extracted from a wild population, just as occurred with Saltwater Crocodiles.

Opposition to Cuba being given permission to trade reflects more the charismatic nature of sea turtles than serious consideration of sustainability (Webb 2000). When the species was listed on Appendix I of CITES, it was assumed that the global wild population was small" and disappearing. But this is inconsistent with Cuba sustaining a harvest of 5 000 subadults and adults per year up to 1990, not to mention the 110000 removed between 1968 and 1990. When one adds to this the reported harvests elsewhere in the Caribbean, it is clear that the regional population was much larger than originally assumed. Indeed, population simulation models originally predicted that the wild population needed to sustain Cuba's harvest alone could have contained 20 000 to 180 000 adults, which means perhaps millions of individuals. The answer to this dilemma (a very abundant "endangered" species) came from Cuban research. In the warm waters of southern Cuba and Mexico, the measured growth rates of Hawksbill Turtles are appreciably higher than in other areas where Hawksbills have been studied (Fig. 8).

Faster growth rates (Fig. 8), mean that the mean relationship between size and age in Cuba and Mexico (Fig. 9) is dramatically different to that in other areas where Hawksbills have been studied. For example, the smallest females which nest in Cuban waters (55 cm SCL) are around 7-8 years of age, rather than 21-22 years of age if the wrong growth rates were applied. The size at which all females appear mature (80 cm SCL) is around 15-16 years of age rather than 49 years if they were growing at rates quantified elsewhere. The Cuban and Mexican populations



Fig, 8. Mean relationship between growth rate [straight carapace length (SCL)] and mean size (SCL) for wild *E. imbricata* recaptured in Cuba (N = 10) compared with reported growth rates from other areas: Mexicq (Mx); Bahamas (Bh); US Virgin Islands (VI); northern Australia (NA); southern Australia (SA); Puerto Rico (PR). Raw data from: Limpus (1992); Limpus and Miller 1996; Kowarsky and Capelle (1979); Bjorndal and Bolton (1988); Boulon (1994); Garduno and Marquez (1994, 1996); MIP, unpublished data; Carrillo *et al.* 1998; Diez and Van Dam (1995); and, Wood and Wood (1993). Where necessary, raw data were converted to units of SCL using the formula in Limpus (1992).

are biologically capable of much faster rates of population increase (Fig. 10), which may be intimately linked to their ability to sustain harvesting. The evidence to support the endangered status of Hawksbills is highly questionable (Webb and Carrillo 1999).

In any overview, Cuba's Hawksbill programme has all the elements for sustaining a harvest, sustaining economic benefits for local people, and sustaining incentives for Government to maintain an active conservation programme. The research associated with the programme is providing new and unpredictable insights into the dynamic nature of Hawksbill populations.

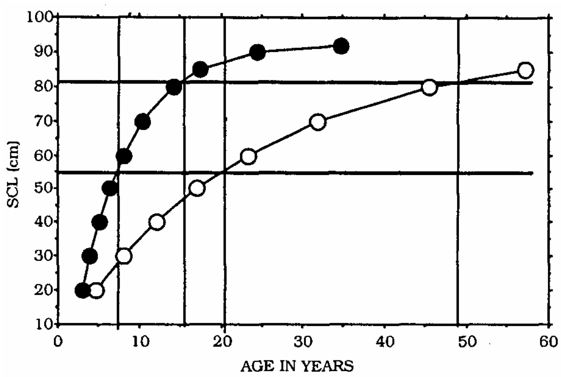


Fig. 9. Mean age-size relationship predicted from the growth rate data on Figure 7 for the combined Cuba-Mexico data (closed circles) and other areas where growth has been measured (open circles). The size (horizontal lines) and mean age (vertical lines) indicate where 5% and 100% of females would be mature in accordance with both relationships.

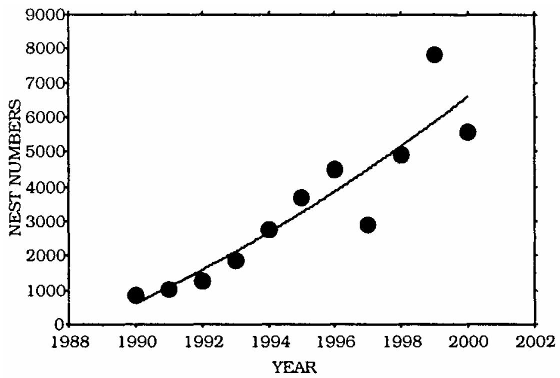


Fig. 10. Trends in Hawksbill Turtle nesting in the Yucatan Peninsula, Mexico, following increased protection of the resource in 1990 (Garduno-Andrade 1999; Garduno-Andrade *et at.* 1999).

The central constraint on the Cuban programme is that opposition to their programme from outside Cuba is strongly biocentric, and attributes little value to whether the harvest is sustainable or not.

POPULATION DYNAMICS VERSUS DYNAMIC POPULATIONS

Wildlife conservation is primarily about managing increases and decreases in abundance, which is not a focus on individual population parameters, but rather the net result of how those parameters interact with each other to produce a final increase or decrease in the population.

All wildlife populations have dynamics which increase (reproduction, immigration) and decrease populations (mortality, emigration), but they are not necessarily constant nor independent of each other. They change with density and interact with each other in unpredictable ways. The only way of knowing with certainty what will happen when a population is manipulated, is to test it. Arguments such as *"we need decades of research before we will understand the populations enough to predict with certainty what will happen if we try to use them"* are usually stalling tactics. The only way of knowing with certainty what will happen is to test it.

Caughley's spring

The late Graeme Caughley likened wildlife populations to a spring. If they are reduced, but their environment is still intact, it is like compressing the spring. The more they are reduced, the greater their potential to recover if given the opportunity. Most wildlife conservation problems today are ones in which the opportunity to recover is denied, because excessive harvesting cannot be controlled or habitats are being lost and the carrying capacity reduced.

When harvest pressures were reduced on Hawksbill Turtles in Mexico (Fig. 10) the population responded with an exponential increase in nesting (Garduno-Andrade *et al.* 1999). Saltwater Crocodiles in the Northern Territory had their populations reduced to less than 5% of original numbers, but retained the ability to recover when given the opportunity (Webb *et al.* 1999, 2000). In neither case could models "predict" what those rates of increase would be. The intrinsic rate of increase of adult nesting Hawksbill Turtles in Mexico between 1990 and 2000 was 0.21 (23% per year).

Harvest theory and extinction

Harvest theory, which has been well tested on many species, basically says that there is a maximum rate at which a reduced population can recover (rate of recovery or increase). The maximum harvest can be extracted and sustained (Maximum Sustainable Yield) when the population is reduced to a level stimulating the maximum recovery in terms of numbers of individuals (e.g., Caughley and Sinclair 1994; Choquenot *et al.* 1998; Erdelen 1998; Butterworth 1999).

Harvesting beyond the MSY, as occurred historically with Saltwater Crocodiles and Hawksbill Turtles in Mexico, does not mean that the population is going to continue declining towards extinction. It simply means that the wild population will stabilize at a smaller level and that the annual offtake will be smaller than would be the case if harvesting was done at MSY.

Totally uncontrolled harvesting will generally reduce a population to a level where the annual harvest is simply too small to make it economically viable to harvest *(commercial* *extinction),* which can occur at levels well above those likely to threaten a species with *biological extinction.* The population level to which crocodiles and sea turtles need to be pushed before they lose the ability to recover, may be much less than 1% of carrying capacity. This is perhaps why they survived the widespread reptilian extinctions of 64 million years ago.

How fragile are wild populations?

The view that wildlife populations are all "fragile" and cannot adapt to changed circumstances, is a common reason for caution with CSU programmes. Reference to intricate ecological food webs adds to the impression that ecological systems are finely balanced, like a house built of playing cards, such that if even one card is removed, the system collapses. Against this, when Saltwater Crocodiles — "a top end predator" — were virtually eradicated, there was no obvious ecological disaster. The system seemed to adapt easily. There is also our inability to make wildlife go extinct, when we try with various pest species. The long history of uncontrolled utilization of crocodilians and sea turtles around the world, suggests they are tenacious survivors (Webb *et al.* 1987b) rather than fragile entities, and that even if experiments with different conservation options do not give the desired results, the knowledge gained is, likely to outweigh costs accrued through unpredictable impacts. Being too conservative and precautionary (Bjorndal 1999), rather than bold and experimental, may not always be in the interests of wildlife conservation.

Models and models

As pointed out by Carrillo *et al.* (1999), caution needs to be exercised when population simulation models, rather than the results of experimental harvests, are used as primary evidence to support or reject CSU programmes. Four concerns are:

1. Simple population models which rely on parameter estimates measured in a stable population, or one at a specific stage of recovery (e.g., Smith and Webb 1985; Congdon *et al.* 1993; Congdon and Dunham 1994; Tucker 1995) largely ignore the dynamic nature of populations, and lead to conclusions about the impacts of harvesting adults that may not prove correct (Webb and Smith 1987; Tucker 1995). Despite model predictions, wild populations of long-live slow-maturing reptiles have coped extremely well with adult harvests (e.g., Joanen and McNease 1987; Woodward *et al.* 1991).

2. The assumption that population structure needs to be stable is rarely supported by data. For example, with US Loggerhead Turtles *Caretta caretta* 498 000 eggs and juveniles were estimated as being "needed" to support a stable adult population of 1 277 individuals (Grouse 1999). To extrapolate this to 996 000 eggs and juveniles being "needed" to support an adult population twice that size, is unjustified: the ratio of juveniles to adults should not be expected to be stable.

3. The relative importance of eggs in simple models tend to be treated as a constant, but in reality, the significance of eggs is perhaps much higher in the early stages of recovery than later. The impacts of harvesting crocodilian eggs can seldom be detected in a recovered population, because survival rates of juveniles adjust to compensate for reduced recruitment (Woodward *et al.* 1991; Webb *et al.* 1999, 2000; Temsiripong *et al.* 2000).

4. Parameters used in a model need to be those related to the population under study, even if they are not as well quantified as those derived from other populations. For example, when trying to model the population dynamics of Hawksbills from Cuba and Mexico, serious errors would be introduced if growth rate parameters from other areas were assumed to apply (Fig. 9) (Heppell *et al.* 1995; Heppell and Crowder 1996; Carrillo *et al* 1998). Clutch size for Saltwater Crocodiles in Papua New Guinea averages 59 eggs (Cox 1985) whereas in Australia it averages 50 eggs (Webb and Cooper Preston 1989). Large Hawksbill Turtltes in Antigua nest 4 times in a season (Richardson *et al.* 1999) whereas those in Mexico and Cuba nest 2.5 times (Garduño-Andrade *et al.* 1999).

In overview, if simple models do not include parameters relative to the population under study, and do not account for density-dependence, compensatory responses, and the dynamic nature of populations, their utility for assessing issues such as "sustainability" may be greatly diminished (Chaloupka and Musick 1997).

Sustaining uses in a recovering population

In many commercial harvest programmes, the aim is to maximize the harvest and the economic benefits derived from it (Tucker 1995; Butterworth 1999). However, a harvest programme can equally be aimed at increasing the value of a species in the eyes of the community, and extracting the maximum harvest may not be needed to do this. Harvests of Saltwater Crocodiles in the Northern Territory, Hawksbill Turtles in Cuba and American Alligators in Louisiana (Joanen and McNease 1987) and Florida (Woodward *et al.* 1991) were clearly well below maximum levels, because the populations continued to expand while the harvests were taking place. In these cases there would have been little utility in waiting until the wild population was fully recovered before initiating a harvest programme, in which the first step would have been to reduce the population to stimulate a faster rate of increase.

Long-lived versus short-lived species

The question about whether longevity is an asset or a liability is confused within the literature. On the one hand it is argued that species which take long periods of time to reach maturity, and then survive for long periods of time after maturity, are difficult to manage (Mortimer 1995). However, it can equally be argued that long-lived species can withstand adverse impacts for long periods of time and still retain the ability to stimulate a population recovery, if given the opportunity. They may thus be far more robust in terms of surviving adverse environmental situations, than shortlived species. It is often implied that sea turtles are particularly vulnerable to extinction because of their marine mode of existence, their migratory habits, their long ages to maturity, etc. Yet common sense dictates that relative to ' terrestrial chelonians of similar size, the marine lifestyle and ability to move are two major buffers against extinction.

CONCLUSIONS

Whether we support "sustainable use" as a conservation tool, whether we understand it, whether we are sceptical of its potential, or oppose its introduction vehemently — it is now impossible to walk away from the conclusion that it is a mainstream conservation strategy and it deserves to be investigated closely. If, as I have argued, it can be an effective way of increasing the value of wild species to a community, so it has the potential to stimulate conservation action at the local level, where it is needed most. We should be able to look at the CSU approach to conservation objectivity, and judge its worth on results — not on predictions.

It has become customary to think of those who promote sustainable use as being rather insensitive beings, who favour big business over aesthetic values. There is little that can be done about this: shooting the messenger is often easier than heeding his or her message. But it is important to establish unequivocally that sustainable use has been an important element of some successful conservation programmes. With Saltwater Crocodiles in the Northern Territory the calls for culling in 1979-80 threatened the conservation programme in its entirety. Introducing programmes with economic incentives proved extremely effective in winning back public support. This is not a hypothesis, it is fact.

In Cuba, there is no doubt that their current traditional harvest of Hawksbill Turtles is sustainable, is providing benefits to local people, is providing incentives for Government to keep investing in sea turtle conservation, and that the results of monitoring and research programmes linked to the harvest are important for the conservation of Hawksbills outside Cuba.

If there is a single closing message, it is that our ability to understand, define and solve conservation problems will continue to be constrained if we:

1. Approach conservation purely from a biocentric viewpoint;
2. Ignore the role CSU can and is playing; and,
3. Ignore the dynamic and adaptable nature of wildlife populations.

APPENDIX 1

IUCN Resolution (2.16) Agreed at World Conservation Union meeting Amman, Jordan October 2000.

Policy Statement on Sustainable Use of Wild Living Resources.

1. Conservation of biological diversity is central to the mission of IUCN, and accordingly IUCN recommends that decisions of whether to use, or not to use, wild living resources should be consistent with this aim.
2. BOTH CONSUMPTIVE AND NON-CONSUMPTIVE use of biological diversity are fundamental to the economies, cultures, and well being of all nations and peoples.
3. Use, if sustainable, can serve human needs on an ongoing basis while contributing to the conservation of biological diversity.
4. At its session of the General Assembly (Perth, 1990) in Resolution 18.24, IUCN — The World Conservation Union recognized that "the ethical, wise and sustainable use of some wildlife can provide an alternative or supplementary means of productive land-use, and can be consistent with and encourage conservation, where such use is in accordance with appropriate safeguards".
5. This position was re-affirmed in Resolution 19.54 at the following, session of the Union's General Assembly in 1994 and subsequently in Resolution 1.39 at the 1st meeting of the World Conservation Congress in 1996.
6. Analyses of uses of wild living resources in a number of different contexts demonstrate that there are many biological, social, cultural, and economic factors, which combine in a variety of configurations to affect the likelihood that a particular use may be sustainable.

7. On the basis of these analyses, IUCN concludes that:

a) Use of wild living resources, IF sustainable, is an important conservation tool because the social and economic benefits derived from such use provide incentives for people to conserve them;

b) When using wild living resources, people should seek to minimize losses of biological diversity;

c) *Enhancing the sustainability of uses of* wild living resources involves an ongoing process of improved management of those resources; and

d) Such management should be adaptive, incorporating monitoring and the ability to modify management to take account of risk and uncertainty.

8. To increase the likelihood that any use of a wild living resource will be sustainable requires consideration of the following:

a) The supply of biological products and ecological services available for use is limited by intrinsic biological characteristics of both species and ecosystems, including productivity, resilience, and stability, which themselves are subject to extrinsic environmental change.

b) Institutional structures of management and control require both positive incentives and negative sanctions, good governance, and implementation at an appropriate scale. Such structures should include participation of relevant stakeholders and take account of land tenure, access rights, regulatory systems, traditional knowledge, and customary law.

c) Wild living resources have many CULTURAL, ETHICAL, ECOLOGICAL AND ECONOMIC values, which can provide incentives for conservation. Where an economic value can be attached to a wild living resource, perverse incentives removed, and costs and benefits internalized, favorable conditions can be created for investment in the conservation and the sustainable use of the resource, thus reducing the risk of resource degradation, depletion, and habitat conversion.

d) Levels and fluctuations of demand for wild living resources are affected by a complex array of social, demographic, and economic factors, and are likely to increase in coming years. Thus attention to both demand and supply is necessary to promote sustainability of uses.

9. IUCN is committed to *ensuring any uses of wild living resources are equitable and ecologically sustainable,* and to this end it has established the Sustainable Use Initiative which incorporates regionally-structured Specialist Groups of the Species Survival Commission to:

a) Identify, evaluate, and promote the principles of management that contribute to sustainability and enhanced efficiency in the use of wild living resources; and

b) Regularly communicate their findings to members and the broader community.

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