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HUMAN USE
OF TURTLES

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 ← EDITORS

sightings of individual bog turtles over a 20-year period, the health and viability of bog turtle populations throughout much of their range had plummeted over this same period (see "Consider Ecological and Population Issues across a Broad Range of Geographical Scales," Klemens, Chapter 10). In the most curious of ironies, a system that was supposed to inform conservation decision making actually worked against conservation, obscuring a rangewide population crash of an endangered species.

Truly effective programs to conserve chelonians require integration of activities and data at several scales, including data on the activities of local human populations. Protection and conservation of turtle populations and species must involve creative biologists who know the animals and their habitats and who are willing to work with local citizens and governments.

There is a strong need for more good science in turtle conservation. We are convinced that conservation efforts on behalf of turtle populations must be conducted by viewing turtles as components of larger systems and with a greater knowledge of the variables in these systems that affect turtle populations. Frazer (1992) pointed out very well that the headstarting of sea turtles, without concurrent efforts to guarantee the health of the marine environment into which they were released, constituted "halfway technology." In a similar fashion, protection of single populations or isolated habitats without consideration of the ecosystems in which they are embedded is just another form of halfway technology that may ultimately doom many turtle species to extinction.

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2000 CHAPTER IN TURTLE CONSERVATION

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2 HUMAN USE OF TURTLES

A Worldwide Perspective

Because they are an easily captured and conveniently stored source of protein, turtles have been an important human food item for millennia. To varying degrees, rural people throughout the world depend on turtles and their products to meet subsistence needs; now more and more people are catching turtles to trade or sell them. As the paragons of long-lived animals, many cultures have imbued turtles with special medicinal or religious qualities that have promoted, or in some cases prohibited, their consumption. The collection of turtles to keep or sell as pets is also a significant threat for certain species. Throughout the world increasing pressure is being placed on wild populations of turtles to meet a variety of demands from growing human communities. The effects of human exploitation on wild turtle populations have not been well quantified, but it is clear that in many cases human use is the principal cause of turtle population declines and, in some cases, extinction (Klemens and Thorbjarnarson 1995). Understanding patterns in the human use of turtles is vital for developing rational conservation and management plans for chelonians.

Certainly foremost among the human uses of turtles is their use as food. Harrisson (1962a, b, 1967) reported the discovery of what appeared to be green turtle (*Chelonia mydas*) bones found in excavations in the Niah Caves of Borneo, providing evidence that sea turtles may have been an important food source for early humans. Levels of prehistoric use of turtles are hard to gauge, but there is evidence that exploitation by human populations could have played a role in the extinction of turtles on some islands and in some mainland habitats (Moodie and Van Devender 1979). Relatively recent human colonization of the Mascarene Islands, off

Madagascar, almost certainly caused the extinction of six species of tortoises (Arnold 1980) and local extinction of sea turtles (Frazier 1982a; Hughes 1982).

Of the endangered chelonian taxa listed in the International Union for Conservation of Nature and Natural Resources' *IUCN Amphibia–Reptilia Red Data Book* (Groombridge and Wright 1982), human consumption for food is the principal factor contributing to population declines in 46% of the taxa, and it is a cofactor in an additional 20% (Klemens and Thorbjarnarson 1995). Patterns of human use are frequently seasonal and depend to a large extent on the biology of the targeted species. Some species, such as large river turtles (South American river turtles [*Podocnemis* spp.], the Central American river turtle [*Dermatemys mawii*; Central America and Mexico], the Madagascan big-headed turtle [*Erymnochelys madagascariensis*], the pig-nosed turtle [*Carettochelys insculpta*; New Guinea], and the Indo-Malaysian river terrapin [*Batagur baska*], painted terrapin [*Callagur borneoensis*], Malaysian giant turtle [*Orlita borneensis*], painted roofed turtle [*Kachuga kachuga*], and Burmese roofed turtle [*Kachuga trivittata*]), softshell turtles, and the green turtle, are threatened principally by excessive exploitation for food. Because of their ability to survive extended intervals with minimal care, turtles can be kept for long periods before they are consumed, an attribute of considerable importance where refrigeration is not available. The eggs and adult females of colonial-nesting species are particularly vulnerable to human exploitation, and this vulnerability has been well documented in marine and river turtles worldwide.

Turtles have been, and continue to be, used for many purposes besides food. The commercial trade in highly valued commodities such as tortoiseshell from hawksbill turtles (*Eretmochelys imbricata*) and leather from olive ridley turtles (*Lepidochelys olivacea*) has fueled worldwide hunting of these species. In certain cultures demand for turtles has been associated with medicinal or religious uses. The carapaces of large species have been used for a variety of purposes, including wash basins, roofing material, canoe paddles, and shields.

For centuries, land turtles have been kept as pets around the world, but the pet market exploded after World War II in Europe, the United States, and Japan (HSUS 1994). The bulk of this expanding market was for hatchling aquatic turtles (especially red-eared sliders [*Trachemys scripta elegans*]), with turtle ranches (open-cycle operations in which adults, hatchlings, and eggs may be taken from wild populations to augment the captive breeding population) in the United States producing millions of hatchlings annually in the 1960s (Warwick 1986). There has also been an increased demand for a variety of other turtle species for pets, primarily pond turtles and tortoises. Hundreds of thousands of tortoises of several species (spur-thighed tortoise [*Testudo graeca*], Central Asian tortoise [*T. horsfieldii*], and Hermann's tortoise [*T. hermanni*]) were imported into Europe to meet this demand, in addition to smaller numbers of a wide variety of other species (Fitzgerald 1989; Smart and Bride 1993). This trade has caused population declines for widespread species (e.g., the spur-thighed tortoise in Morocco and common box turtle [*Terrapene carolina*] in the United States) as well as local extirpations, such as that suffered by the Egyptian tortoise (*Testudo kleinmanni*) in Egypt (Baha el Din 1994). Concern over the exploding demand for turtles as pets led to the first international controls over the trade in turtles in the 1970s (Fitzgerald 1989), but significant trade continues, in many cases simply shifting from protected to unprotected species.

In this chapter we will summarize information on use of turtles by humans on a regional basis according to two major themes: use for food, medicines, and other products and use as pets. The first section is subdivided into coverage of tortoises and freshwater turtles and then of marine turtles. With few exceptions, most notably marine turtles, little quantitative information is available on levels of exploitation and the effects of exploitation on turtle populations. Also, as the use of turtles by humans is so widespread, it is beyond the scope of this chapter to provide a comprehensive review (see, however, additional discussions in chapters 4 through 7 of this volume). Instead, we concentrate on regional overviews of representative cases. To provide a more detailed account of the human use of certain species, we present a number of case studies. We close by examining some of the biological and cultural factors that affect the exploitation of turtle populations and the implications for management programs based on sustainable use.

USE OF TURTLES FOR FOOD, MEDICINES, AND OTHER PRODUCTS

Tortoises and Freshwater Turtles

EUROPE AND NORTHERN ASIA

The use of turtles as a food resource in Europe and northern Asia is relatively minor. In most areas turtle populations are small, and there is a lack of a cultural tradition for the use of chelonians as food. However, in Bulgaria both the Asia Minor subspecies of the spur-thighed tortoise (*Testudo graeca ibera*) and Hermann's tortoise are consumed, particularly in areas where tortoises are relatively abundant. Before World War I, the collection of tortoises was mostly restricted to an area near Plovdiv. During World War II, the sale of tortoises became more common, and many were sent to Germany or to restaurants in the interior of the country (Sofia, Varna, and Burgas). Some estimates suggest that up to 90% of some local tortoise populations were collected at this time. Today, approximately 35 to 40% of Bulgarians have consumed tortoise. Over the past two decades, blood, meat, eggs, and other products have been used as "cures" for cancer and leukemia (Beshkov 1993).

NORTH AMERICA

Although the consumption of turtles is frequently associated with tropical, developing countries, it is still common in parts of the rural United States. In the early twentieth century, the consumption of diamondback terrapin *(Malaclemys terrapin)* contributed to the species' precipitous decline. In the mid-Atlantic states of Maryland, Virginia, Pennsylvania, and New Jersey the use of snapping turtle *(Chelydra serpentina)* in the preparation of soup is still widespread (Babcock 1919; King 1978; Klemens 1993a). In the southeastern United States the alligator snapping turtle *(Macroclemys temminckii)* is currently threatened by collection for food (Pritchard 1989; Sloan and Lovich 1995). In the 1960s and 1970s, commercial trapping of this species was intense in Mississippi, Georgia, Louisiana, Alabama, and Texas, and the meat was used domestically in a commercially produced turtle soup. Reports indicate that populations in Louisiana have been reduced to a level at which commercial trapping is no longer a viable activity (USFWS 1996a).

A significant trade in red-eared sliders for food exists in the southern United States. Commercial trade in wild-caught specimens for pets began around 1950 and later led to the establishment of commercial ranches (Warwick 1986). Most individuals exported from the United States go to the Far East, but the breakdown between animals being used as food versus sold as pets is unclear. Even small red-eared sliders are reportedly used for stews or fried. In the late 1980s, one turtle ranch estimated that during the collecting season 25,000 to 30,000 wild-caught turtles were sold every 1 to 2 weeks, almost all for human consumption. Most were shipped to California to supply the large Chinese population there. Warwick and Steedman (1988) reported that large numbers of adult turtles are shipped overseas from San Francisco. Shipments included North American softshell turtles *(Apalone* spp.), map turtles (*Graptemys* spp.), and, most commonly, species of sliders *(Trachemys)* and cooters *(Pseudemys)*. Up until about 1980, many turtle ranches purchased adult turtles for resale as food. Thereafter, buyers purchased turtles directly from professional hunters and dealers (Warwick and Steedman 1988). North American softshell turtles are consumed locally in the southern and central United States. Small, live animals and the meat from larger ones are sold to Chinese-American markets or exported, mostly to Japan, Hong Kong, and China. In 1993, 8,107 kg of North American softshell turtle meat were exported from the United States, and in 1994, 34,467 live North American softshell turtles were exported (USFWS 1996b).

In the southeastern United States the use of the gopher tortoise *(Gopherus polyphemus)* as food has a history that stretches back more than 4,000 years. More recently, it was an important food source during the Great Depression of the 1930s (Diemer 1989). Use of this species for food continues (Moler 1992).

SOUTH AND CENTRAL AMERICA AND THE CARIBBEAN

Throughout Central America turtles are frequently used as food. E. O. Moll and Legler (1971) found that in Panama subsistence hunters would harpoon common sliders *(Trachemys scripta)* at night, as they slept at edges of floating grass mats, and collect adult females as they emerged to nest. Mora and Ugalde (1991) reported that eggs and adults of common sliders were hunted by Guatuzo Indians and settlers in the Caño Negro Reserve in northern Costa Rica. Pritchard (1993) reported that a ranching program had been initiated at this site; eggs are collected and incubated, and the hatchlings are sold as pets in San José. Local egg collectors receive 50% of the proceeds. In northern South America the eggs and adults of common sliders are widely sought as food (Pritchard and Trebbau 1984; Rodriguez and Rojas-Suárez 1995). In Haiti the Hispaniolan slider *(Trachemys decorata)* is collected for food and sold in local markets. Consumption of turtle meat by children is believed to keep supernatural beings *(lutgaru)* from drinking the blood of the children (J. Thorbjarnarson, personal observation).

Along the Caribbean coast of Mexico, Belize, and Guatemala, the large Central American river turtle is much sought after as a food item (Alvarez del Toro et al. 1979). In Belize these turtles are collected using a variety of techniques, including harpooning, free diving, and netting, and are principally sold in local markets. Population size and structure and capture rates were used to assess the effects of exploitation on wild populations; data suggested that in the principal harvest areas exploitation levels were not sustainable (Polisar and Horwich 1994; Polisar 1995).

Throughout the Amazon and Orinoco river basins, river turtles (family Pelomedusidae) are preferred food items in riverine communities (Brito and Ferreira 1978). Historic use by indigenous groups throughout the Amazon and Orinoco river basins centered on the largest species, the giant South American river turtle *(Podocnemis expansa)* (N. J. H. Smith 1979; see "Giant South American River Turtle: Change in a Traditional Exploitation System," this chapter). With the depletion of populations of this species, human use has shifted to smaller species, particularly the yellow-spotted Amazon River turtle *(Podocnemis unifilis)*, the red-headed Amazon River turtle *(Podocnemis erythrocephala)*, the six-tubercled Amazon River turtle *(Podocnemis sextuberculata)*, and the big-headed Amazon River turtle *(Peltocephalus dumeriliana)* (Mittermeier 1975; Alho 1985; Ojasti 1995; Thorbjarnarson et al. 1997). In recent years the influx of large numbers of miners to the upper Orinoco region has resulted in considerable impact on local populations of the big-headed Amazon River turtle and other turtles in Venezuela (E. Seplaki, personal communication) as well as the red-headed Amazon River turtle in Colombia (O. Castaño-Mora, personal communication). Turtles continue to be highly valued as food and are widely hunted (Pritchard and Trebbau 1984; O. Castaño-Mora, personal communication; R. Vogt, personal communication).

All four species of South American giant tortoises (*Geochelone* spp.) have a long history of use by humans. Tortoises were an important source of protein in pre-Columbian times and continue to be widely consumed by many indigenous groups (Werner 1978; Mittermeier 1991; Vickers 1991). The widespread presence of *Geochelone* spp. throughout the Lesser Antilles has been attributed in part to dispersal by Amerindians who used them for food (A. Schwartz and Henderson 1991). In addition to their importance as food, tortoises were used ritualistically in some societies, as among the Ka'apor of northeastern Brazil: tortoise meat was traditionally eaten by girls at puberty and by menstruating women (Balée 1985). Since the European colonization of South America, the use of tortoises as food has become extremely seasonal, limited largely to the Holy Week prior to Easter. During this time the Catholic Church prohibits the consumption of meat but conveniently classifies tortoises as fish (Pritchard and Trebbau 1984; Rodríguez and Rojas-Suárez 1995). The collection of tortoises for food can be a particular problem for the red-footed tortoise (*Geochelone carbonaria*) because it frequently uses mixed woodland–savanna habitats, where it can be easily found and collected. The tragic history of the exploitation of the Galápagos tortoise (*Geochelone nigra*) is presented in more detail below as a case study.

AFRICA

Compared with Central and South America, the use of turtles for food is relatively minor in Africa. Pritchard (1979) noted that the large African softshell turtle (*Trionyx triunguis*) is widely used for food, but little is known about this use. Two species of *Cycloderma* and two species of *Cyclanorbis*, all African flapshell turtles, can attain large sizes, but little is known about levels of human exploitation. Pritchard (1979) reported that both species of *Cyclanorbis* were eaten by Bari tribesmen.

The largest African tortoise, the African spurred tortoise (*Geochelone sulcata*), is not consumed widely because it is found principally in Moslem areas where religious taboos prohibit eating turtle flesh (Broadley 1989b). However, Lambert (1993) reported that in Mali this species was collected for food and for shipment overseas. Small, easily transportable specimens are usually taken within 10 km of villages for consumption. The Yoruba people of southwestern Nigeria use hinge-back tortoises (*Kinixys* spp.) for food and medicinal purposes (J. A. Butler and Shitu 1985). The head and intestines are prized as medicine and used to treat cholera and burns and to prevent the death of children due to supernatural causes. Klemens (1992) reported that the pancake tortoise (*Malacochersus tornieri*) is consumed by the Hadza people in the Kidero Mountains of Tanzania. The leopard tortoise (*Geochelone pardalis*) of sub-Saharan Africa is commonly used as food (Broadley 1989c; M. W. Klemens, personal observation). In Madagascar consumption of radiated tortoises (*Geochelone radiata*) is prohibited by tribal taboos in some regions, but it is considered a delicacy in urban markets (Juvik 1975; Durrell et al. 1989b).

Large river turtles, which can be an important food source for people in South America and parts of Asia, are not found in most parts of Africa (Iverson 1992a). In Madagascar the Madagascan big-headed turtle is regularly caught in fishing nets set in lakes, and local populations can be rapidly extirpated by fishers (Kuchling and Mittermeier 1993). A major factor contributing to overexploitation is the structure of the habitat: small, shallow, open lakes that are easily fished. Populations appear to be more depleted in areas where seine nets are used as opposed to hoop nets or lines. Subsistence hunting by local communities does not appear to be a problem in the areas surveyed by Kuchling and Mittermeier. Commercial hunting is closely associated with fishing, however, and turtle populations seem more depleted in areas near roads where fish can be marketed fresh as opposed to more remote areas, where fish are salted. Populations of the Madagascan big-headed turtle are reported to be declining as hunting pressure increases due to a growing human population and expanding inland fisheries.

INDIAN SUBCONTINENT

Choudhury and Bhupathy (1993) reported that in India 22 of 26 species of turtles were exploited either commercially or on a subsistence level. Turtles were sold in 12 of 61 markets surveyed, with the principal species offered being softshells: Indian flapshell turtle (*Lissemys punctata*), Indian softshell turtle (*Aspideretes gangeticus*), and Indian peacock softshell turtle (*Aspideretes hurum*). The preference for softshell turtles as food can be related in part to taste, in part to the large size of some species, and in part to the ease of butchering them (E. O. Moll 1990a). The larger softshell species, weighing up to 20 kg (e.g., Indian softshell), are preferred and bring a high price. Other species, such as the crowned river turtle (*Hardella thurjii*) and the three-striped roofed turtle (*Kachuga dhongoka*), are occasionally seen in markets. E. O. Moll (1990a) reported that human consumption of turtles in India is greatest in West Bengal. Howrah (Calcutta) is the major marketing center for turtles from the Ganges and Mahanadi river basins. At this one market, a survey in 1981 and 1982 estimated the annual trade was 50,000 to 75,000 small trionychids (Indian flapshell turtle), 7,000 to 8,000 large trionychids (Indian softshell, Indian peacock softshell, and narrow-headed softshell turtle [*Chitra indica*]), and 1,000 to 1,500 emydids (Vijaya and Manna *in* E. O. Moll 1990a). Softshells were present in 32 of 35 markets or villages where E. O. Moll (1990a) found turtles for sale. The sale of turtles and tortoises is reported to be considerable in the Himalayan foothills of Nepal (Shrestha 1997).

Turtle shells are widely used for medicinal purposes throughout India, usually as a by-product of eating the meat. In Uttar Pradesh turtle shells are sold for approximately US$0.15 each to manufacturers of combs and brushes. (Within this chapter, all monetary amounts are given in U.S. dollars.) Ground turtle shell, particularly that of softshells, is used for the treatment of eye allergies and the meat to relieve the symptoms of tuberculosis (Hanfee 1995). In Assam and West Bengal the consumption of turtle meat is believed to have medicinal value.

In Bangladesh, although the consumption of turtle meat is prohibited under Islamic law, the export of turtles is not (Das 1990). Since the late 1970s, a large export market has developed for two species of large softshell turtles, the Indian softshell and Indian peacock softshell, which are shipped to Hong Kong, Malaysia, South Korea, and Japan (Das 1990). The smaller Indian flapshell turtle is exported to India. Large numbers of animals are kept in holding facilities in Dhaka prior to shipment; the trade peaks during the winter months. Most turtles are exported alive, but meat, eggs, cartilage, and turtle oil are also exported. Das (1990) indicated that in many areas turtle populations still appear to be viable; however, continued, uncontrolled exploitation will threaten the resource. Sarker and Hossain (1997) reported that the value of turtles exported from Bangladesh is in excess of $600,000 annually and wild populations are in rapid decline.

SOUTHERN ASIA

Southern Asia represents the largest regional market for freshwater turtles and tortoises in the world. In part this is due to cultural beliefs regarding the health benefits of eating turtle meat and the use of turtles as medicines. What was once a domestic trade has now become a large-scale business, mostly with mainland China (Jenkins 1995). In some countries (e.g., Thailand and Vietnam) turtle populations appear to have been significantly affected, although a lack of data on the status of turtle populations in these areas hampers detailed analysis. Increased trade with China appears to be related to recent changes in currency convertibility, the rapid economic growth of southern China, and the increased demand for traditional Chinese foods and medicines (Li Wenjun et al. 1996). Softshell turtles are the most widely used group; however, trade in tortoises and emydid turtles is also important. Certain species, particularly the large river turtles, the river and painted terrapins, have been significantly affected by exploitation, traditionally for eggs but increasingly for meat.

Throughout Southeast Asia, high human population densities and habitat loss have severely affected turtle populations. In many areas habitat loss has historically been compounded by exploitation of turtles for subsistence purposes. Over time subsistence use shifted to the commercialization of turtles in local markets. Today, the import and export of turtles for food and medicinal purposes is becoming increasingly important, and a large and complex trade in turtles has emerged. This trade deserves special mention due to the significant impacts it is presumably having on wild populations.

Jenkins (1995) noted that patterns of trade in turtles in Southeast Asia have changed dramatically in recent years, primarily due to increased commercial demand by mainland China. Large numbers of turtles from throughout Indochina are shipped through Vietnam to supply that demand. Within China turtles are used for a wide variety of medicinal purposes and for food. The level of trade is hard to quantify because much of it is illegal, but anecdotal evidence suggests that in many areas turtle populations are severely affected. Softshell turtles are the most sought-after food species, and evidence suggests that this trade is a huge regional business and growing. There are a large number of farms for the Chinese softshell turtle *(Pelodiscus sinensis)* in eastern China, but most turtles sold in restaurants are wild caught because these are larger and less expensive than are farm-reared animals (Cen Jianqiang, personal communication). Most softshell turtles originate from outside China and are sold in markets in southern China (e.g., Yunnan and Guangxi). Aside from the softshells, many other turtle and tortoise species are traded for their shells, which are used for medicinal purposes. Some of the larger river turtles (especially the river and painted terrapins) are widely consumed and vulnerable to overexploitation.

A large regional export market exists in Indochina. In Thailand all native species of softshells have been intensively exploited, and evidence suggests that populations have declined as a result (Jenkins 1995). Thirakhupt and van Dijk (1997) concluded that subsistence use of turtles has severely depleted the populations of most species in unprotected areas of western Thailand. Most collecting now is opportunistic because it is no longer economically viable to hunt specifically for turtles. Within Thailand, however, trade still exists, and the Asiatic softshell turtle *(Amyda cartilaginea)* is the most important species in economic terms. Throughout the country the small-scale trade of turtles for local consumption or to supply restaurants and turtle breeding farms is common. Breeding farms in this region appear to concentrate on the exotic Chinese softshell turtle. In Thailand the trade in other turtle species appears to be much less important than is the trade in softshells. In Myanmar, even though trade in turtles is illegal, they are viewed as an essential part of the diet and are consumed locally or traded over the border in Thailand and China, where native populations are more depleted (Jenkins 1995).

A large and expanding illegal trade in softshells has resulted in turtles being shipped from Laos to Vietnam (Jenkins 1995), with the final destination for many of these turtles being China. Jenkins (1995) reports that in some areas the commercial sale of turtles began as late as 1994. Previously, turtles had been consumed locally. Some of the turtles traded in Vietnam reportedly originate in Cambodia, which has a widespread system of collecting tortoises (particularly the elongated tortoise [*Indotestudo elongata*]). An estimated 2 to 4 metric tons of turtles are exported daily from Phnom Penh, principally to Vietnam (E. B. Martin and Phipps 1996).

Within Vietnam a highly organized turtle trade exists, involving at least 17 of the 21 known native species. Regional turtle collection points exist throughout the country. Also, changes in the country's economic system have opened Vietnam to foreign markets. Within Vietnam trade passes through Ho Chi Minh City or Hanoi, and an estimated 90% of the turtles go to China. Total trade of all turtle species in Vietnam has been estimated at 200,000 individuals annually but may be much higher (Le Dien Duc and Broad 1995).

In Sumatra softshell turtles are commonly eaten, particularly by ethnic Chinese. Many turtles are also exported. In 1988, 66,500 kg of the Asiatic softshell and 37,000 kg of the Malayan box turtle (*Cuora amboinensis*) were exported from Sumatra (P. van de Bunt, unpublished). Other freshwater turtles and tortoises are consumed to a much lesser degree (P. van de Bunt, unpublished). Recent reports suggest that the trade of turtles for food is growing and spreading throughout Indonesia (Jenkins 1995).

In Peninsular Malaysia the Asiatic softshell and Malayan box turtle appear to be the most heavily exploited species. Softshell turtle eggs are also eaten when found (Jenkins 1995). Historically, a large industry sprang up around the colonial-nesting sites of river terrapin along the Perak River (E. O. Moll 1987 *in* Jenkins 1995; E. O. Moll 1989a; see "Habitat Alteration," E. O. Moll and Moll, Chapter 5). Traditional egg harvest rights were owned by the Sultan of Perak, who had guards protect beaches during the nesting season, collected all the eggs from the first two nesting events, and left eggs from the last nesting to hatch naturally. A somewhat similar system was reported for the Sungei Muda and Sungei Kedah Rivers in Kedah State, Malaysia. The population of river terrapin in the Perak apparently did not begin to decline until World War II, when adult river terrapin were killed for food during the Japanese occupation (Siow and Moll 1982). After World War II the system controlling the harvest of eggs changed when permits were sold to egg collectors with the stipulation that one-third of the eggs go to the Sultan and one-third be reburied (E. O. Moll 1987 *in* Jenkins 1995). However, few eggs were reburied, and by the 1960s annual egg harvests were 20,000 to 30,000 (Siow and Moll 1982), down from a pre–World War II level of 375,000 to 525,000 eggs. Licenses are still issued for collecting, the eggs sold either to the government hatchery (for release) or on the open market (Jenkins 1995). A similar system is in effect for the collection of eggs of painted terrapin from ocean beaches along the east coast of Peninsular Malaysia. In recent times the consumption of river terrapin meat in Peninsular Malaysia has varied along cultural lines. Indigenous groups and

Chinese and Indian ethnic groups eat the meat, whereas the Islamic Malaysians do not (E. O. Moll 1976).

Throughout southern Asia tortoises are also consumed. Das (1986, citing Blyth 1863) reports the Burmese star tortoise (*Geochelone platynota*) was very highly sought after as a food item. Other species, including the Travancore tortoise (*Indotestudo forstenii*), elongated tortoise, and Asian brown tortoise (*Manouria emys*), are also important food resources.

AUSTRALIA AND OCEANIA

In Australia the exploitation of turtle populations for food is not considered to be a significant problem, although the pig-nosed turtle is eaten by Aborigines. In New Guinea, however, the greater availability of boats with outboard motors has resulted in increased exploitation of pig-nosed turtle populations for meat and eggs (Georges and Rose 1993). Here, female pig-nosed turtles are collected on nesting beaches, and eggs are located by probing the sand beaches with sticks or spears. In some cases pitfall traps are used to capture adults (Groombridge and Wright 1982). During the non-nesting season, this species is captured by hand, on baited lines, or in basket traps. Harvesting of adults and eggs was considered the principal threat to this species in southern New Guinea, and populations were reported to have declined significantly between 1960 and 1980 (Groombridge and Wright 1982).

In Papua New Guinea the Asian giant softshell turtle (*Pelochelys bibroni*) is consumed on a subsistence basis and sold in local markets. Due to its large size this species is an important dietary item. This species is also highly sought for its carapace, which is used for decorative purposes, including ceremonial masks (Rhodin et al. 1993).

Marine Turtles

A considerable volume of information exists, both historical and recent, on human exploitation of sea turtles. Although there is a substantial body of evidence reflecting the human use of sea turtles, these reports offer only brief snapshots of patterns of exploitation over time. Coverage is most complete since the 1960s.

ATLANTIC OCEAN

Probably the first mention by Europeans of sea turtles as a food source was during the discovery of the Cape Verde Islands in 1456 (Cadamosto 1937 *in* Parsons 1962). In the late 1400s and early 1500s, the French and Portuguese sent those afflicted with leprosy and syphilis to the Cape Verde Islands to be cured by eating fresh turtle meat. In addition, those afflicted with leprosy would rub affected areas of their skin with turtle blood (Simmonds 1885, Fontoura da Costa 1939, and Villiers 1958 cited *in* Parsons 1962). More recently in the Cape Verde Islands, hawksbills have been exploited for their shells, and stuffed specimens have been sold to tourists (Maigret 1977 *in* Brongersma 1982). In addition, D. Graff (unpublished) reported that eggs of all four species found in the region (olive ridley, green turtle, hawksbill, and leatherback turtle *[Dermochelys coriacea]*) were harvested.

Due to its geographic location between Europe and the West Indies, and its once-large assemblage of nesting and foraging green turtles, Parsons (1962) suggested Bermuda as the site where commercial turtling began. By 1620, however, only 8 years after permanent English settlement, there was so much concern over the extirpation of green turtles that the Bermuda Assembly passed legislation for their protection (Garman 1884 *in* Carr 1952). In spite of this legislation, within 150 years green turtle populations around Bermuda were so depleted that boats sailed to the Bahamas and Ascension Island in search of turtles (Parsons 1962). Carr (1954) suggested that Bermuda was the first-documented green turtle rookery to be extirpated.

In 1671, Bahamian officials prepared legislation that would protect green turtles against high exploitation levels; however, no action was taken (Great Britain Public Record Office 1889, 1893, 1898). By the 1700s, green turtle populations were severely depleted in the region, and boats traveled north to Florida to harvest turtles (Carr 1954).

In the 1660s, the Cayman Islands were settled by English from Jamaica. These fishermen were renowned for their turtling skills, possibly learned from the Miskitu Indians of Nicaragua during previous contact (Parsons 1962). By 1688, 40 boats were engaged in transporting green turtles from the Cayman Islands and south cays of Cuba to Jamaica (Sloane *in* Lewis 1940). Trade between the West Indies and London began in the mid-1700s (Parsons 1962). By 1802, green turtle populations around the Cayman Islands had become so depleted that the islanders took turtles from Cuban waters. When these waters were depleted, Cayman Islanders moved on to the Gulf of Honduras and then to the Miskito Cays of Nicaragua (Lewis 1940; Carr 1954; Parsons 1962). Today, green turtles no longer nest on the Cayman Islands and are rarely found in the surrounding waters.

The documented exploitation of green turtles throughout the Caribbean spans over 400 years. Carr (1954) credited the combined characteristics of the green turtle as making it the single most important resource that opened up exploration into the Caribbean and supported colonization, buccaneering, and naval operations in the region. He describes the species as big, abundant, available, herbivorous, savory, tenacious of life, air breathing, and easy to catch with simple equipment in shallow water or, easier still, on the nesting beaches. The green turtle provided the colonists with a continuous source of readily available meat.

In the eastern United States sea turtle fisheries developed in Florida, Georgia, Louisiana, Mississippi, North Carolina, Texas, and Virginia (Ingle and Smith 1949; Rebel 1974; Cato et al. 1978). In Florida, Georgia, and the Carolinas turtle eggs, almost exclusively loggerhead turtle *(Caretta caretta)* eggs, were in demand because of their excellent qualities for baking (Caldwell and Carr 1957; Cato et al. 1978). Carr and Ingle (1959) speculated that prior to the arrival of Seminole Indians and Europeans, Florida was the site of large assemblages of nesting green turtles. In the Dry Tortugas, the green turtle rookery was extirpated within 100 years of the initiation of commercial exploitation (King 1982).

Ingle and Smith (1949) reviewed the annual take of turtles from a number of southern states at the onset of the twentieth century. In general, from Texas to North Carolina, the turtle fishery was in decline. In Florida the turtle fishery consisted of the green turtle, Kemp's ridley turtle *(Lepidochelys kempii)*, and loggerhead turtle; the green turtle was the most valuable as a food resource (Caldwell and Carr 1957). By the late 1950s, the Florida sea turtle fishery was confined to the Gulf Coast, and by then only small, immature green turtles were captured, almost entirely for local markets (Caldwell and Carr 1957; Caldwell 1960).

As early as 1519, in the area of the Bay of Campeche along the Atlantic coast of Mexico, Spaniards encountered Indians carrying turtle shell shields (Díaz del Castillo 1908). In 1554, sea turtles were used as a form of payment to the Spaniards (Archivo General de la Nación 1952 *in* Parsons 1962). In recent times Mexico has had one of the largest sea turtle fisheries in the world. Target species were the green turtle and olive ridley on the Pacific coast and the loggerhead and green turtle on the east coast (Cato et al. 1978). The Kemp's ridley is the most endangered of the seven species of sea turtles; one of the principal contributing factors has been the overharvest of eggs (Pritchard and Márquez M. 1973; Ross et al. 1989; Márquez M. 1994).

Along the Atlantic coast of Central America sea turtle eggs are eaten and turtles are hunted for their meat. In the early 1960s, sections of the 35-km nesting beach at Tortuguero, Costa Rica, were leased for egg collection (Parsons 1962). The 1970s decline in nesting females at Tortuguero was attributed to this harvest (Carr 1984).

Miskitu Indians on the Caribbean coast of Nicaragua have long been known for their turtle hunting skills (Parsons 1962). Nietschmann (1973, 1979a) studied patterns of resource use by a Miskitu–Creole community in the late 1960s and early 1970s, documenting changes in Miskitu society as it moved from a subsistence to a cash-based economy. As the international demand for green turtle products increased, turtle hunters sold more of their harvest and returned home with fewer animals to share among family and community members. In one coastal community Nietschmann (1973) documented a 228% increase in the annual turtle harvest and a 1500% increase in the sale of turtles to outside markets, whereas the amount of turtle meat consumed in the community decreased by 14%. From 1969 to 1976, up to 10,000 green turtles were exported annually. During this period the average amount of time it took to capture one turtle increased from two person-days in 1971 to six person-days in 1975 (Nietschmann 1979b, 1982). Although green turtle products are no longer exported from Nicaragua, Miskitu Indians continue to harvest turtles to meet their dietary and monetary needs. From 1985 to 1990, J. Montenegro Jiménez (unpublished) recorded the sale of 16,700 green turtles in the Puerto Cabezas, Nicaragua, market.

Today, a very active Miskitu and Rama Indian and Creole marine turtle fishery continues off the Caribbean coast of Nicaragua (Lagueux 1991, 1998). Green turtles are the focus of the fishery, and the majority of those harvested are large juvenile females. The current annual harvest rate, a minimum of 10,000 green turtles, has probably remained fairly constant since 1991. Hawksbills are taken occasionally, and loggerheads and, rarely, leatherbacks are captured incidentally in nets set for green turtles. Loggerheads and leatherbacks are discarded unconscious or dead when they are captured; loggerheads are sometimes used for lobster or shark bait. Although loggerheads and leatherbacks are sometimes released alive, most turtlers club them unconscious to facilitate their removal from the nets.

INDIAN OCEAN

Less is known about the historical and current human use patterns of sea turtles from the coasts and nearshore areas of continental Africa than about those from any other geographic region. There is, however, evidence of a long history of exploitation of these species.

The decline in the number of green and hawksbill turtles on the Kenyan and Somalian coasts is due to 2,000 years of exploitation and, more recently, coastal development and pollution (Parsons 1962; Frazier 1982b). In Tanzania, Frazier (1982b) reports that sea turtle populations have probably been reduced since prehistory.

Hughes (1973, 1975, 1982) reported that in Madagascar the green turtle, loggerhead, olive ridley, and leatherback are exploited for domestic consumption; only the hawksbill is exploited commercially. From as early as 1613 until the early 1970s, tortoiseshell has been an important export for Madagascar (Decary 1950). Parsons (1972) cited the Red Sea as the source of the tortoiseshell of antiquity. The tortoiseshell trade in the Indian Ocean was well established by the first century (Freeman-Grenville 1962). Frazier (1975) cited overharvest and habitat destruction by humans as the two main causes of the decline of both green and hawksbill turtles in the western Indian Ocean. For example, he estimated that in the early 1970s there were fewer than 5,500 nesting green turtles in the western Indian Ocean. Only 38 years earlier, 12,000 animals were harvested in 1 year from the vicinity of Aldabra Atoll alone. Sea turtles no longer nest on Mauritius due to human exploitation (Frazier 1982a; Hughes 1982).

Hawksbill and green turtles have been important resources for the inhabitants of the Republic of Seychelles since those islands' discovery in 1609 (Frazier 1982b; Mortimer 1984; Stoddart 1984). As early as the eighteenth century, there was concern expressed over the exploitation of both species (Frazier 1974; Mortimer 1984). Until July 1994, the Turtle Act of 1925 was the basis for management of the turtle harvest in the Seychelles. The focus of the Turtle Act was not protection of turtles but rather establishing ownership rights and compilation of catch statistics. Many changes have been made to the Turtle Act over the years, including the setting of minimum size limits, protection of female turtles and their eggs, seasonal harvest restrictions, and the regulation of local and international trade (Mortimer 1984). In 1994, complete legal protection for all sea turtles and their eggs was imposed under the Wild Animals and Bird Protection Act. In the same year the government began a program to purchase the available stock of hawksbill shell and to assist hawksbill shell artisans in securing alternative forms of employment through compensation and job training (Collie 1995). No legal export of sea turtle products now occurs, although illegal trade may be occurring with Asian markets (J. Mortimer, personal communication).

The general decline in the annual number of hawksbills captured in the Seychelles between 1894 and 1959 reflects a population decline caused by overharvest (Mortimer 1984). For green turtles the most drastic decline began in the early twentieth century with the organized exploitation of the species for **calipee** (the cartilaginous material located between bones of the plastron). Indications that the harvest negatively affected the population were (1) a decrease in numbers harvested, (2) a decline in the size of turtles, (3) a change in the distribution of green turtles throughout the islands (Hornell 1927), and (4) a decline in the number of nesting animals encountered (Hornell 1927; Hirth and Carr 1970; Frazier 1975, 1979; Mortimer 1984, 1985).

Based on a study conducted from 1981 to 1984, Mortimer (1984, 1985, 1988a) concluded that the estimates of green turtle nesting density on Aldabra were more than twice as high as those made by investigators in the 1960s and early 1970s. She attributed the apparent increase to a reduction of human-induced mortality on the nesting grounds, periodic historical decreases in exploitation, and a 6-month-long closed season imposed each year from 1948 to 1962.

There are sea turtle populations in the northwestern Indian Ocean that have not been significantly reduced by exploitation. This could be due, in part, to the large Muslim population in the region. Islamic law prohibits the consumption of turtle meat (but not turtle eggs). There are, however, several areas in the region where this religious prohibition no longer is followed, and some turtle meat is consumed locally (J. P. Ross and Barwani 1982). Throughout the Persian Gulf it is a common practice to render oil from leatherbacks for use in treating wooden boats (J. P. Ross and Barwani 1982). In Iran green turtle eggs are harvested (J. P. Ross and Barwani 1982). From both the Persian Gulf and Red Sea coasts of Saudia Arabia, sea turtles and their eggs are harvested for subsistence use and sale at local markets (J. D. Miller 1989). On the Red Sea coast the turtle penis is considered an aphrodisiac, and thus turtlers select males (J. D. Miller 1989).

In India sea turtles are captured at sea or on the nesting beaches, and their eggs are collected (Kar and Bhaskar 1982). In addition to the use of sea turtles for protein and as a source of income, sea turtle oil is used to caulk boats and to protect wood against boring insects, and salted flipper skin is sometimes used to make shoes (Kar and Bhaskar 1982). The state of Orissa has the largest known concentration of nesting olive ridleys in the world (Mohanty-Hejmadi and Sahoo 1994). In 1975, the Bhitar Kanika Wildlife Sanctuary was established, in part to protect the nesting turtles and their eggs. Prior to 1975, the government sold rights to collect approximately 2 million eggs per year, which were sold locally and regionally (Bustard 1980; Kar and Bhaskar 1982). People along the southeast coast use sea turtle meat and blood to treat certain ailments (Silas and Rajagopalan 1984), and for several decades a green turtle fishery has existed in this area, both for subsistence use and export to Sri Lanka (Silas and Rajagopalan 1984).

In Sri Lanka, historically only green turtles were eaten; other species of sea turtle were released if caught accidentally in fishing nets. However, Frazier (1982a) reported that now all species are eaten, and there is tremendous pressure on nesting turtles. Salm (1975 *in* Frazier 1982a) estimated that 50,000 people in Sri Lanka were dependent on the turtle fishery. Hawksbills have been extirpated from the waters of the southern part of the country (Frazier 1982a).

PACIFIC OCEAN

In Thailand green, hawksbill, olive ridley, loggerhead, and leatherback turtles have been exploited for their eggs. From 1963 to 1973, there was a 70% decrease in the number of eggs collected annually at one site (Polunin 1975; Settle 1995). Today, the loggerhead is believed to be extirpated, and the other four species are seriously depleted (Polunin 1975; Mortimer 1988b, Settle 1995). Due to Islamic influence, sea turtles have not been harvested in Malaysia for the past 500 years (Hendrickson 1958; Polunin 1975). However, local customs do not prohibit eating sea turtle eggs, which are considered delicacies with aphrodisiac properties (Hendrickson 1958; Siow and Moll 1982). Egg collecting has had particularly drastic consequences on the Rantau Abang leatherback rookery in the state of Terengganu, where Mortimer (1990a) and Chan and Liew (1995) have documented a 98% decrease in nesting. Egg collecting has also been a significant factor in the decline of green turtle and hawksbill nesting on Pulau Redang Island (Mortimer 1991; Limpus 1994).

As early as 1839, it was reported in Sarawak that between 5,000 and 6,000 green turtle eggs were collected daily from Talang Talang Kechil (one of three Sarawak islands known for green turtle nesting); however, the duration of this harvest rate was not reported (Hendrickson 1958; Harrisson 1962a). By the mid-nineteenth century, systematic collection of green turtle eggs began. Sections of beach were leased by the government, and nearly 100% of the eggs were collected (Harrisson 1951, 1962b; Hendrickson 1958). Prior to the nesting season an elaborate ceremony was held so that turtles would return to lay their eggs (Harrisson 1951, 1954–59). In 1941, management of the egg harvest fell under the jurisdiction of a Turtle Board of Management, specifically created to oversee egg collection, a hatchery program, the management of funds produced from the sale of the eggs, and disbursement of the proceeds to Malaysian mosques and charities (Hendrickson 1958; Harrisson 1962b). In spite of a controlled egg harvest, and the implementation of an egg hatchery program, there has been more than a 90% decline in egg production on the three turtle islands of Sarawak since 1927 (Harrisson 1962a, 1962c, 1966, 1967; Chin 1968, 1969, 1970, 1975; G. S. de Silva 1982; Limpus 1994).

In Sabah, hawksbill and green turtle nesting occurs on three main island groups off the coast. Exploitation of turtle eggs began over 50 years ago (G. S. de Silva 1982). From the mid-1950s to the mid-1970s there was a 50% decline in green turtle egg yields (Harrisson 1964, 1966, 1967; G. S. de Silva 1982; Limpus 1994).

Aside from the overharvest of eggs in both Sabah and Sarawak, much of the decline in sea turtle populations has been attributed to exploitation by the Japanese for meat and for turtle soup (Harrisson 1964). In Sabah factors that have resulted in the continued decline of sea turtle populations are (1) the presence of brightly illuminated fishing vessels near the nesting beaches, (2) illegal hunting of turtles, (3) increased powerboat activity off the nesting beaches, (4) dynamiting of fish near turtle rookeries, and (5) the uncontrolled harvesting of large numbers of turtles outside Sabah waters by Filipino and Japanese fishing vessels (G. S. de Silva 1982).

In Indonesia large numbers of adult and juvenile sea turtles are killed for their meat and shell, and thousands of eggs are collected annually from nesting beaches. These activities appear to have had a significant impact on the wild populations (e.g., a more than 80% reduction in the number of green turtle eggs laid from 1934 to 1984; Schulz 1984). Barr (1992) estimated that 7 to 9 million sea turtle eggs are collected annually in Indonesia, essentially 100% of all eggs laid. Most are consumed locally, but an export trade has been reported (Chin 1968).

One of the world's largest green turtle fisheries occurs in southern Bali (Barr 1992; Limpus 1994). Turtles are killed mainly for their meat and are used in religious ceremonies and feasts. Barr (1992) reported that at the peak of the trade over

30,000 sea turtles were brought into Bali every year. By 1950, local sea turtle populations were seriously depleted (Sumertha Nuitja 1974 *in* Polunin and Sumertha Nuitja 1982). With the depletion of the Bali populations, sea turtles have been taken from a wider area. Limpus (1994) reported an annual harvest of 25,000 animals, mostly large green turtles taken from throughout Indonesia. In addition to being sold for meat, skin, eggs, and bones, green turtles are exploited for their shells, which are used in Jakarta as a furniture veneer and possibly in Chinese medicine (Barr 1992).

In 1953, over 1 million eggs were harvested from the Philippine turtle islands (Parsons 1962). Kajihara (1974 *in* Polunin 1975) reported that between 1961 and 1972, 5,000 adult hawksbills and 50,000 green turtles were captured annually in the Sulu Sea, and tortoiseshell from approximately 45,000 hawksbills was exported to Japan. From 1951 to 1984 there has been a greater than 75% decline in green turtle egg production from the Philippine turtle islands (Domantay 1953). The population decline is attributed to overharvesting of eggs from 1951 to 1993 (Domantay 1953; Ramirez-de Veyra 1994). Hawksbill populations have also been severely depleted due to overharvest of eggs and the large international demand for tortoiseshell (Ramirez-de Veyra 1994).

For 40 years prior to 1954, green turtles were hunted on Capricorn Reef, Australia, transported to Brisbane, and shipped to England (F. McNeill 1955). A turtle-processing plant located on the west coast of Australia processed meat until 1951. Today, marine turtles are protected in Australia; however, indigenous people in Queensland and Western Australia are allowed to take turtles for their own use (Limpus 1982). It is estimated that 10,000 green turtles are harvested annually from the Torres Straits. Of these, approximately 4,000 are harvested by the Torres Strait Islanders and used locally; the remainder are harvested by Papua New Guineans and sold in their coastal markets (Limpus 1982; Daly 1990). In areas where indigenous communities are located near nesting beaches, nearly 100% of the eggs laid are collected (Limpus 1982).

In Papua New Guinea, marine turtles are second to fish as a source of protein (Spring 1982a). Traditionally, marine turtles were important for feasts, celebrations, and ceremonies, for example, repayment for a bride, funerals, building of a canoe or house, and the birth of a first child. Today, turtle meat plays a role in nontraditional celebrations regarding business, political, and religious activities, with up to 60 turtles used for a feast. Adult female green turtles are preferred over juveniles and males because of their higher fat content (Spring 1982a). Because of overharvesting, turtles are no longer found feeding in offshore areas or nesting on beaches in proximity to villages (Spring 1982b). However, in villages where the residents have become Seventh-Day Adventists (who do not eat turtle meat), an increase has been noticed in turtle populations over 30 to 50 years (Spring 1982a, b). Spring (1982a) attributed the decline of marine turtle populations to the breakdown of traditional restraints on catching turtles, the introduction of modern fishing methods, the increase in human population, and the change to a cash-based economy in the villages.

Hawksbills are consumed in Papua New Guinea, and the shell is used to make jewelry, combs, and, in some places, bride-price items. In the past, the scutes were used to make everyday household items (Spring 1982a). Entire hawksbill shells are also kept as wall decorations or sold to tourists. Leatherback meat and eggs are usually consumed locally (Spring 1982b).

In the eastern Pacific a large industry developed after 1960 based on hunting sea turtles for their skins. The amount of skin available on a sea turtle is small (it is only removed from the front flippers and underside of the neck), and sea turtle skin is thinner than cowhide and not as durable (Pritchard 1979). Mexico and Ecuador have been the leading exporters of turtle skins and leather, the majority of which was imported by Japan followed by France, Spain, Italy, and the United States (Pritchard 1978; Mack et al. 1982; Milliken and Tokunaga 1987a). Both the olive ridley and the green turtle were hunted for their skin, but the olive ridley's habit of congregating in large aggregations to nest made it an especially easy target.

Along the Pacific coast of Mexico, the commercial exploitation of olive ridleys began in 1961 (Márquez M. et al. 1976; Cato et al. 1978). The government's strategy was to conserve sea turtles through controlled exploitation of the adults and total protection of eggs on the nesting beaches (Pritchard 1978, 1979). However, according to Cato et al. (1978), the marine turtle industry developed in response to the high prices paid for turtle skin and the discovery of large aggregations of nesting turtles. During the 1960s and 1970s, new markets had developed for sea turtle skin and leather because of the decrease in availability of crocodilian skins (Mack 1983; Alvarado et al. 1990). Eggs of all species of sea turtle continued to be collected illegally.

Extremely high levels of exploitation of olive ridleys were reported by Márquez M. et al. (1976), and during the peak period of exploitation, between 1965 and 1969, over 30,000 metric tons of olive ridleys, representing over 775,000 individuals, were slaughtered. Exploitation was so great that at Piedra de Tlacoyunque, one of only four principal nesting beaches on the Pacific coast of Mexico, the nesting aggregation of turtles had been reduced from 30,000 to only a few hundred between 1968 and 1969 (Carr 1972; Pritchard 1979). By the early 1970s, three of the four nesting aggregations of olive ridleys had been destroyed (Carr 1967, 1979; Frazier 1981), and the remaining site, at Playa Escobilla, Oaxaca, was being heavily exploited.

By 1969, Mexican law allowed the exploitation of sea turtles only by those companies that used the entire animal (Cahill 1978; Cliffton et al. 1982). Illegal hunting, and legal harvesting with reduced quotas, continued through the 1970s and the 1980s. In 1976, a ban on taking turtles during the nesting season was lifted (Cahill 1978; Cliffton et al. 1982). In 1977, there were between seven and nine fishing cooperatives licensed by the government to harvest up to 1,500 turtles per month. All of the cooperatives sold their catch to one company, Pesquera Industria de Oaxaca (PIOSA) (Cahill 1978). Based on what was legally allowed, Cahill (1978) estimated 40,000 animals were slaughtered between July and September 1977. However, Cliffton et al. (1982) estimated that PIOSA had processed 70,000 animals in 1977, almost twice as many as the legal limit. An estimated 90% of them had been gravid (Cahill 1978; Pritchard 1978; Frazier 1981; Cliffton et al. 1982). During this time olive ridley meat was also smuggled into the United States as Tabasco river turtle. In October 1977 no *arribada* (mass nesting) occurred. In May 1990, Mexico declared a permanent ban on all harvest and trade in sea turtles and their products, and in 1991 Mexico became a signatory to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 1973; Aridjis 1990; D. A. Rose 1993).

In Mexico green turtles have a long history of exploitation. Nineteenth-century whaling vessels replenished their meat stores with green turtles captured along the Pacific coast of Baja California (Caldwell 1963; Felger and Cliffton 1977), and by the turn of the twentieth century there was an active turtle fishery. Green turtle meat was canned in Baja California and exported to England, and live turtles were shipped to San Francisco. Though the cannery was closed 15 years later, shipments of live turtles to San Francisco continued (E. W. Nelson 1921). By the 1930s, the market for fresh turtle meat in the United States had declined; however, the demand grew in the border towns and cities of Baja California and Sonora. According to Felger and Cliffton (1977), green turtles once nested along the coasts of Nayarit, Sinaloa, southern Sonora, and Baja California. Today only the olive ridley nests in these areas. The increase in human population and subsequent exploitation of nesting females were responsible for the disappearance of these northern Pacific Mexico populations.

Green turtles once nested widely along the southern Pacific coast of Mexico, but today only one major nesting site remains at Colola-Maruata Bay, Michoacán. Here Nahuatl Indian informants reported to Cliffton et al. (1982) that there were 10 to 20 times more nesting green turtles in 1970 than in 1979. Apparently prior to the 1970s, these people disapproved of the wholesale slaughter of green turtles at turtle rookeries because they viewed such slaughter as a threat to their own commercial interests. However, by the early 1970s, they began to sell large numbers of sea turtle eggs.

In 1978, PIOSA began processing green turtles legally harvested by the fishers (Cliffton et al. 1982). At about the same time, a coastal highway was completed that passed adjacent to the two nesting beaches. The highway provided the means to increase the commerce in sea turtle products between the nesting beaches and other states within Mexico. Although in 1979 the Mexican government declared a closed season for the nesting beaches in an attempt to gain control of the green turtle fishery, approximately 3,000 green turtles were illegally taken (Cliffton et al. 1982). In 1980, the government decided to allow a legal take of 250 male green turtles per month between September and December. Cliffton et al. (1982) reported that the attitude of the local fishers toward the green turtle recovery program changed when they were allowed to harvest turtles legally. The local people now participate in protecting the nesting females and respect the established harvest quota.

In Ecuador, olive ridleys were harvested during the 1970s and early 1980s, primarily for their skin but also for their meat. In the 1970s, at least six companies were involved in exporting frozen sea turtle meat for human consumption and salted skin for the leather trade. (Green and Ortiz-Crespo 1982). Some reports state that turtles were often skinned on board ship and the meat discarded (Hurtado G. 1982).

From 1965 to 1987 in Peru the estimated annual harvest (based on an average weight per individual) ranged from less than 100 to more than 22,000 animals (Aranda and Chandler 1989). The harvest most likely represented a combination of green turtles, olive ridleys, and leatherbacks. Aranda and Chandler (1989) reported an increase in the consumption of turtle meat in larger cities, more out of economic necessity than for any particular desire to consume it. Turtle meat is sold in local markets and restaurants, and varnished shells are sold to tourists. Vargas et al. (1994) reported that the marine turtle fishery in Peru was still active. Apparently, the mesh size of nets used to trap turtles has decreased from a 59-cm bar used in 1979 to a 25-cm bar used in the early 1990s (Hays Brown and Brown 1982; Vargas et al. 1994), which may indicate that the animals have decreased in size due to overharvesting.

It is clear that turtles have a long history of exploitation for their meat, eggs, and a variety of by-products. Apart from the well-known cases of the giant tortoises (*Geochelone* spp.) of the Galápagos and islands of the Indian Ocean, and some of the larger river turtles (e.g., the giant South American river turtle and the river terrapin), relatively little is known of the extent or levels of use of most tortoise and freshwater turtle species, either historically or at present. In contrast, there is a great deal of information regarding the exploitation of marine turtle populations. This is undoubtedly due to the exceptional value of marine turtles as both a food source and an item of commerce throughout recent human history. Both historic and recent records document the extraordinary level of take from marine turtle populations around the world. In some regions (e.g., the Caribbean) these intensive levels of exploitation began earlier than in others, corresponding with the beginning of intensive European explorations and colonizations and the resulting expansion of trade activity in these regions. In other parts of the world the use of marine turtles and their eggs remained of a subsistence nature, and only relatively recently (i.e., during the twentieth century) have levels of exploitation increased significantly in some of these areas. During the twentieth century the exploitation of marine turtle populations has increased to unprecedented levels. Experience has shown that the unregulated take of both adult turtles as well as their eggs, whether intentional or otherwise, results ultimately in greatly diminished populations levels.

USE OF TURTLES IN THE PET TRADE

EUROPE AND NORTHERN ASIA

Europe is a major market for both native and exotic species of turtles as pets. Information is most complete for this region and provides us with a picture of the dynamics of the turtle pet trade. Also, concerns about the overexploitation of turtle populations for international pet markets originated in Europe and initiated some of the first controls on the trade. Most detailed information on the European market is limited to the United Kingdom, and these data are mostly limited to CITES statistics (Smart and Bride 1993). Because many countries do not accurately track imports and exports, and do not report illegal activities, CITES information represents the minimum level of trade in these species. Data on the international trade in species not listed under CITES and domestic turtle trade are rarely available.

In the early 1980s, chelonians were a major segment of the reptile trade in the United Kingdom, especially tortoises of the genus *Testudo*, North American box turtles (*Terrapene* spp.), and hatchling red-eared sliders. From 1980 to 1983, imports of tortoises were dominated by just two species: the spur-thighed tortoise (84%, or 131,640) and Hermann's tortoise (14.4%, or 22,500). Since 1984, the European Union has prohibited the importation of these two species as well as the marginated tortoise (*Testudo marginata*) (Fitzgerald 1989; Smart and Bride 1993). The Testudinidae (tortoises) represented almost 70% of all imports of species listed under CITES into the United Kingdom from 1980 to 1989 (Smart and Bride 1993).

Tens of thousands of other, nonlisted species of turtles were also imported into the United Kingdom during the late 1980s. Over 43,000 turtles not listed under CITES were imported from 1986 to 1989, almost one-quarter being common box turtles from the United States (Smart and Bride 1993). The next three most prevalent species were imported in numbers half that of the common box turtle. Emydids dominated nonlisted species imports, though 5,300 kinosternids (three species), 4,000 chelydrids (two species), and almost 3,000 trionychids (two species) were imported as well. In addition, red-eared slider hatchlings were a dominant component of the pet turtle market. Although exact import figures are unknown, some reports suggested as many as 200,000 per year were imported prior to a consumer campaign to discourage their purchase (Rowe 1993).

According to Smart and Bride (1993), a change in the market appears to have occurred during the 1980s, when turtles no longer formed a dominant segment of reptiles listed under CITES that were imported into the United Kingdom. By the early 1990s, only about 10% of the reptile species on dealer lists (in the United Kingdom) were turtles. This change may be attributed partly to the import ban on the main species of turtles traded in the early 1980s. As discussed below, it appears from the limited information on nonlisted species that other turtles, primarily North American box turtle species, were imported in ever-increasing numbers to replace the banned species of *Testudo*.

Two tortoise species, the spur-thighed and Central Asian tortoises, bore the brunt of most of the collecting prior to the 1980s. Populations of the widespread spur-thighed tortoise were greatly affected by the trade, and Lambert (in Smart and Bride 1993) found that adult survivorship of spur-thighed tortoises was reduced by 20% in areas with high collection pressure. During the 1970s, Morocco was the main supplier of this species to Europe, and populations of the spur-thighed tortoise were estimated to have been reduced by up to 90%. Morocco banned exports in 1978 when it ratified CITES. From 1980 to 1985, Turkey became the main supplier, exporting at least 263,000 tortoises to Europe. With the European import ban, Turkish exports plummeted. In 1996, Turkey finally set an export quota of zero, after pressure from the CITES Significant Trade Review recommended that all signatories to CITES suspend imports of the species from Turkey until the CITES Secretariat assessed the status of the species and effect of the harvest (USFWS 1996c).

Large numbers of the Central Asian tortoise were exported to Europe in the 1970s and 1980s. In 1967, 43,000 Central Asian tortoises were reported to have been collected in Kazakhstan; this number climbed to over 110,000 per year through the 1970s and was about 150,000 per year in the 1980s (Brushko and Kubykin 1982 in Inskipp and Corrigan 1992). Such high collection levels have led to the complete extirpation of the species from large areas. Although in 1984 the former Soviet Union imposed restrictions on the trade of the Central Asian tortoise, in 1989, 52,000 were reported in international trade statistics, mostly from the Soviet Union (Inskipp and Corrigan 1992). The Soviet Union reported exporting 20,000 Central Asian tortoises to France in 1989, a year after the European Union imposed a ban on importation of the species (Inskipp and Corrigan 1992). From 1989 to 1994, the Central Asian tortoise was the species most commonly imported into the United States, predominantly from the Soviet Union (HSUS 1994).

Once the European import bans on species of *Testudo* were in place, trade in turtles switched to other genera. Importations of hinge-back tortoises increased until 1992, when the European Union prohibited imports of this genus (Smart and Bride 1993). However, according to the United Kingdom and United States statistics, imports probably switched more to nonlisted species under CITES, predominantly to North American box turtles (Smart and Bride 1993). The United States began to track nonlisted species in international trade in the late 1980s and reported that from 1990 to 1994 over 100,000 common box turtles were exported, mostly to Europe (Bright 1994 *in* HSUS 1994). This volume of trade is not much lower than the imports of spur-thighed and Central Asian tortoises into Europe in the first four years of the 1980s, indicating that within several years of the *Testudo* ban, trade had shifted to other species and could have reached the same levels by the early 1990s.

NORTH AMERICA

Surveys by the pet industry and veterinary associations have estimated that between 1.5 and 3.2% of the general public in the United States owns a pet turtle. Because many people own more than one turtle, there might be 2.5 million to 15 million pet turtles in the United States alone (HSUS 1994). These surveys also have found that 35% of the owners obtained their turtle(s) from the wild, whereas 50% purchased them from a pet store. Species native to the United States are also collected in large numbers for export to overseas pet markets, predominantly in Europe and the Far East. Analysis of the domestic market is difficult because there are no requirements for the U.S. government to track domestic sales, and it has been only since 1989 that systematic records of turtle imports and exports have been kept.

The United States imports at least 30,000 turtles a year, primarily for sale as pets, and almost all of these are wild caught according to the Humane Society of the United States (1994). According to U.S. statistics, from 1989 to mid-1994 all but one of the most common genera imported were covered by CITES (HSUS 1994). The one exception was the genus *Cuora* (Asian box turtles), with approximately 33,000 imported over the 5-year period. The highest level of trade was found for the Central Asian tortoise, reportedly at over 22,000 animals. The total import volume for the 5-year period was a minimum of 124,000 turtles.

Recently, the United States has imposed import prohibitions on six species based on CITES recommendations: since late 1994 a ban on pancake tortoises and since mid-1995 a ban on leopard tortoises from Tanzania; since mid-1995 bans on three species of hinge-back tortoises from Ghana; and a ban for the first half of 1996 on spur-thighed tortoises from Turkey (USFWS 1994a, 1995a, 1996d). The ban on the spur-thighed tortoise was lifted once Turkey set an export quota of zero, thus making exports illegal (USFWS 1996c).

The sale of captive-hatched red-eared sliders was an economically important business in the southern United States during its peak years in the 1960s (Warwick 1986). Approximately 150 turtle ranches, all stocked from the wild, were supplying the U.S. market with 10 million animals every year. In 1975, the U.S. Food and Drug Administration linked the turtles to the transmission of *Salmonella* bacteria to humans, and the sale of turtles with a carapace less than 10 cm in length was banned (Warwick 1986). In the mid-1980s, some 50 ranches remained in business, supplying approximately 4 to 5 million hatchlings annually to overseas markets, for pets and food. Adults were also sold for food both domestically and internationally, though to a much lesser degree (Warwick 1986; Warwick and Steedman 1988). Just over half of this production went to Europe, with the remainder going to Asian markets. In 1993, South Korea was the single largest importer (over 1.38 million), Italy second, and Japan third (HSUS 1994). Red-eared slider ranches rely on large numbers of wild-caught, gravid females for stocking their facilities. For four ranches Warwick (1986) estimated 9,400 adults were collected annually, which could mean about 100,000 adults removed each year by the entire industry. Warwick and Steedman (1988) referred to anecdotal reports of tenfold decreases in the catch of adults in harvested areas. These operations appear to have caused a decline in local red-eared slider populations.

In the late 1980s and early 1990s, the United States also exported large numbers of other turtles, almost all wild caught. Federal statistics reported about 800,000, whereas a compilation of exporter declaration reports for 2 years during the same period totaled an additional 140,000 for just four of the commonly exported species. Turtles from six different genera constituted the most common exports. With the exception of one genus (*Clemmys*, with approximately 5,000 exported), more than 10,000 individuals of each genus were exported, and several genera had over 100,000 individuals exported (HSUS 1994). At over 300,000, painted turtles (*Chrysemys picta*) topped the export list, and North American box turtles and map turtles averaged around 100,000 each. Over 60,000 North American softshell turtles were exported, followed by 10,000 to nearly 30,000 mud (*Kinosternon* spp.), musk (*Sternotherus* spp.), snapping, and alligator snapping turtles. Export prices of around $5 per turtle indicate that these are not for the high-end pet market, though underreporting the value of exports is a common practice. Only the export price of alligator snapping turtles reached nearly $20 per turtle (HSUS 1996).

There is growing cause for concern that high export levels are negatively affecting wild populations of many of these commonly exported species. The effects of overcollection of North American box turtles for the pet trade, both domestic and international, are exemplary. Once ubiquitous throughout the eastern third of the United States, the common box turtle has declined in the wild due to habitat loss and fragmentation and collection for the pet trade. The U.S. government began tracking exports in the late 1980s. Though it is unknown how many common box turtles are collected for the domestic pet trade, international trade rose dramatically from only 1,000 turtles in 1989 to an average 25,000 annually from 1992 to 1994 (USFWS 1994b). Combined with tens of thousands of ornate box turtles (*Terrapene ornata*), at least 100,000 North American box turtles of both species were exported in the first half of the 1990s (Bright 1994 *in* HSUS 1994; USFWS 1994b). Following the European ban on imports of species of *Testudo*, these exports were predominantly to Europe. Though both species of North American box turtles were protected in 50% of the political jurisdictions where they occur, many states report extensive illegal trade activity.

Based on a 10-year study of a disjunct population of the ornate box turtle, Doroff and Keith (1990) concluded that the population could not withstand the loss of just one adult per year. Such results suggest that collecting can be a serious threat to wild populations. This study, in conjunction with the rapid increase in exports and the lack of controls on collection, motivated the U.S. federal government to list the genus *Terrapene* on CITES Appendix II in 1994 (USFWS 1994b). Listing on Appendix II requires that all trade be conducted in a manner that will not jeopardize the wild source populations. Although the CITES listing required an assessment that exports would not be detrimental to wild populations, in 1995 the United States approved an export quota of nearly 10,000 North American box turtles based on prior export levels from Louisiana, the only state approved to export this species. Upon reevaluation, and pressure from experts and conservation groups, the 1996 export quota was set to zero; thus, no exports of North American box turtles from the United States are currently allowed (USFWS 1996e).

Demand by collectors, both in the United States and overseas, drives the exploitation, legal and otherwise, of rare species. In 1992, the bog turtle (*Clemmys muhlenbergii*) was transferred from CITES Appendix II to I, thereby prohibiting international trade (USFWS 1992a). This small species has very specific habitat requirements and is found in isolated populations throughout its range. Already suffering from severe habitat loss, collection for the pet trade can easily eradicate entire populations or drastically reduce reproductive potential. In 1992, the wood turtle (*Clemmys insculpta*) was also listed on Appendix II of CITES because of trade that took advantage of the varying levels of protection afforded the species by different states within its range (USFWS 1992b).

SOUTH AND CENTRAL AMERICA AND THE CARIBBEAN

There is only limited information regarding Central and South American turtle species in the international pet trade. Several species of *Geochelone* are exported, primarily to the United States. Three species appear in U.S. import statistics for 1989 to 1994, with two having been imported in the thousands—the South American red-footed tortoise and yellow-footed tortoise (*G. denticulata*). A CITES review in 1988 highlighted the trade in the Chaco tortoise (*G. chilensis*). Though this review reported that there was insufficient information to assess whether collection of this species for international trade was a threat, it found that populations in central and northern Argentina suffered marked declines due to this trade in the early 1980s. Tens of thousands were collected annually for both domestic sale and international markets, principally the United States (see Waller 1997).

A CITES review of international trade in the red-footed tortoise reported high trade levels from Guyana (Inskipp and Corrigan 1992). About 8,500 were exported from Guyana between 1983 and 1988, though volumes were erratic (as high as 3,600 one year and less than 200 in another). In 1991, the country set an export quota of 1,000 per year and then instituted a ban on exports before establishing new quotas in 1996. Export quotas that CITES signatory countries must honor are 500 each of red-footed and yellow-footed tortoises. For Suriname, a quota of 630 and 692 was established for each species, respectively (CITES Secretariat 1996). These quotas are about the same as the recent annual imports into the United States alone (HSUS 1994).

AFRICA

The Egyptian tortoise and pancake tortoise (see "Pancake Tortoise: Exploitation for the Pet Trade," this chapter) are prime examples of species for which the demand for pets has driven overexploitation and caused severe population declines. The rarity of other African species, and those endemic to Madagascar, makes individuals worth thousands of dollars in the wildlife trade.

Collection for the pet trade is responsible for the near extirpation of the Egyptian tortoise from Egypt (Baha el Din 1994); only a few small, probably nonviable, populations remain. In the 1970s, Egypt was the main source of this species for Europe (Baha el Din 1994; Egypt 1994). Exports in the mid-1980s were relatively low, probably around 200 annually. In 1990, trade increased due to smuggling of Egyptian tortoises from Libya. Based on visits to Cairo's main wildlife market, Baha el Din (1994) estimated that over 8,000 Egyptian tortoises, almost all from Libya, were sold from 1990 to 1994, 80% domestically and the remainder for export. In 1994, the resurgence in exports, collection pressure, and illegal activity prompted the transfer of the species from Appendix II to I of CITES to buttress the domestic prohibitions with an international ban (Egypt 1994).

Tanzania's lack of control over exports and illegal trade was also a concern in regard to the leopard tortoise and hinge-backed tortoises. A 1992 CITES review concluded that international trade was probably threatening these species in Tanzania (Inskipp and Corrigan 1992). From 1989 to 1994 more than 10,000 leopard tortoises and about 30,000 hinge-back tortoises were imported to one major U.S. market (HSUS 1994).

INDIAN SUBCONTINENT

The Indian pet trade in turtles targets only a few species. Two, the Indian black turtle *(Melanochelys trijuga)* and the Indian roofed turtle *(Kachuga tecta)*, appear to have entered the markets recently but are not commercially exploited on a large scale (Choudhury and Bhupathy 1993). The Indian star tortoise *(Geochelone elegans)* is the most important Indian species in both the domestic and export pet trade. Export markets include Europe, the United States, and major Southeast Asian cities (Choudhury and Bhupathy 1993; HSUS 1994; Jenkins 1995). The annual trade volume in the 1990s was estimated to be around 10,000, but there are no accurate figures because this trade is illegal under Indian law. Animal dealers in many Indian cities reported selling hundreds, and from 1991 to 1992 over 1,500 were seized by authorities in cities in India and in European markets such as Amsterdam.

In Bangladesh the nature of the turtle pet trade is unclear. Das (1990) reported that in the 1980s there was no pet trade in native turtles. However, Jenkins (1995) reported the export of over 1 million kilograms of elongated tortoises from 1991 to 1993. Though this species is commonly exported to the United States and Japan from Southeast Asian countries for the pet market, it is entirely possible that this huge increase in exports from Bangladesh represents animals destined for human consumption, especially if the turtles were exported to China.

SOUTHERN ASIA

There are both local and regional pet markets for turtles in Southeast Asia, and in recent years there has been a rapid growth in the international trade in turtles originating from this region. This expansion has been driven largely by the Chinese markets for turtles, primarily for food and medicine, and it is difficult to determine to what extent the pet trade is involved. In general, hatchlings are sold as pets for export to Europe and the United States. Export of live turtles to China are presumed to be used mostly for food, although a recent market study indicates that demand for pet turtles is growing in southern China and Hong Kong (F. Holland 1996).

The sale of turtles for pets regionally, especially hatchlings, is common. Interestingly, the most popular turtle species sold in pet stores in the larger cities, such as Bangkok and Kuala Lumpur, is the red-eared slider, imported from the United States (Jenkins 1995). Also popular in the pet trade in these cities is the Indian star tortoise, mostly illegally imported (Jenkins 1995). Pet stores in Peninsular Malaysia commonly sold two native species, the Malayan box turtle and Asiatic softshell turtle (Jenkins 1995). However, it is unclear to what degree these sales are for pets versus human consumption because pet shops in this area sell turtles for both purposes.

Local markets for turtles as pets have also been reported in Vietnam. There is a seasonal business in Tam Dao hill resort in Vinh Phu Province, about 85 km north of Hanoi, where juvenile black-breasted leaf turtles *(Geoemyda spengleri)* are sold to tourists for around $0.05 (Le Dien Duc and Broad 1995). A second native species, the Malayan snail-eating turtle *(Malayemys subtrijuga)*, is extensively traded in Vietnam and commonly sold as pets. It was the most common turtle species recorded in the main wildlife market, Cau Mong, in Ho Chi Minh City (Le Dien Duc and Broad 1995). The juveniles are sold for pets, and the adults are a major export to China, presumably for food and medicine.

Southeast Asian turtles are exploited for the international wildlife trade although they do not appear to be a major component of that market. Reported export statistics of species listed under CITES are not as high as the numbers reported from other regions. This difference could be a reflection of the lack of access to species from this region, which may be changing, as the rising Vietnamese export market indicates. Also, in many cases, it is not clear if turtle exports are for pets or human consumption because they are used for both purposes. In evaluating the existing information, unless specified it was assumed that exports to Western markets were to supply the pet trade and exports to China were for food. In some cases the prices paid for these turtles imply that they are destined for the higher-end pet market in Western or Far Eastern markets.

Several species of Asian box turtles are commonly exported from the region, especially the Malayan box turtle but also the Indochinese box turtle *(Cuora galbinifrons)* and the Chinese three-striped box turtle *(Cuora trifasciata)* in lesser quantities. Because this genus is not listed by CITES, there are few international trade statistics, and those that do exist are rarely species specific. From 1989 to 1994, this genus was the most common import into the United States with almost 33,000 specimens reported (HSUS 1994). The main exporting countries, in descending order, were Indonesia, Malaysia, Thailand, Hong Kong, Philippines, and China, though some of these countries are reexporting nations (based on U.S. federal import statistics supplied by the Humane Society of the United States). Indonesia apparently exported hundreds of thousands of Malayan box turtles annually in the late 1980s, though not all for the pet trade. This export level far exceeds the domestic harvest quota of 10,000 a year set in 1991 (Jenkins 1995).

A variety of Southeast Asian tortoises are also targeted for the pet trade. Indonesia and Malaysia are the primary exporting countries, and the main importing nations are the United States and Japan. The main species exported are the Asian brown tortoise, impressed tortoise *(Manouria impressa)*, elongated tortoise, and Travancore tortoise. From 1988 to 1993, Indonesia exported 450 Asian brown tortoises (75% to the United States) and over 1,700 Travancore tortoises (>50% to the United States) (Jenkins 1995). From 1990 to 1993, Malaysia exported 5,400 elongated tortoises, 1,200 Asian brown tortoises, and 800 impressed tortoises, in all cases with approximately two-thirds going to Japan and the remainder to the United States (Jenkins 1995). Most of these animals are for the high-end pet market, with few seen in local markets in Malaysia. Prices of $150–180 for an impressed tortoise and around $100 for an Asian brown tortoise appear to reflect the degree of difficulty in obtaining these species.

A CITES review in 1992 indicated that international trade in the Asian brown tortoise and elongated tortoise from Malaysia was a problem and recommended that Malaysia justify the biological basis of its export levels and undertake a field assessment of both species (Inskipp and Corrigan 1992). Malaysia set 1996 exports at 300 and 1,000, respectively. These numbers were similar to the average annual exports of these species from 1990 to 1993 (CITES Secretariat 1996). These export levels, of animals primarily for the pet trade, are tiny compared with the 1 million kilograms of elongated tortoises exported from Bangladesh during the same period (Jenkins 1995).

Other Southeast Asian turtle species, usually rare ones, are in demand by the pet market as well. This demand is not high but can constitute a major threat to rare or dwindling wild populations. Though Myanmar is not a member of CITES, statistics from CITES signatory countries indicate a pet trade in the endemic Burmese star tortoise. Japan reported importing over 1,000 specimens from 1990 to 1992, even though this species is apparently quite rare in the wild due to local consumption (Jenkins 1995). The narrow-headed softshell is severely depleted in its range in Thailand from habitat loss, pollution, and hunting, and Japanese collectors have put added pressure on the species. Large adults are taken alive and sold for hundreds of U.S. dollars (Jenkins 1995). In Malaysia large river turtles, such as the river terrapin, painted terrapin, and Malaysian giant turtle, are also exported for pets, with adults bringing high prices (Jenkins 1995). International trade in river terrapin has been prohibited by CITES since 1975.

CASE STUDIES

Giant Tortoises: The Overharvesting of Vulnerable Populations

Human-related factors have caused the decline, and in some cases the extinction, of giant tortoises on islands in the Indian and Pacific Oceans. These factors include

high levels of exploitation for food (both commercial and subsistence), the introduction of feral animals that act as predators of eggs and young or competitors for scarce vegetation, and the modification of habitat associated with growing human populations. On Madagascar, two species of giant tortoises became extinct at approximately the same time the island was being settled by humans (Van Denburgh 1914). The Malagasy species *Geochelone grandidieri* is known from subfossil remains and was also found on many of the oceanic islands that surround Madagascar, such as Réunion, Mauritius, and Rodrigues (Van Denburgh 1914; Pritchard 1979). The world's most critically endangered tortoise, the angonoka *(Geochelone yniphora)*, a Madagascar endemic, was intensively exploited by Arab traders from the seventeenth to the nineteenth centuries. A small population remains in the Baly Bay region of northwest Madagascar, where indigenous peoples have a taboo against eating it (Juvik et al. 1981).

Historically, tortoises on the Seychelles and Mascarene Islands were reported to be extremely abundant, but populations on most islands were extirpated prior to 1800, and the subgenus *Cylindraspis* was driven to extinction on the Mascarenes (Stoddart and Peake 1979; Arnold 1980). Reports by early visitors to these islands comment on the high quality of the turtle meat and oil, which was used as butter (Stoddart and Peake 1979). One account of six voyages to the Mascarenes reports the taking of almost 21,000 tortoises from the islands (Milne Edwards 1874). The survival of the only remaining population, on Aldabra Atoll, has been attributed to its distance from traditional shipping lanes and the inhospitality of the atoll for human habitation (Pritchard 1979).

When humans first arrived on the Galápagos Islands, the Galápagos tortoise was abundant throughout the archipelago. The first recorded visit to the islands was in 1535 by Fray Tomas de Berlanga, a Spaniard so impressed by the quantity of tortoises *(galápagos)* that he named the archipelago after them (Van Denburgh 1914). Over the next 350 years, the islands became a convenient stopping point for all types of ocean travelers in need of replenishing their supplies of water and food. Van Denburgh (1914) provided a fascinating account of the succession of buccaneers, whalers, fur sealers, and others who left written records of their visits to the islands. Galápagos tortoises made excellent repast. They were easy to catch, tasty, and an ideal protein-rich food source easy to store in oceangoing vessels. Animals were collected alive and would frequently live for months (some reports state tortoises were stored alive in the ship's hold for over 1 year) without food or water. Tortoise meat was a welcome change from the usual sailors' fare, and oil rendered from the fat was saved in jars and used instead of butter or shortening (Van Denburgh 1914; Townsend 1925). Even the water in the bladder and pericardium was consumed (Darwin 1845). Visitors to the archipelago would make short visits, usually to outlying islands, for the purpose of collecting tortoises. Boats arriving from the north would typically stop at Isla Pinta, and those approaching from the south would frequent Isla Española or Santa María (Pritchard 1979). Hundreds of Galápagos tortoises could be collected at a time. Some reports suggested that medium-sized tortoises (23 to 34 kg) were preferred because they were more easy to carry long distances to the landing beaches yet provided appreciable quantities of meat (Townsend 1925). This preference may have led to a bias in the sex of the animals taken, as tortoises of this size were mostly females. Large tortoises were sometimes killed and their meat taken for immediate consumption (Townsend 1925). Only the extremely rough terrain, the predilection of Galápagos tortoises to inhabit mountainous parts of the islands, and their large size prevented the harvest from having an even more dire effect on tortoise populations.

Few records are available regarding the harvest of Galápagos tortoises by early explorers, buccaneers, seal hunters, and military vessels. British whalers began hunting sperm whales *(Physeter macrocephalus)* in the eastern Pacific in the early 1790s, and the Americans followed within a decade (Creighton 1995). Based on the logs of 79 whaling vessels that made 189 visits to the Galápagos, Townsend (1925) calculated a minimum take of 13,013 tortoises from the islands between 1831 and 1868. He stated that this represented only a small fraction of the overall take; there were in excess of 700 whaling vessels in the North American fleet at one time, and most of these made repeated visits to the Pacific. Based on these figures he estimated (as of the time of his writing) that following 1830 the total harvest by the North American whaling fleet was at least 100,000 Galápagos tortoises.

The first permanent settlement on the Galápagos, composed principally of political prisoners from Ecuador, was established in 1832 on Isla Santa María and was later moved to Isla San Cristóbal. These inhabitants survived principally on tortoise meat (Darwin 1845). The introduction of domestic animals to the Galápagos Islands also adversely affected Galápagos tortoise populations, and by 1838 the number of feral dogs, pigs, goats, and cattle had increased to such a number on Santa María that no Galápagos tortoises were found by a visiting vessel, which had to buy them at six shillings each on Isla San Cristóbal. By 1875, Galápagos tortoises were reported to be extirpated on Santa María, and numbers on three other islands (Española, San Salvador, and Santa Cruz) were reported to be so reduced that hunting had stopped (Van Denburgh 1914). Only on two of the larger islands (Isabela and Pinta) were tortoises still relatively common. Van Denburgh (1914) confirmed the extirpation of Galápagos tortoises on Santa María and Isla Santa Fé in 1905 and reported that many were killed on Isabela for meat and oil.

Galápagos tortoises were also used for the commercial production of oil. A group of hunters killing turtles for oil were reported on Isla San Salvador as early as 1835 (Van Denburgh 1914). Beck (1903) noted that hunting on the south end of Isabela was leading to the rapid decline of one of the last remaining populations. Oil hunters would camp near water holes during the dry season, kill the tortoises, and cut out their fat before moving on to a new site. When sufficient fat was collected it was "tried out" by cooking it in metal pots. Hunters concentrated on large tortoises, which yielded more oil (4 to 11 l each). At one site Beck (1903) reported finding 4.5 kl of oil.

The exploitation of tortoises for food and oil continued through much of the twentieth century. With the growth of the local fishing industry, Galápagos tortoises from many parts of the archipelago were taken for food. The island of Santa Cruz was settled during the 1920s, and in the following decade at least 1,000 to 2,000 tortoises were taken by oil hunters. Tortoises were also hunted for oil by workers at a salt mine on San Salvador. What was reported to be an almost untouched population of Galápagos tortoises on the southern volcanoes of Isabela were almost wiped out by oil-hunting occupants of a prison colony who operated until 1959 (MacFarland et al. 1974). Since 1959 most of the archipelago has been declared a national park, and although human consumption of tortoises has declined, it still continues in some areas (MacFarland et al. 1974; Swingland 1989a).

Giant South American River Turtle: Change in a Traditional Exploitation System

Throughout the Amazon and Orinoco river basins, river turtles (family Pelomedusidae) have long been a preferred food item of riverine communities (Brito and Ferreira 1978; Klemens and Thorbjarnarson 1995). Historic use by indigenous groups throughout the Amazon and Orinoco river basins centered on the largest species, the giant South American river turtle, and its eggs (N. J. H. Smith 1979). Prior to contact with Europeans, peoples living along the Amazon included river turtles as one of their dietary staples. Father Gaspar de Carvajal, who accompanied Francisco de Orellana on his fabled trip across the Andes and down the Amazon in 1541 to 1542, made reference to a large number of villages along the Amazon, and turtles were listed as the most common food item (Medina 1988). It was stated that in one village more than 1,000 turtles were found in enclosures and pools. Throughout the period of European colonization of the Amazon basin, turtles were noted to be an important dietary component of indigenous groups and colonists (Ferreira 1786; La Condamine 1992).

A well-regulated system of exploitation of turtles and their eggs was established on the major nesting beaches of the Amazon. N. J. H. Smith (1979) summarized several historical accounts of turtle exploitation in the vicinity of Itacoaitiara, Brazil. In the late eighteenth century, the giant South American river turtle was reported to be particularly abundant in the area and remained common through at least the 1850s. N. J. H. Smith (1979) estimated that at this time some 48 million eggs were harvested annually along the Amazon in the Rio Negro–Rio Madeira region. Oil extracted from the eggs was in much demand for use in cooking and as fuel for lamps. At the same time an estimated 2 million turtles were taken annually for food in the state of Amazonas.

The vivid accounts of Bates (1863) indicate that at that time the Brazilian upper Amazon supported large populations of South American river turtles. The giant South American river turtle was intensively exploited at all life stages. Although turtles were captured almost exclusively during the low-water dry season, in the mid-1800s residents of the town of Tefê lived almost year-round on turtle meat because every house in the village was reported to have a pen for holding turtles. Turtles were caught using a variety of techniques (e.g., seine nets and bow and arrow) from inland lakes or along the rivers during the dry season. Bates reported that the smaller yellow-spotted Amazon River turtle was used to a lesser degree at this time, not only because of its smaller size but also because it apparently did not live as long in captivity as did the giant South American river turtle and did not use the forest lagoons as readily as did its larger congener. Little mention was made of the consumption of the smallest of the region's species of *Podocnemis*, the six-tubercled Amazon River turtle, at the time.

In addition to the meat, turtle eggs were a valuable resource. Giant South American river turtles nest colonially during the dry season on elevated sand beaches. In perhaps one of the first attempts to manage the exploitation of a species, the excavation of giant South American river turtle eggs was controlled by Amazonian municipal councils based on a system established by the Portuguese governors more than a century before it was reported by Bates (ca. 1855). Each year the council of Tefê would appoint a commandante to supervise the excavation of eggs at each of four *praia reales* located within 240 km of the village. Sentries were posted at each beach to monitor the nesting of the turtles and protect the beaches from unauthorized egg harvesters. When the turtles had finished nesting, an announcement was made of the date for initiating the excavation of eggs at the nesting beaches. The commandante would record the names of the heads of households and collect a tax from each (the money was used to pay the beach sentinels). On a signal, all participants (Bates reported 400 at one beach) were permitted to begin digging up the eggs, which were tossed into canoes, mashed, and then placed in the sun to allow the oil to rise to the surface. This oil was then skimmed off, purified in copper kettles, and stored in jars. The oil was used for a variety of purposes, the most important being fuel for lamps. Bates estimated that 48 million eggs were destroyed annually in this fashion on the upper Amazon alone. This represents the reproductive output of approximately 400,000 nesting females.

Even during the period that Bates lived in the Brazilian Amazon (1848–1859) he reported that turtles were becoming scarcer and more expensive to purchase. Since that time continued exploitation has reduced populations of both the giant South American river turtle and yellow-spotted Amazon River turtle near Tefê to small numbers of animals of little commercial importance. The third species, the relatively small six-tubercled Amazon River turtle, is the only one still regularly captured in large numbers. These turtles are captured principally using long nets, *malhadeiras*, in bays along the rivers.

A similar chronology of exploitation occurred in the Orinoco River basin (Gumilla 1741; Humboldt 1859; Carvajal 1956; Castro 1986). Prior to the arrival of Europeans, a large indigenous population depended heavily on turtles, particularly the giant South American river turtle. According to accounts by early missionaries, Otomaca Indians organized the exploitation of turtles on the nesting beaches, assigning guards to minimize the disturbance to nesting females. After egg laying was finished, groups of Indians would congregate, some from considerable distances, to harvest the eggs. Aside from organizing the collection of turtles and eggs, this annual event had considerable economic and social importance for it facilitated trade and interaction among groups.

Although eggs, hatchlings, and adult turtles were eaten, the primary harvest was eggs. Throughout the Orinoco, indigenous groups anointed themselves with oil, usually at least twice daily for utilitarian (protection against insects) and social reasons (Gumilla 1741). Painting with dyes that used a turtle oil base was an extremely important part of the lives of all the indigenous groups along the Orinoco. The oil from turtle eggs was also used for cooking and as a hair cream. Enormous quantities of eggs were harvested, and the oil was prepared using methods very similar to those reported for the Brazilian Amazon. During the colonial period, two of the four principal Jesuit missions were established (in the 1740s) adjacent to turtle nesting beaches, and the missionaries took charge of the egg harvest (Castro 1986). According to Humboldt (1859), the Jesuits measured the size of the nesting area and set aside a part not to be harvested. Following the expulsion of the Jesuits in 1767, the Capuchin and Franciscan monks did not employ such an enlightened system, and the entire nesting beach was excavated.

The Europeans brought their own demand for turtle oil, principally as a fuel for lamps. The Capuchin monks reserved a section of the beach for the commons, the oil from which was used by their missionaries throughout southern Venezuela (Castro 1986). The monks, as well as local merchants, also purchased oil from the Indians. The traditional annual festival of Indian groups was soon changed into a seasonal market with buyers coming from all over Venezuela and even Trinidad to buy turtle oil and trade a wide variety of items. In the early 1800s, the Spanish Crown began demanding a tribute from the Indians in the form of oil, which was used for lighting in Angostura, the provincial capital. Following the war of independence, the new republic began poorly organized attempts to tax the production of oil. Rights to collect the oil tax were auctioned off to members of the commercially important families in the nearby town of Caicara. The winner of the auction would use their position for personal financial gain.

Although a decline in the production of eggs in the early 1800s had been noted by Humboldt (1859), enormous quantities of eggs were still taken throughout the nineteenth century. Control of exploitation during this period was spotty and was principally in the form of beach judges nominated by state authorities. Even this limited control seems to have completely broken down during the 1890s, with the large-scale capture and sale of adult females on the nesting beaches. Beginning at this time alternative fuel sources for lamps became available, and the principal objective of the exploitation shifted to the adult females, which were sold for meat. During the twentieth century, the traditional fair and market associated with the annual egg harvest disappeared, and people living along the Orinoco lost the opportunity to earn money from the harvest. However, the dwindling number of turtles was still exploited intensively, focusing almost entirely on nesting females. By 1945, the estimated number of turtles on the two principal nesting beaches was approximately 124,000. Despite attempts to regulate the harvest by the national government, by 1956 this number had been reduced to just over 24,000 animals (Ojasti 1973). In 1962, the sale of the giant South American river turtle was outlawed in Venezuela, but this control measure was largely ineffectual, and in the early 1990s the number of nesting females was just over 1,000 (Licata 1992).

Turtles are still consumed throughout both the Amazon and Orinoco river basins, but instead of providing a basic food staple to riverine communities it is now an expensive, and illegal, delicacy (Alho 1985). In Brazil, a large-scale government-sponsored effort to protect nesting beaches has resulted in a 1200% increase in egg production over the last 13 years (Cantarelli 1997). A more recent small-scale effort has been undertaken in Venezuela but has yet to result in any significant recovery of the population (Licata 1992).

Olive Ridley: An Attempt at Conservation through Controlled Egg Harvest

The controlled harvest and commercialization of olive ridley eggs from the Ostional National Wildlife Refuge on the Nicoya Peninsula of Costa Rica's Pacific coast is an example of a project that is attempting to conserve a natural resource through local use and community participation (Campbell 1998). The beach is the location of one of several large, synchronous nesting emergences *(arribadas)* of the olive ridley worldwide. During the nesting season, *arribadas* tend to occur

monthly over a 3- to 4-day period, and thousands of turtles emerge on the beach to lay their eggs (Richard and Hughes 1972).

The egg harvest program at Ostional is based on the premise that a harvest of a portion of the eggs during each *arribada* will not diminish current recruitment levels into the population. The probability that a female will find an area on the beach where another female has not already laid her eggs, and in which a subsequent nesting female will not dig her nest chamber, is density dependent. The more females in an *arribada* the greater the chance that a female will lose her clutch to intraspecific destruction of nests. Nest destruction is caused not only by intra-*arribada* competition for nest space but also by inter-*arribada* competition. The average incubation period for olive ridley eggs is 45 days; however, the monthly *arribadas* result in females from a subsequent *arribada* destroying eggs that have not yet hatched. In addition, a large percentage of egg clutches that survive to term do not hatch. Therefore, allowing the harvest of egg clutches at the beginning of an *arribada* will likely remove egg clutches that have a high probability of being destroyed by subsequent nesting females. Sea turtles and their eggs have been protected in Costa Rica since 1966 (Campbell 1997, 1998). Although sea turtles on the Pacific coast of Costa Rica are not killed for their meat, the demand for eggs of all species remains high.

The residents of Ostional and surrounding communities have harvested eggs for food and as a source of income from this nesting beach for decades (Cornelius 1982). According to Cornelius (1982), olive ridley eggs harvested from Playa Ostional have illegally supplied the Guanacaste Province and Central Highland markets of Costa Rica for years. Since 1971, the other Costa Rican *arribada* beach, Playa Nancite, has been protected through the establishment of Santa Rosa National Park. Nancite is geographically isolated, and as a result human exploitation of olive ridley eggs has probably been minimal (Cornelius et al. 1991).

Residents of Ostional have long advocated for a legalized subsistence and commercial harvest of turtle eggs (Cornelius et al. 1991). In 1977, a controlled harvest of eggs was proposed but subsequently denied (Cornelius 1982). In 1979, the Costa Rican rural guard began to patrol the beach at Ostional to protect the eggs, primarily against human and domestic animal predation (D. Robinson, personal communication *in* Cornelius 1982). In 1980, Cornelius and Robinson (1981, 1982, 1983, 1984, 1985) began a 5-year study to (1) evaluate nest survival and hatching success at both Nancite and Ostional, (2) examine the parameters that influence reproductive success at both sites, and (3) develop a management plan for Ostional. In late 1983, Ostional was declared a National Wildlife Refuge under a new wildlife conservation law. The new law included stricter fines and penalties for the illegal sale of wildlife products but, more importantly, allowed for exceptions to the prohibition on the sale of wildlife products under two conditions. These conditions were that human use would have to be justified through scientific study and a legal community-development association would have to be formed (Cornelius 1985; Cornelius and Robinson 1985; Campbell 1997, 1998).

Scientific support for the egg harvest program was based on a comparison of several parameters associated with egg hatching between Nancite, a rookery with little or no human predation of eggs, and Ostional, a rookery with a well-documented history of human predation. Cornelius et al. (1991) compared Nancite and Ostional and found that (1) Ostional had a significantly greater percentage of nests survive to term, (2) the average hatching rate of successful nests was significantly higher at Ostional, and (3) there was no difference between the percentage of nests that were at least partially successful. However, as Cornelius and Robinson (1985) and Cornelius et al. (1991) have pointed out, differences found between the parameters measured at the two sites could have been a result of other factors, such as variations in available nesting space and possible differences in the length of time during which *arribadas* have occurred at these two sites. Based on their findings, Cornelius and Robinson (Cornelius 1985) proposed an egg harvest program at Ostional as a solution to the then illegal and uncontrolled collection of eggs and the poor socioeconomic conditions of the area. Concerns were expressed that enforcement would be difficult and the legal egg harvest program at Ostional might provide an outlet for the illegal harvest of marine turtle eggs from other rookeries throughout the country. Proponents of the egg harvest program argued that the price of eggs from Ostional would be kept so low that the price of illegal eggs would be undersold. By law, the retail price of an olive ridley egg was not to exceed 50% more than the cost of a chicken egg (Araúz Almengor et al. 1993).

In 1984, Cornelius and Robinson (Cornelius 1985) recommended that (1) the egg harvest should occur during the first 24 hours from the onset of an *arribada* (>200 turtles on the nesting beach), (2) egg harvest for commercial purposes should occur only on the main nesting beach within the refuge, (3) the harvest of eggs from all species of marine turtles within the refuge but outside the main nesting beach should be prohibited, (4) Ostional residents should be allowed to harvest eggs for personal consumption but only from the main nesting beach, and (5) vehicles and horses should be allowed on the beach only during daylight hours. Eggs should be transported from Ostional in sealed (nonreusable) bags to bars and restaurants preapproved by the government for legal sale of eggs from Ostional (Cornelius 1985). All of these recommendations were incorporated into the harvest program.

By 1985, a community-development organization was established to operate the legal harvest of eggs. Since 1987, the organization has been known as the Asociación de Desarrollo Integral de Ostional (ADIO) (Cornelius 1985; Cambpell 1997). Only residents of Ostional are permitted to join the organization, although being a resident does not automatically entitle one to membership (Cornelius 1985). Campbell (1997) found membership requirements and regulations ambiguous.

By law, revenues from the sale of turtle eggs were to be divided between ADIO (80%) and the Costa Rican Departamento de Vida Silvestre (Department of Wildlife) (20%). The community association funds were to be used to pay egg collectors and to fund community-development projects. Government funds were to be used for constructing research facilities at Ostional, hiring biologists and guards, and implementing conservation programs (Cornelius 1985). However, there is no evidence that the government portion of the revenues was ever reinvested back into the egg harvest program or the community (Campbell 1997, 1998).

Since the original agreement, several changes have been made regarding the egg harvest, administration of the program, and distribution of revenues. According to Campbell (1997, 1998), the egg harvest period has increased from a 24-hour period to a 36-hour period from the onset of an *arribada*, and the 36-hour harvest period applies only during the wet-season months (May to December). There are no restrictions on egg collection during the dry-season *arribadas*, when neonate production is extremely low. The biologist at Ostional is now employed by ADIO, which now receives only 60% of the proceeds of the egg harvest. The government, represented by the Ministerio de Agricultura y Ganadera (Ministry of Agriculture and Livestock), now receives 40% of revenues.

According to Campbell (1997), the egg harvest is well organized. Eggs are packaged in plastic bags, sealed, and stamped to identify them as legally harvested from Playa Ostional. According to Araúz Almengor et al. (1993), the community initially distributed the eggs in the local market; however, because profits were low ADIO contracted a national distributor from 1989 at least through 1991. Campbell (1997) reported that eggs are now distributed by people from the community who are selected yearly to work a specified route. Theoretically, the selection of community sellers is based on economic need; however, there are complaints of favoritism. Drivers are also selected on a yearly basis, and theoretically the positions are rotated among community members. However, because drivers must own a vehicle or at least have access to one, the same people from year to year are selected. Conflicts often develop among the sellers because they earn their income based on the number of eggs sold. If illegally harvested eggs have met the market demand or the buyers on their route do not purchase as many eggs as usual, the sellers may encroach on other routes to sell their supply (Campbell 1997).

Capital improvements to the community through the sale of olive ridley eggs have been many, including (1) structural improvements to the school, (2) construction of a community center, (3) improvements to the soccer field, (4) construction of a basketball and volleyball court, (5) road improvements, and (6) electrification of the community. Many socioeconomic problems, however, still remain (Ordoñez et al. 1994; Ordoñez and Ballestero 1994; Campbell 1997, 1998; A. Chaves, personal communication).

The critical question that remains to be answered about the Ostional project is whether the egg harvest is sustainable. At present, this is unknown. Compared with the numerous socioeconomic reviews and evaluations, very little information has become available regarding the status of the nesting population since the studies of Cornelius and Robinson in the early 1980s. Only recently have methods been developed for even counting the number of females that nest during an *arribada* (Gates et al. 1996; Valverde and Gates 1999). Without an accurate record of the number of females nesting annually at Ostional over time (long enough for hatchlings that were produced since the harvest began to return to nest), recruitment rates, and thus sustainability, cannot be evaluated. Unfortunately, knowledge of the demography of olive ridley populations currently lags behind that of most other marine turtles. Similarly, the life history of olive ridleys is poorly understood, and thus rates of survival of the various life history stages remain unknown. A sustainable harvest will require a thorough understanding of all the mortality factors that affect the population both at the nesting beach and on distant foraging grounds (e.g., pelagic fisheries), and these factors are constantly changing.

Although sustainability is not known, it seems clear that the Ostional project confers better protection to the nesting population than would otherwise exist. Marine turtle eggs are being heavily exploited on the Pacific coast of Costa Rica, as they are elsewhere along the Pacific coast of Central America. What has not been evaluated—and what is of vital interest to marine turtle conservation on a broader scale—is the effect the legal Ostional egg trade has on the illegal trade of eggs of other olive ridleys and other marine turtles that nest in Costa Rica. The eggs of olive ridleys cannot be reliably distinguished from those of hawksbills and green turtles on the basis of size, and once the Ostional labels are removed, all appear the same. To enforce the regulations governing which eggs can be legally sold in the thousands of bars and street markets seems a daunting task. Illegal egg harvest has escalated on the Atlantic coast of Costa Rica in recent years, affecting both green turtles and leatherbacks. Whether the legal Ostional trade has stimulated trade in general or has reduced it by saturating at least a part of the booming market for eggs has never been evaluated.

The Ostional egg harvest represents an interesting and useful experiment in community participation in resource conservation, but its utility as a model for use with other marine turtle species is extremely limited. Only the Kemp's ridley and olive ridley nest in *arribadas*, and the Kemp's ridley is now restricted to one major breeding aggregation in the entire world. The other species almost never nest in the densities that occur in *arribadas*, and therefore the situation that exists

at Ostional—in which eggs are harvested that would otherwise be destroyed—does not exist. For all other species, harvesting would remove viable nests and could only be presumed to decrease recruitment rates.

Pancake Tortoise: Exploitation for the Pet Trade

The pancake tortoise is distributed in patches of rocky savanna–woodland habitat from central, possibly even northern, Kenya south to central Tanzania. When the first specimens of this bizarre, flattened tortoise became known to science in the 1920s, experts speculated that the tortoises were deformed or injured individuals of one of the known species of East African tortoises. Subsequent studies have shown that the pancake tortoise is a highly specialized land tortoise for which the bony elements of the shell have become reduced as an adaptation to living in rock fissures. The species is small, usually no more than 15 to 18 cm long, 10 cm wide, and 2.5 cm or less in height. Apart from being flat, the shell is soft and flexible, offering little protection from birds of prey, ground hornbills, secretary birds (*Sagittarius serpentarius*), or small carnivores, particularly genets (*Genetta* spp.) and mongooses. The pancake tortoise depends totally upon its crevice retreat for protection from predation and desiccation in the hot sun.

When a pancake tortoise ventures outside its protective crevice to eat grasses and succulents, it remains ever vigilant, moving rapidly back into its retreat at the slightest hint of danger. When threatened in its crevice, the animal withdraws its legs tightly into its shell, causing the soft central part of the plastron to balloon out, increasing the animal's height and wedging it between the top and the bottom of the rock fissure. Availability of suitable crevices may be the single most important factor limiting the size of pancake tortoise populations. For example, within each habitat surveyed, the number of crevices of suitable depth and dimensions was only a small percentage of all the available natural crevices (Klemens and Moll 1995; D. Moll and Klemens 1996).

During the 1980s, pancake tortoises began to appear with increasing frequency in the wildlife trade, finding their way into pet shops across the United States, Europe, and Japan. Although the tortoises were initially quite expensive, over time, as the numbers of pancake tortoises in the trade increased into the thousands, prices began to drop dramatically from a high of perhaps \$300 to as low as \$30 per tortoise. And when the pancake tortoises became inexpensive, they were purchased as novelty items by people who lacked both the knowledge and the commitment to care for them properly.

Pancake tortoises have been listed on Appendix II of CITES since 1975. However, as there was little information on this species' geographic distribution, population size, life history, and ecology, the effects of trade could not be properly evaluated. In 1991, Dutch customs officers at Schiphol Airport intercepted a shipment of several hundred pancake tortoises bound for the United States (see more at "Repatriation, Relocation, and Release Programs," McDougal, Chapter 7). The crate was packed so tightly that many of the tortoises were crushed by the weight of other tortoises on top of them. This well-publicized seizure helped focus attention on the trade in tortoises and other reptile species originating from Tanzania. The CITES Animals Committee added the pancake tortoise to its list of significantly traded species that are listed on Appendix II and urged a complete assessment of the trade and its impact on wild pancake tortoise populations.

In 1992, a study was initiated to investigate the scope and impact of trade on Tanzania's wild populations of pancake tortoises (Klemens and Moll 1995) and to gather data on the ecology and life history of this rupicolous species (D. Moll and Klemens 1996). It was discovered that accessible areas in north Tanzania had been severely depleted. Pancake tortoises had become scarce, even eliminated, at sites where collectors had previously been active. Habitats lying within easy access of the road between Arusha and Dodoma had been heavily collected, and populations there were in serious jeopardy. Collectors had destroyed rock outcrops and crevices, using car jacks to pry the rocks apart to reach the reptiles. Collectors had even poached pancake tortoises by means of this technique within Tarangire National Park. Collectors readily admitted that pancake tortoises were becoming more difficult to find because they had taken so many. To compound this tragedy, the money that the collectors received for their efforts was minimal, often barely enough to buy a few soft drinks or cigarettes.

These disturbing findings were reported to the Tanzanian government. Officials were informed that in less than 10 years of intensive collection, the pancake tortoise had become severely threatened throughout much of its range in Tanzania. Three options for the management of this species were presented to the government: (1) continued uncontrolled trade, which would deplete as-yet-untapped pancake tortoise populations while reaping low economic returns, particularly at the local level, (2) strictly regulated trade coupled with a substantial export tariff, perhaps providing the capital and the incentive to manage and conserve the country's pancake tortoise populations, though possibly encouraging illegal trade, and (3) the option the Tanzania government decided to adopt, a moratorium on pancake tortoise trade exports. Because this moratorium must be enforced by the more than 100 countries that are signatories to CITES, controlling the trade became a joint effort, not solely the responsibility of Tanzania. The importing countries, primarily the United States, were now legally mandated to refuse entry of pancake tortoise shipments originating from Tanzania. The trade moratorium is now entering its fifth year, which is good news indeed for the pancake tortoise.

However, in 1997 new attempts were made to revive the pancake tortoise trade by means of exports destined for the United States "originating" from northern Zambia, far outside the range and natural habitat of this species. Apparently, pancake tortoises are now being smuggled across the southern border of Tanzania for reexport from adjoining countries. The "laundering" of Appendix II reptiles, through extralimital countries, presents new challenges to the enforcement of CITES and other laws that protect wildlife from exploitation.

DISCUSSION

It is abundantly clear that the human use of turtles is widespread and can have significant effects on the status of wild populations. The presence of large numbers of easily harvested turtles has probably played a significant role in the history of human enterprise and endeavor. Pre-Columbian and early European populations in the Amazon and Orinoco river basins subsisted to a large degree on turtle meat. Carr (1954) commented that the exploration and colonization of the Caribbean was, to a large degree, facilitated by protein availability in the form of sea turtles. Long ocean voyages were made possible by stocking giant tortoises from the Galápagos and islands in the western Indian Ocean. The use of turtles for medicinal purposes also has a long history, which has been exacerbated in recent decades by increased long-distance trade, a result of trends in the globalization of the world economy. An international market for turtles as pets is a relatively recent development, but for certain species this trade has had major impacts on wild populations.

The collection of both eggs and adult turtles has had drastic effects on turtle populations worldwide. In some areas the exploitation of turtles has resulted in the extirpation of local populations; in some cases, exploitation has resulted in extinction. Within historical times, populations of sea turtles in the Cayman Islands, Bermuda, the Dry Tortugas, and the Mascarenes have been extirpated. Tortoises in the Mascarenes were also extirpated soon after humans colonized the islands. Although not as significant globally as the exploitation of turtle populations for food or medicine, the pet trade is a major, if not the primary, threat to certain species (see the case study regarding the pancake tortoise). The effects of overcollection are amplified in rare species that have specialized habitats, like the pancake tortoise (Klemens and Moll 1995) and bog turtle (USFWS 1992a).

The exploitation of turtles has been well documented; however, the effects of human use on wild populations is difficult to evaluate in a quantitative fashion (see Doroff and Keith 1990). Much available information concerns colonial-nesting species such as sea turtles and certain freshwater species such as the giant South

American river turtle and river terrapin. For these species the number of nesting females can be used as an index of population size, and data for many of these species demonstrate precipitous population declines. It is also true that predictable patterns of nesting in time and space make these same species extremely vulnerable to human exploitation. Nevertheless, many biological and human factors play roles in determining which species are more likely to be exploited and how this exploitation will affect wild populations.

Biological Factors in Turtle Exploitation

Several features of the biology of turtles have facilitated their exploitation by humans. Like all reptiles, turtles have low metabolic rates and reduced energy requirements when compared with endotherms (Pough 1980). As a result, turtles have high production efficiencies and are often found at much higher biomass levels than are mammals or birds (Iverson 1982). The ecological consequences of reptilian metabolism play an important role in determining the number and rate at which turtles can be taken sustainedly. Another consequence of their physiology is that turtles are able to survive long periods without food and, in some cases, water. Before the advent of refrigeration, turtles were one of the few animals that could be collected during periods of seasonal abundance and kept alive for long periods with minimal care. In many rural areas this is still an important factor in the exploitation of turtles.

Although there is considerable variation within the group, turtles are characterized by a coevolved suite of life history characteristics that typically include slow, indeterminate growth, delayed sexual maturity, and a long reproductive life span (Gibbons 1987; Wilbur and Morin 1988; Congdon et al. 1993; see "Demographic Issues in Turtle Conservation," Gibbs and Amato, Chapter 8). These life history traits place biological constraints on the levels of harvest that turtle populations can sustain. Recent studies of a number of North American species have drawn attention to the consequences of longevity and delayed reproduction (Doroff and Keith 1990; Congdon et al. 1993). These analyses have demonstrated that turtle populations are very sensitive to increases in mortality of adults and large juveniles (Crouse et al. 1987; Congdon et al. 1993). Chronic reduction in the survivorship of adult turtles would require an increase in the survivorship of eggs or juveniles or density-dependent increases in fecundity (by increasing clutch size or clutch frequency or decreasing age at sexual maturity) to maintain a stable population size (Congdon et al. 1994). However, even significant increases in the survivorship of eggs and juveniles are unlikely to compensate for increases in mortality of adults (Heppell et al. 1996). Given the need for high rates of survivorship in large juvenile and adult life stages, these studies have questioned whether any level of turtle harvest can be sustainable.

Another general attribute of turtle life histories is high rates of egg and hatchling mortality (Wilbur and Morin 1988; Iverson 1991a). This suggests that harvesting systems based on the collection of eggs are less likely to have negative impacts on the population than those based on killing adults. In fact, many of the long-term, traditional turtle exploitation systems were based on harvesting eggs (e.g., see the case study regarding the giant South American river turtle). However, although some of these traditional systems existed over a period of centuries, there is little evidence that they were sustainable. In fact, in the case of the giant South American river turtle there are strong indications that egg collecting was so intense it significantly reduced populations of this species. Although this system was certainly less damaging than the large-scale collection of nesting females (which occurred later and devastated populations), some early accounts decried the shortsightedness of egg exploitation. What is clear is that the growth in demand for eggs associated with European colonization, the local shift to a cash-based economy, and the taking of nesting females doomed the traditional system.

Societal Factors in Turtle Exploitation

Religious beliefs and cultural factors play an important role in shaping the patterns of human exploitation of turtles. In some cases these factors have played a key role in limiting the consumption of turtles, whereas in others they have promoted it. For example, throughout Central and South America turtles are classified as fish by the Roman Catholic Church, creating a traditional seasonal increase in the consumption of chelonians during the week before Easter when eating meat is discouraged. In Papua New Guinea, villagers who have become Seventh Day Adventists have stopped eating turtles and report increases in nesting sea turtles. The unsubstantiated belief that turtle eggs act as an aphrodisiac has led to their widespread consumption in bars throughout Central America. Religious considerations prevent higher castes (e.g., Brahmins) from eating turtles in Nepal (Shrestha 1997). The prohibition on the consumption of turtle meat by Islam has certainly been an important factor in reducing the exploitation of turtles in parts of India and the Middle East, as well as in Bangladesh, Indonesia, and Malaysia. Despite intense human population pressures, freshwater turtles are still abundant in Bangladesh, a fact that Das (1990) attributes to 90% of the population being Muslim. Nevertheless, no religious restrictions are placed on eating turtle eggs or catching and selling turtles domestically or for export. In recent years exports have increased to meet changing regional patterns of the consumption of turtles for both food and medicine.

The widespread consumption of turtles in China and Southeast Asia is intimately tied to cultural beliefs (Jenkins 1995). In China turtles are classified as a "hot" food that strengthens the body during the winter. Eggs are believed to be aphrodisiacs, and turtle blood is thought to provide an energy boost. The shells of turtles are believed to have a wide variety of medicinal uses in Chinese cultures (Jenkins 1995). The carapace of softshell turtles is used to produce *bie jia*, which is used in a variety of forms including powders and jellies, to treat problems of the kidney, spleen, and liver (Jenkins 1995). The plastron of hard-shelled species is used to produce *gui ban* for treatment of the heart, liver, and kidneys. Recent changes in regional economic systems have also resulted in the increased ability of the Chinese to purchase imported turtles, and this expanded market has important implications for the conservation of chelonian biodiversity, particularly in southern Asia.

Aside from cultural influences, common patterns in the human use of natural resources have tended to result in overexploitation. As human populations have grown, turtle populations have decreased due to a variety of human-related factors, principally habitat destruction and overexploitation. With time, the general pattern of exploitation has been one of growing demand and dwindling supply of a resource that is typically considered to be open, or accessible, to all members of the community. The common ownership of a resource such as turtles makes managed use problematic, as perhaps has been best promulgated by Hardin's (1968) *The Tragedy of the Commons*. Hardin's basic thesis is that as a collectively owned resource begins to decline due to overexploitation, individuals will compete for a larger share of the dwindling supply, and the ultimate demise of the stock can be seen as a logical result. In the example of a community harvesting turtles from a lake, each person receives a direct benefit from each turtle captured and suffers delayed costs from the deterioration of the turtle population. As the number of turtles declines, each person is motivated to put more effort into capturing turtles because each one obtains a direct benefit from the turtles caught but bears only a portion of the cost of the declining population, which is shared among neighbors. Hardin (1968) summarized by stating, "Ruin is the destination towards which all men rush, each pursuing his own best interest in a society that believes in the freedom of the commons."

However, this argument ignores the presence of cultural values and social institutions and their role in regulating the use of common property (McGoodwin 1990; Gadgil et al. 1993). For turtles, there is some evidence of cultural controls in traditional harvest systems. The historic cases of green turtles in Sarawak and river terrapin in Malaysia, and the contemporary example of olive ridley eggs in Costa Rica, are instances in which attempts have been made to regulate harvests. Conversely, the case of regulated South American river turtle egg harvests in the Brazilian Amazon appears to have been specifically designed to maximize the number of eggs collected, with no regard whatsoever to the concept of sustainability.

Cultural and economic changes have resulted in the loss of traditional systems. Inhabitants throughout the central Pacific Ocean have harvested marine turtles for thousands of years, and green and hawksbill turtles supply the basic needs of these communities (Daly 1990). Turtle hunting helped to pass on traditional knowledge that has formed the basis of rituals, taboos, and ownership rights that regulated the levels of harvest in the past. The breakdown of traditional practices was a result of the introduction of cash-based economies, the decline of traditional authority, and the imposition of colonial laws and practices (Balazs 1982a; M. A. McCoy 1982), and marine turtle populations are reported to have declined within historical times (Balazs 1982a). M. A. McCoy (1982) cited an increase in human populations and the preference for modern boats and motors as other factors that have contributed to the decline in marine turtle populations, and he emphasized the need for a conservation system to replace the traditional taboos and social restrictions. Another example is the Kiwai people of Papua New Guinea, who hunt green turtles for ceremonial uses and for trade. Prior to the transformation to a cash-based economy, the Kiwai people viewed green turtle meat as a resource to be shared among family and kin relations and to establish reciprocal obligations. With a cash-based economy, green turtles are now viewed as individual property and consideration of cultural obligations are ignored (Eley 1989).

Options for Management of Turtles

With increasing pressures being brought to bear on turtle populations, and the loss of traditional systems that may have regulated harvest levels in the past, new ways must be found to manage turtle populations. Solutions to the overharvest of common resources are usually framed in two major contexts, these being governmental regulation of exploitation or privatization (Ostrom 1990; Hardin 1994). Both approaches can offer substantial pitfalls, including the failure to take local social institutions into consideration when designing management programs and the assumption that privatization will lead to sustainable management.

Instances of governmental regulation of turtle exploitation are usually limited to total prohibitions, which in many cases are unenforceable. Bermuda is an example of what was perhaps the first-ever protective legislation for a turtle (in 1620), as well as the first case of a green turtle rookery to be extirpated. In most instances, restrictions can reduce large-scale commercial harvests, but subsistence consumption and small-scale commercial sales continue. Virtually all documented attempts to regulate a managed harvest have been related to the collection of eggs on traditional nesting beaches. However, due to the demographic importance of adults, protective measures or managed collection schemes involving eggs can be ineffective if high rates of human-related adult mortality are not addressed (Crouse et al. 1987; Heppell et al. 1996). Attempts by the Mexican government to regulate the commercial harvest of olive ridleys by permitting the harvest of adults while protecting nesting beaches were a dismal failure. In some cases, exploitation is permitted for subsistence purposes by certain cultural groups. In Australia, a national ban on killing sea turtles is in place with the proviso that aboriginal groups can still legally harvest them for subsistence. If harvest levels are low enough, off-take may be sustainable, but there are no cases known for which enough information is available on the population ecology of the turtles and exploitation levels to allow the evaluation of the sustainability of the harvest. At the international level, CITES and national trade restrictions on turtles in the pet trade have shifted market demand from one species to another, where similarly unsustainable levels of exploitation occur.

For some, privatization is a means of eliminating some of the perceived problems with common property resources. However, the managed harvest must have enough built-in controls to ensure that harvest levels are sustainable and that the benefits of the harvest accrue to a wide segment of society. In the case of the giant South American river turtle in Venezuela (see the case study above), control of the resource was managed by the upper socioeconomic strata of one town, principally for their own financial gain (Castro 1986). This led to a situation described by May (1992) as the "tragedy of the non-commons," where "resources have been privatized so as to curtail benefits obtained through common management, and those excluded are denied compensation due to lack of either bargaining power or of legal legitimization of property rights." The program developed to manage the giant South American river turtle population in Venezuela in the eighteenth and nineteenth centuries was loosely based on the pre-Columbian system but was controlled by individuals removed from the resource. This resulted in the economic marginalization of the local communities, which had traditionally managed the turtle beaches. Nonetheless, approaches like the one at Ostional (see the case study above) offer hope that at least under certain circumstances community institutions, governmental organizations, and scientists can work together to discuss the need to limit harvest levels. Although the case of harvesting eggs from *arribadas* of ridley turtles is perhaps the most extreme example of this, it can provide useful lessons as to how, and how not, to implement community-based egg collection programs. It is clear, however, that the situations most amenable to the development of managed-use programs are those involving egg collection at mass-nesting events. Whether programs can be developed for the sustainable use of eggs of solitary nesters, the collection of live animals for pets, or the harvesting of adults for meat has not yet been addressed.

The sustainable use of wildlife has been widely promoted as a practical means of promoting conservation (McNeely et al. 1990). Conservation efforts have focused to a large degree on the declaration of protected areas in tracts of sufficient size to protect natural resources from human use and habitat alterations. However, this approach often has neglected the needs of the human communities that live in the area, and communities surrounding protected area often have found that their customary areas for resource exploitation have been incorporated into the protected area without their consent or consideration, at times leading to volatile situations between protected area managers and local inhabitants.

In recent years, several authors have argued to include local communities in natural resource management and to allow local inhabitants continued, controlled access to the resource (Bodmer 1994; J. G. Robinson and Redford 1994). It is believed that conservation efforts will have a much better chance of succeeding if local people are included in initial planning discussions, as well as in the implementation and management of the protected area. In these cases the local population must be allowed to benefit directly or indirectly from the resource. If communities choose to be involved, they not only must be allowed to participate in the conservation of local resources but should be expected to share in the responsibility of maintaining local biodiversity.

The challenge of creating sustainable-use management programs for turtles is daunting. There is very little biological or economic foundation upon which sustainable-use programs for turtles can be based, and virtually no case studies have evaluated harvest programs. The past history of turtle exploitation has clearly been one based on short-term economic gain. Although the rapid overexploitation of a resource such as a turtle population may seem extremely shortsighted, it can also be viewed in economic terms as a rational course of action. The key is understanding the effect of discount rates on the present value of future earnings from a renewable resource. As shown by C. W. Clark (1976), the best economic and biological strategies of harvesting are the same only when the exploited population's growth rate is high. Capital "expands" at an annual rate of approximately 10% (Caughley and Gunn 1996; C. W. Clark [1976] suggested a lower rate of 2 to 4%). Therefore, harvesting wild animals sustainedly pays only when a population's growth rate following harvest exceeds 10% per year. Otherwise, a purely economic analysis of the situation would recommend a large "capital reduction" (overharvest) and investment of the earnings in another enterprise with a higher rate of return. The almost universal status of turtles as a common-property resource, and the relatively low economic return of a sustained yield from turtle populations, creates problems for the concept of commercial-use management of these animals. As long as the return on investing profits from present overharvesting exceeds the expected return on future (sustainable) harvests, and as long as profits accrue to the individual and costs are spread among society, the overexploitation of turtle populations is inevitable.

It is clear that before we can evaluate the potential of sustainable-use management of turtles as a conservation tool, we need three things: (1) a better understanding of the population biology of those species harvested, (2) more information on the resource economics of sustained-yield harvesting, and (3) case studies that evaluate trial sustained-yield harvest programs.

The development of management efforts based on sustainable use is relatively new, and most efforts have suffered from a long series of technical, political, and institutional impediments (Ojasti 1995). As we have seen, a wide variety of biological and human factors play significant roles in exploitation and how it affects turtle populations. Sustainable-use programs involve a complex milieu of biological, economic, sociological, and political factors that need to be addressed for each individual case (see also Seigel and Dodd, Chapter 9). Implementation of such a program requires a multidisciplinary effort rarely found in most conservation initiatives. For instance, programs must be able to evaluate the potential levels of harvest in terms of economic benefits for the various program stakeholders. Aside from generating economic incentives for local communities to protect turtles and turtle habitat, the program should, ideally, generate revenues (through taxes and user fees) to the governmental entity responsible for program oversight. These fees would be used to support the enforcement of program regulations as well as monitoring of the program to measure the effects of harvest on the turtle population.

From a theoretical standpoint, sustainable-harvest programs of wild populations, and ranching programs that invest in rearing individuals but still rely on wild stock to maintain the operation, can give direct economic justifications to maintaining wild populations. A ranching effort, for example, can involve selling hatchlings as pets, and this is the basis of an egg collection program for the Nicaraguan slider (*Trachemys scripta emolii*) in Caño Negro Nature Reserve in Costa Rica (Pritchard 1993). Eggs collected from the wild are incubated, and the hatchlings are sold as pets in San José. The local egg collectors receive 50% of the sale, thus providing local communities with direct economic benefits from the wild population and incentive to protect this resource. For such a program to avoid overexploitation of the population, collection levels have to be below natural replacement levels.

Farming of turtles has been suggested as a means to produce food and to remove hunting pressure on wild turtles. However, a completely closed-cycle farming program (captive adults produce all future generations) does not need any wild stock and, therefore, does not offer the economic incentives to conserve wild populations or habitat that are inherent in programs based on the sustainable use of wild populations. Nonetheless, the conservation benefit of closed-cycle farming,

or captive breeding, is that it is possible to reduce, and even replace, the need to use wild-caught specimens. This reduction has been seen in other wildlife utilization programs, such as the large market for two of the most common pet birds, budgerigars (*Melopsittacus undulatus*) and canaries (*Serinus canarius*), which are completely supplied by breeding operations (Bolze 1992). Captive breeding may have the potential to meet the demand of the market for rarer turtle species, and captive-breeding efforts have increased in recent years to supply this market as the wild-caught supply has declined due to trade controls or reduced population levels. Trade controls and reduced wild populations have driven up prices, making investments in captive breeding economically attractive (Smart and Bride 1993; HSUS 1994). Commercial farming or breeding programs for rare and threatened species could be required to contribute a portion of their economic proceeds toward conservation efforts as a mechanism for these programs to benefit wild populations.

Whitaker (1997) has proposed a village pond-rearing system for the Indian flapshell turtle as a means of producing meat for sale and removing hunting pressure on wild populations in India. In Thailand, Jenkins (1995) reported that approximately 15 turtle farms were commercially breeding and rearing Chinese softshell turtles, though he does not specify if these farms are really closed-cycle operations. This small species of softshell appears to be particularly well adapted to captive farming, and total production is estimated at 3 to 6 million hatchlings per year ($3 to 6 million annually). The industry is expanding quickly, and Jenkins (1995) cited one source as saying there could be 100 farms by 1996. Attempts to breed native softshells have met with little success, and as softshells native to Thailand become more scarce, the captive-bred species is more common in markets. Chinese softshell turtles are also being commercially farmed in Malaysia (Jenkins 1995). In Shanghai, China, there are reported to be over 100 farms rearing Chinese softshell turtles for the domestic pet trade and sale to medicinal companies (Cen Jianqiang, personal communication).

In Brazil, the sale of meat from farm-reared giant South American river turtles and yellow-spotted Amazon River turtles has been authorized by the government. Farms obtain hatchlings from a government-run program that protects nesting beaches and annually releases millions of hatchling turtles (IBAMA 1989). However, it is unclear what, if any, direct conservation benefits will accrue to the wild turtle populations from the sale of captive-reared individuals for food.

Farming operations may not be generally applicable to a wide range of turtle species or market uses. Although they may be economically viable for some uses (e.g., red-eared slider hatchlings as pets), turtles tend to grow slowly, and their meat usually has a relatively low market value. Also, given the difficulties of rearing captive animals, the capture of wild individuals may be hard to discourage. The experience in Thailand suggests that the husbandry of many species is difficult on a large-scale commercial basis. Jenkins (1995) doubted that captive breeding would remove much hunting pressure on wild stocks while viable populations remained. Alho (1985) proposed a system for the giant South American river turtle, combining annual releases of headstarted animals (see "Headstarting," E. O. Moll and Moll, Chapter 5) and the harvest of captive-reared turtles after 8 years, but trial programs have suggested that this scheme is not economically viable. For sea turtles, Dodd (1982a; see also Seigel and Dodd, Chapter 9) concluded that farming programs would detract overall from conservation goals. Although breeding of certain species that do well in captivity is an option for supplying turtle meat markets, the risk of introducing exotic species from farming operations becomes an additional concern.

The value of sustained-use programs as a conservation tool for turtle populations is difficult to evaluate at this time. Very little biological information is available for most species, and in very few cases is there any type of quantitative information on levels of harvest and the effects of harvest on wild populations. Nevertheless, turtles are, and will continue to be, widely used for food, medicinal purposes, and as pets around the world. In most cases these uses will have negative effects on wild populations. It is clear that one of the greatest challenges that faces turtle conservationists is to understand these patterns of use, the effects they have on wild populations, and how the use and intrinsic value of turtles can be used as a tool for their conservation.

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JOSEPH FLANAGAN

3

DISEASE AND HEALTH CONSIDERATIONS

Chelonians are increasingly threatened throughout the world by habitat loss and fragmentation, the introduction of exotic species, and exploitation by humans for food, medicine, and the pet trade. In free-living animals, disease is considered to be an important factor in natural selection (Bush et al. 1993). It is a factor that has likely always played a role in turtle population dynamics but which has only recently become a threat to the survival of species. Whereas historically, infectious disease may have played a role in the demise of localized populations of chelonians, a rebound of those populations could occur through the increased survivorship of progeny from surviving individuals or the migration of individuals from adjacent populations or geographic areas that may have been unaffected by the specific disease process. Currently, many populations are so small or so isolated that recovery after a disease outbreak is not likely. Adjacent populations may have been eliminated due to habitat fragmentation or other factors, resulting in little or no potential for migration from a previously contiguous range. Surviving individuals within a population may be unable to reproduce the population due to factors such as predation, limited nesting or feeding habitat, lack of access to migration corridors, and stochastic demographic, genetic, or environmental processes.

Increasing human activity in chelonian habitats is resulting in new impacts of disease on wild populations. In the Galápagos Islands the introduction and establishment of populations of exotic mammal species has dramatically influenced vegetational structure and interfered with the ecology and behavior of native tortoise populations (Swingland 1989a). These changes can lead to nutritional disease processes or increase an animal's susceptibility to disease-causing organisms. The suspected release of ill tortoises in the southwestern United States may have reits local environmental and zoning ordinances by creatively planning to protect natural resources through the creation of open-space reservations that correspond to ecosystem realities, not arbitrary legislative mandates. To create these open-space reservations, the town planner, as well as the planning and zoning boards, sought out technical expertise from scientists to gain knowledge of ecosystem functions within the community. This knowledge provided them with not only the information upon which to create these open-space reservations but the ability to communicate effectively the rationale behind those planning decisions to other town officials, the electorate, and the development community.

Last, but by no means least, I see an urgent need to expand and enhance avenues for formal and informal public education regarding two key issues: we need to address the limited, often disconnected view that many people have of how ecosystems function, and we must begin to dispel the widely held notion that a community must choose either conservation or economic development. A better understanding of the full range of options that communities can exercise to balance environmental stewardship with economic well-being is required. These educational efforts should focus not only on appreciating the complexity of ecosystems and the effects of cumulative impacts upon them but on evaluating the short- and long-term costs and benefits of various types of land use scenarios. In terms of understanding the spectrum of choices available, curricula for the general public and decision makers could be developed that make a clear distinction between economic growth, measured by short-term productivity, and more sustainable development, measured by a set of qualitative standards that place a higher premium on long-term benefits, such as overall quality of life.

CONCLUSION

Today we face a tremendous challenge—to save the world's 260 or more species of turtles from the specter of extinction. I remain optimistic that we can build upon the experiences of the past, using those hard-learned lessons to chart a more secure and sustainable future for the world's turtles. This is the driving force that motivates those who seek, through this book and other avenues, to reorient our approach to turtle conservation.

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