CHAPTER 1. GENERAL INTRODUCTION

1.1 CHAPTER SUMMARY

In this thesis, I investigate the nature of sea turtle by-catch, the response of sea turtles to trawl capture and the relative distribution of sea turtles in order to spatially assess priority areas for the monitoring of compulsory Turtle Excluder Devices (TEDs) and the conservation of critical sea turtle habitats. This thesis also contributes to the knowledge of sea turtle biology by improving the understanding of factors that influence their spatial distribution. These aspects contribute to the scientific basis for a comprehensive approach to the sustainable management of sea turtle by-catch. This will also enhance the ability of humans to manage anthropogenic impacts on sea turtle populations at much larger spatial scales than is possible with the data that are currently available.

1.2 BACKGROUND

1.2.1 Sustainable Development of Marine Resources

Humans have always exploited marine resources to support their needs for food and other items. However, during the 20th Century our ability to exploit marine resources, particularly via fishing activities increased dramatically (Grainger and Garcia 1996). The current size and geographic extent of fishing has raised serious concerns about the sustainability of present exploitation rates (FAO 1997; Caddy and Cochrane 2001). The overexploitation of many fish stocks (FAO 1997) and the recognition that fishing impacts species and communities beyond the target species (Dayton *et al*. 1995; Goñi 1998; Hall 1999) have contributed to a change in fisheries management philosophy. It now well recognised that the social and economic welfare of human beings relies on the maintenance of ecological systems and their biodiversity (WCED 1987; CFWG 1991). The term 'sustainable development' encapsulates these ideals, and whilst there are many definitions of this term, all contain the same essential concepts, which are to:

(i) Enhance individual and community well-being and welfare by following a path of economic development that safeguards the welfare of future generations;

(ii) Provide for equity within and between generations; and

(iii) Protect biological diversity and maintain essential ecological processes and life support systems (CoA 1992).

Fisheries management has embraced Sustainable Development by incorporating its concepts into international conventions and policies such as the FAO Code of Conduct for Responsible Fisheries (FAO 1995). Under the principles of Sustainable Development, fisheries managers must balance the efficient exploitation of fisheries resources while maintaining the integrity of the ecological system on which the resources depend (Bergin and Hayward 1995; Maynes 1995). These objectives are often perceived as being in conflict (Chesson *et al.* 1999).

In Australia, the concept of Sustainable Development is referred to more commonly as Ecologically Sustainable Development (ESD). The principles of ESD have been incorporated into Australian fisheries legislation at Federal and State levels as fisheries management objectives. Continual improvement in documenting the sustainable management of Australian fisheries is a requirement of the Commonwealth *Environment Protection and Biodiversity Conservation Act* 1999 (EPBC)[1]. In combination with the Commonwealth *Wildlife Protection Act (Regulation of Exports and Imports*) 1982 (WP(REI)), Australian fisheries can be prevented from exporting their products if sustainability criteria are not met.

The Commonwealth Fisheries Working Group on ESD identified overfishing as the major threat to the ecological sustainability of Australian fisheries resources (CFWG 1991). The Group also considered that the impact of fishing on the environment is a major issue that potentially affects the ecological sustainability of fisheries resource exploitation. The direct effects of fishing include the mortality of target species, as well as the catch and mortality of non-target species, and the physical disturbance and destruction of marine habitats (Goñi 1998; Hall 1999; Harris and Ward 1999). Indirect effects of fishing deal with the ecological consequences of the direct effects of fishing (Alverson *et al*. 1994; Hall 1999). They include predation and competition resulting from the removal of some species by fishing (i.e., changes in community structure and trophic cascades), the environmental effects of discards (i.e., diet supplementation enabling some populations to increase) and the continued effects of discarded or lost fishing equipment (i.e., ghost fishing).

1.2.2 By-catch

The most obvious and politically prominent effect of fishing is the capture and mortality of non-target species, often referred to as by-catch (Hall 1999).

Definition

There are multiple definitions of the term 'by-catch' in the scientific literature (Table 1.1), but a common theme is that by-catch is that part of the catch taken incidentally to the target catch and returned to the sea as a discard. To a large degree, this common theme addresses the key social, political and environmental concerns about by-catch that are: (i) the impact of incidental mortality on populations of by-catch species; (ii) the 'wastefulness' of the practice; and (iii) the ecosystem effects (Harris and Ward 1999).

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[1] In January 2001, the EPBC Act was revised to incorporate the WP(REI) Act, so that the ecological assessment and approval of export permits is covered under a single piece of Commonwealth legislation.

Table 1.1 Definitions of by-catch

Source	Definition
Saila 1983, p. 1 Andrew and Pepperell 1992, p. 528	"That part of the gross catch which is captured incidentally to the species toward which there is directed effort. Some, all or none of the by-catch may become the discard catch."
Alverson *et al.* 1994, p. 6 Hall 1999, p. 17	"Discarded catch plus incidental catch", where discarded catch is "that portion of the catch returned to the sea as a result of economic, legal or personal considerations" and incidental catch is "retained catch of non-targeted species".
Hall 1996, p. 322 Hall *et al.* 2000, p. 206	"That portion of the capture that is discarded at sea dead (or injured to an extent that death is the most likely outcome) because it has little or no economic value or because its retention is prohibited."

Scale

All fisheries incur some level of by-catch, but the highest discard rates are consistently reported for prawn (= shrimp) trawl fisheries (Saila 1983; Andrew and Pepperell 1992; Alverson *et al.* 1994). The proportion of the total catch that is by-catch compared to that which is the target catch in prawn trawl fisheries varies seasonally and spatially, but has been generalised at 5:1 (by-catch:target catch) for temperate latitudes and 10:1 for tropical latitudes (Allsopp 1982; Andrew and Pepperell 1992). Discards are highest in tropical prawn trawl fisheries because they occur on continental shelves in tropical waters where there is a great diversity and abundance of invertebrates, bony fish and other organisms (Andrew and Pepperell 1992). High discard rates occur because only a few target species plus some incidental species (=non-target catch) are retained. Most of the catch is discarded back into the sea in a dead or dying state (Wassenberg and Hill 1989; Hill and Wassenberg 2000). In this situation, trawling is a highly non-selective fishing method. By-catch can be comprised of numerous species in tropical prawn trawl fisheries, including molluscs and crustaceans, bony fish and larger species such as stingrays, sharks and sea turtles. This thesis focuses on sea turtles, which are a group of species with a global by-catch problem in many demersal trawl fisheries occurring in tropical and sub-tropical waters.

Sustainability

The scale of by-catch mortality for a species varies from small to extremely large, but the direct ecological consequences of by-catch mortality depend upon the life history of individual by-catch species (Alverson *et al.* 1994). Stobutzki (*et al.* 2001a) proposed that the ability of a species to sustain by-catch impacts depends on two factors: (i) the 'susceptibility' (i.e., exposure) of a species to capture and mortality; and (ii) the capacity of a species for 'recovery' once the population is depleted. Susceptibility is a function of the spatial and temporal distribution of the by-catch species (Table 1.2). It is also a function of the distribution of fishing effort. Susceptibility can change with time, as fishing grounds expand or contract, fishing effort intensifies or reduces, and the density of a by-catch species increases or decreases. Recovery is essentially how quickly a population can compensate for individuals killed by fishing and is determined by the biology and life history strategy of the by-catch species involved (Table 1.2). Species that are large, slow-growing, with low reproductive rates, delayed maturity and naturally low adult mortality will have low recovery potential and therefore will be impacted more severely by fishing mortality than species that are small, fast-growing, early maturing, highly fecund and experience naturally high adult mortality rates (Alverson *et al*. 1994; Heppell *et al.* 1999, Musick 1999; Roberts and Hawkins 1999). Marine species that display low recovery potential are generally the 'long-lived' species, and include sea turtles, marine mammals, sharks, sea snakes, some sea birds and large bony fish (Musick 1999). The inherently low recovery potential of long-lived species implies that even limited levels of by-catch mortality can lead to stock collapse (Heppell *et al.* 1999). In addition, these species often incur other anthropogenic impacts (e.g., direct harvesting) and as such, any incidental mortality to these species will contribute to a conservation problem (Hall *et al.* 2000).

Table 1.2 Criteria that determine the sustainability of by-catch species

(Adapted from Stobutzki *et al*. 2001a)

Susceptibility criteria	**Recovery criteria**
Diet	Maximum age (i.e., longevity)
Diel activity	Reproductive strategy
Distribution in regards to the fishing grounds	Probability of breeding before capture
Capture survival	Fishing mortality rate

The by-catch of endangered or charismatic species such as sea turtles, marine mammals, and sea birds is a major factor in fisheries management (Harris and Ward 1999). Some fisheries are now closed when the by-catch limits are exceeded e.g., sea lion by-catch in the New Zealand hoki fishery (Hall *et al.* 2000). Therefore, assurance that marine fisheries are sustainable must be underpinned by: (i) understanding and quantifying the by-catch problem; (ii) evaluating appropriate by-catch management strategies; and (iii) monitoring to ensure adopted by-catch management strategies are effective in achieving their objectives. These aspects form a comprehensive approach to the sustainable management of by-catch in commercial fisheries. Such an approach for sea turtle by-catch in trawl fisheries of the Queensland east coast is explored in this thesis.

1.2.3 Sea Turtle by-catch

Globally

Sea turtle by-catch has received considerable global attention from researchers, managers and conservationists, and has been subject to international political intervention. Sea turtles are charismatic species (Harris and Ward 1999) that are listed as threatened by the World Conservation Union (IUCN 2000). Significant numbers are incidentally caught and killed in commercial fisheries (Magnuson *et al*. 1990). Sea turtles are susceptible to capture in many types of fishing operations (Table 1.3), but are caught primarily in coastal set nets for teleost fish and sharks (Paterson 1990; Dudley and Cliff 1993; Brady and Boreman 1994; Gribble *et al.* 1998), pelagic longlines for sharks, swordfish and billfish (Skillman and Balazs 1992), driftnets for cephalopods (Wetherall *et al.* 1994) and demersal trawl nets for penaeid prawns (Hillestad *et al*. 1981; Henwood and Stuntz 1987).

The scale of sea turtle by-catch and associated mortality varies between and within different fishing methods. The number of sea turtles incidentally caught is a function of the relative density of sea turtles within the fishing grounds. For example, high numbers of sea turtles will be caught in areas where effort and sea turtle density are both high, whereas low numbers will be caught in areas where effort and sea turtle density are low. Not all sea turtles incidentally caught in fishing operations are killed. Capture survival is a function of the characteristics of fishing operations. For example, mortality is negligible in fisheries where sea turtles are able to surface to breathe (e.g., shark drumlines, Gribble *et al.* 1998). In contrast, mortality is greater in fisheries where sea turtles are forcibly submerged (e.g., trawl nets) and depends on the duration of the forced submergence (Watson and Seidel 1980).

In trawl fisheries

In the early 1990's, prawn trawl fisheries were identified as having the greatest anthropogenic impact on many sea turtle populations (Magnuson *et al.* 1990). Despite acknowledged limitations, fishery-dependent studies have provided baseline data on when, where and how many sea turtles were caught and directly killed in trawl nets prior to the use of TEDs in many of these fisheries (Table 1.3).

Table 1.3 Annual catch and effort statistics of sea turtle by-catch in commercial fisheries

Fishery	Location	Target catch (t)	Estimated catch (± s.e.*) or (95% C.I.**)	Estimated kill (± s.e.*) or (95% C.I.**)	Comments
Trawl Prawn	Terengganu, Malaysia		[A] 742	[A]742	Based on interviews of fishers. Assumes all sea turtles killed. Nesting-grounds for *Dermochelys coriacea*.
	SE Atlantic, USA	13,000	[B] 33,881 (± 3,522*) [C] 26,075	[B] 7,115 (± 740*) [C] not estimated	Total fleet effort of 704,376 standard net hours. Based on 1.4% observer coverage and interviews with fishers. Mitigated through Turtle Excluder Devices (TEDs).
	Gulf of Mexico, USA	122,000	[B] 12,497 (±6,042*) [C] 3,135	[B] 3,755 (± 1,752*) [C] not estimated	Total fleet effort 4,315,698 standard net hours. Based on 0.38% observer coverage and interviews with fishers. Mitigated through TEDs.
	Mexico	87,106	[C] 48,779	[C] 11,324	Desktop study. Estimated from prawn landings, assuming catch:by-catch ratio for sea turtles being the same as in the USA. Mitigated through TEDs
	Central America	27,132	[C] 15,195	[C] 3,528	Desktop study. Estimated from prawn landings, assuming catch:by-catch ratio for sea turtles being the same as in the USA. Mitigated through TEDs
	South America	82,217	[C] 46,042	[C] 10,628	Desktop study. Estimated from prawn landings, assuming catch:by-catch ratio for sea turtles being the same as in the USA. Mitigated through TEDs
	Northern Prawn Fishery, Australia	6,267	[D] 5,730 (± 1,907*) [E] 5,357	[D] 344 (± 125*) [E] 777	Total fleet effort of 26,921 boat days (~323,052 standard net hours). Based on research surveys and selective logbook. Mitigated through TEDs, mandatory from 2000.
	Queensland east coast, Australia	7,000	[F] 5,295 (± 1,231*)	[F] 58 (± 14*)	Total fleet effort of ~85,000 days (~918,474 standard net hours). Based on selective logbook with 7.6% coverage. Mitigated through TEDs, mandatory from 1999 to 2002.
	Torres Strait, Australia [H]	2,000	[G] 652 (537 – 788**)	[G] (5 – 8**)	Total fleet effort ~8,634 days. Based on selective logbook information. Mitigated through TEDs, mandatory from 2001.
Longline	Western Atlantic Ocean		[H] 316 (± 334*)		Total fleet effort of 11,459,800 hooks fished with light sticks and 6,338,350 hooks fished without light sticks. Based on logbook information and observers.
Longline Swordfish	Western Mediterranean		[I] 18,000		Alverson *et al*. 1994 estimates a 45% mortality rate in longline fisheries.
Driftnet Squid	North Pacific Ocean		[J] 6,100	[J] 1,700	Observer program. Fishing by Japanese, Korean and Taiwanese fleets. Effort ~2.6 million tans. Drift netting subsequently banned. Fishery closed in 1996.

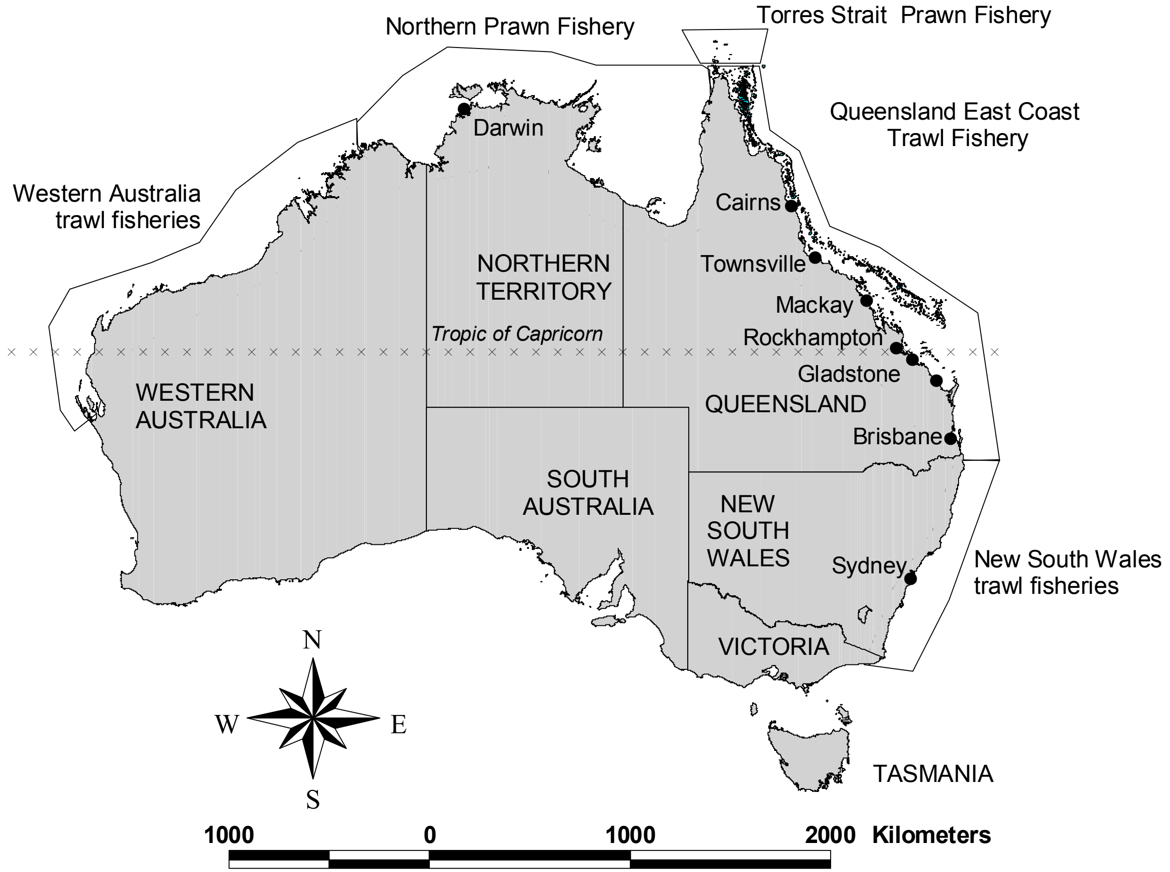
[A] Chan *et al.* (1988), [B] Henwood and Stuntz (1987); [C] Henwood *et al.* (1992); [D] Poiner *et al.* (1990); [E] Poiner and Harris (1996), [F] Robins (1995),[G] Robins and Mayer (1998); [H] Witzell (1999); [I] Goñi (1998); [J] Wetherall *et al*. (1994); *Standard errors (s.e.) or **95% confidence interval (C.I.) were not always available.

In Australia

Sea turtle by-catch occurred in the prawn trawl fisheries of northern Australia prior to the mandatory use of Turtle Excluder Devices (TEDs). The scale of sea turtle by-catch was estimated at ~5,500 sea turtles per year (of which ~800 were estimated to die) in the Tiger Prawn sector of the Northern Prawn Fishery (Poiner and Harris 1996). The annual by-catch mortality of sea turtles was estimated to be less than 2% of the speculated number of sea turtles occurring within the area of the fishery for *Chelonia mydas* (green turtles), *Natator depressus* (flatback turtles) and *Lepidochelys olivacea* (Pacific Ridley turtles), about 2.5% for *Caretta caretta* (loggerhead turtles) and about 3.0% for *Eretmochelys imbricata* (hawksbill turtles). The maximum direct and indirect mortality of sea turtle by-catch in the Tiger Prawn sector of the Northern Prawn Fishery was estimated to be about 2,100 sea turtles per year.

Prior to the mandatory use of TEDs, three other prawn trawl fisheries negatively impacted upon sea turtle populations of northeastern Australia: (i) the New South Wales Trawl Fishery; (ii) the Torres Strait Prawn Fishery; and (iii) the Queensland East Coast Trawl Fishery (Figure 1.1). Sea turtle by-catch data are limited for New South Wales, speculated to be “insignificant” (Dr Steve Kennelly, NSW Fisheries, personal communication 1994) and are not mitigated through the compulsory use of TEDs. The Torres Strait Prawn Fishery has a restricted level of fishing effort (i.e., maximum of 13,570 nights) and the annual sea turtle by-catch was estimated to be about 650 (95% C.I. 537 to 788), of which 4% were reported to die (Robins and Mayer 1998). *N. depressus* and *C. mydas* dominated the sea turtle by-catch in the Torres Strait, with fewer than 100 *C. caretta* being caught per year (Robins and Mayer 1998). The Queensland East Coast Trawl Fishery was estimated to catch about 5,300 sea turtles of which 1.1% died (Robins 1995), although these were preliminary estimates based on only two years of data. Of the trawl fisheries in northern Australia, the Queensland East Coast Trawl Fishery had the greatest potential to impact the endangered *C. caretta* sub-population of eastern Australia. This is because greater than 85,000 days of trawling wer recorded in the fishery per year and fishing effort overlaps in distribution with areas of high density of *C. caretta* in feeding- and nesting-grounds in southern Queensland (Limpus and Reed 1985b; Marsh and Saalfeld 1990; Limpus *et al.* 1994a; Limpus and Reimer 1994; Tucker *et al*. 1995).

Figure 1.1 Locality map of trawl fisheries in Australia.



More detailed estimates of the impact of the Queensland East Coast Trawl Fishery than that provided by Robins (1995) would assist in assessments of the causes of the decline in nesting numbers of *C. caretta*, such as those conducted by Heppell *et al.* (1996) and Chaloupka and Limpus (1998). Quantification of by-catch mortality by species and size-class is useful in such assessments to quantify the rate of population decline and proportional impact of various anthropogenic sources of mortality. In addition, a greater understanding of sea turtle by-catch would contribute to simulations of how the east Australian *C. caretta* sub-population might respond to the mandatory use of TEDs in prawn trawl fisheries of northern Australia. These aspects are explored in Chapter 3 and Chapter 4 of this thesis.

1.2.4 Managing by-catch

There are many ecological, political and economic reasons to manage by-catch (Andrew and Pepperell 1992). Two main strategies have been used to manage the by-catch of threatened or charismatic species, such as marine mammals and sea turtles. For some species, by-catch is managed on the basis of allowable mortality rates that determine a sustainable kill of a by-catch species. For example, fisheries in the northwestern USA are closed once a pre-set number of marine mammals are caught incidentally in fishing operations (Demaster *et al.* 1982; Young *et al.* 1993). This strategy can only be applied to species where current biological knowledge is sufficient to allow the prediction of sustainable levels of by-catch mortality. This is not possible for many species and is not appropriate for threatened species with depleted populations.

The other strategy for managing by-catch is to minimise its occurrence by: (i) modifying the spatial or temporal distribution of fishing effort; (ii) changing fishing practices; or (iii) improving the selectivity of fishing operations. Spatial and temporal closures are effective strategies where by-catch is predictable or aggregated (Hall 1996). For example, inshore gill-net fisheries are excluded from selected areas of the Queensland east coast (e.g., Shoalwater Bay) where there is a history of dugong deaths (Marsh 2000). Alternatively, simple alterations to fishing practices can sometimes be a cost-effective strategy to reduce by-catch or mortality of particular species. For example, high-seas purse seine fisheries have changed their fishing practices to minimise dolphin by-catch by not targeting schools of tuna associated with dolphins and using 'back-down' procedures to minimise dolphin mortalities (Hall 1996). Improving the selectivity of fishing operations is the most widely adopted strategy to deal with by-catch in prawn trawl fisheries. For example, Turtle Excluder Devices (TEDs) and By-catch Reduction Devices (BRDs) reduce the by-catch of sea turtles and non-target fish species respectively (Watson and Seidel 1980; Watson *et al.* 1999). TEDs have been adopted in numerous countries as the primary means of managing sea turtle by-catch in prawn trawl fisheries (Table 1.3; see also Robins 1997).

TEDs as a solution

TEDs were first developed in the early 1980's with the primary objective of releasing sea turtles from nets during trawling operations (Watson and Seidel 1980; Watson 2000). TED designs have evolved through research and industry use (Watson *et al*. 1986; Brewer *et al*. 1998; Robins *et al*. 1999) and are now consistently a rigid barrier between the main body of the net and the codend. TEDs direct animals larger than a certain size (determined by the bar spacing of the TED) towards an escape hole that may or may not be covered by a flap. The design and placement of a TED influences its efficiency at excluding sea turtles. In the USA, TED designs must pass a rigorous certification process to ensure that 97% of sea turtles encountering the TED will be excluded (Crowder *et al*. 1994; Mitchell 1996). Many countries lack the resources to undertake such stringent testing, but it is generally assumed that all TEDs will exclude 95% to 97% of sea turtles.

Monitoring the effectiveness of TEDs is difficult when they are applied in prawn trawl fisheries. This is because many prawn trawl fisheries occur over large geographic areas, have numerous participants and sea turtle by-catch can be a relatively rare event i.e., less than 0.05 sea turtles per hour of trawling (Henwood and Stuntz 1987; Poiner *et al.* 1990; Epperly *et al.* 1995a; Robins 1995). Indications of the short-term effectiveness of TEDs have been derived from significant reductions in the stranding rates of sea turtle carcasses adjacent to trawling grounds where TEDs are compulsory (Crowder *et al*. 1995). Modelling suggests that sea turtle populations should recover slowly after TEDs are regulated into an impacting fishery, assuming that trawl by-catch is the main anthropogenic mortality factor acting on the sub-population (Crowder *et al.* 1994). In theory, TEDs can reduce sea turtle by-catch in prawn trawl fisheries to less than 5% of former levels, but several practical issues can interfere with TEDs being an effective solution to sea turtle by-catch.

Limitations of TEDs

TEDs are only as effective at excluding sea turtles as: (i) their defining regulations; and (ii) their exclusion rate during actual trawling operations. In the USA, regulations defining TEDs have continually changed since 1990 in order to limit the lateral interpretation of TED definitions (Mr John Watson, NMFS, personal communication 1997). TEDs can be modified or disabled temporarily at sea to reduce their effectiveness at excluding sea turtles. This may be undertaken because fishers attribute perceived reductions in catch rates of commercial species to the presence of TEDs.

The greatest challenge associated with the regulation of TEDs into a fishery is ensuring compliance. This is particularly a challenge when there are numerous participants in a fishery (e.g., USA and India), when the fishery occurs in remote areas (e.g., Gulf of Carpentaria, Australia), or is distributed over a large geographic scale. Enforcement resources are often limited or insufficient to ensure fishery-wide compliance with TED regulations. Therefore, one of the most important aspects of reducing sea turtle by-catch through the use of TEDs is the formulation of a strategy to monitor the effectiveness of TEDs and ensure high compliance. Ensuring the effectiveness of TEDs is particularly important in areas where sea turtle by-catch is high or where endangered species are caught most commonly. However, the issue of enforcement receives little attention in the literature, despite the universality of the problem. The alternative to enforcement is observers, which is impractical in many trawl fisheries as a result of excessive cost or scale. Methods for identifying high priority areas for the enforcement and monitoring of TEDs in a prawn trawl fishery are explored in Chapter 5 and Chapter 6 of this thesis.

TEDs in Australia

TEDs were regulated into the prawn trawl fisheries of northern Australia between 1999 and 2002 (Robins and Dredge 2000). A 95% reduction in sea turtle by-catch in these fisheries is a management or conservation target of: (i) the By-catch Action Plan for the Northern Prawn Fishery (NORMAC 1998); (ii) the Queensland Fishery Management Plan: East Coast Trawl (QFMA 1998); and (iii) the Draft National Marine Turtle Recovery Plan (EA 1998). However, there is limited observer coverage of the commercial use of TEDs and in general, is insufficient to validate the 95% reduction target in sea turtle by-catch. In addition, initial TED definitions for the Queensland East Coast Trawl Fishery were very broad and were considered ineffective in defining appropriate TEDs that would ensure a 95% reduction in sea turtle by-catch (Mr Peter Tanner, QFBP, personal communication 1999). This situation has been remedied in 2002 with more stringent regulations. Enforcement of TEDs is necessary to ensure that the equipment being used is effective in allowing sea turtles to escape. However, the extensive spatial and temporal scale of the Queensland East Coast Trawl Fishery makes enforcement difficult. Many countries have similar enforcement problems that need to be addressed if the problem of sea turtle by-catch is to be effectively resolved.

1.3 THE NEED FOR A MORE COMPREHENSIVE APPROACH

In general, sea turtle by-catch is managed as an all-or-nothing approach, despite the spatial and temporal heterogeneity in the probability of a sea turtle being caught and killed. There are strong conservation and management reasons supporting the use of this approach, but the widespread regulation of TEDs has resulted in social unrest amongst fishers (Margavio *et al*. 1993; Moberg and Dyer 1994) or has translated to the slow implementation of TEDs for economic or political reasons (Watson 2000). A more comprehensive approach is required that utilises the available information to build our understanding of the interaction between fishing and endangered sea turtle species.

Little consideration appears to be given to the appropriate scale at which TEDs or other by-catch management strategies are implemented in fisheries and the potential for enforcement of these strategies to ensure a genuine outcome (Tucker *et al.* 1997). By-catch management strategies are currently applied at large spatial scales because of the inability of enforcement at small spatial scales or in remote areas. However, advances in technology, such as transponders and Vessel Monitoring Systems are beginning to change the potential for fine scale fisheries management (Caddy and Cochran 2001) and could be applied to managing by-catch at more efficient scales.

Identifying the areas most critical for effective TED usage i.e., where sea turtle density or trawl-related mortality is highest, would assist in focusing enforcement efforts in areas with the greatest conservation benefit. Such information would allow a comprehensive approach to sea turtle by-catch through the use of compulsory TEDs supported by monitoring of TED performance and enforcement for high compliance in areas of conservation priority.

1.3.1 Data requirements

Several pieces of information are required to develop a comprehensive approach to sustainably managing sea turtle by-catch. These include: (i) the relative distribution of sea turtles; (ii) the relative distribution of the fishery; and (iii) the scale and nature of sea turtle by-catch i.e., where does most sea turtle by-catch occur and where does the greatest mortality occur, particularly of endangered species. These types of data were collected for this thesis.

The lack of knowledge of the scale and distribution of anthropogenic impacts is one of the underlying problems of sea turtle conservation. This is partly the consequence of the focus of sea turtle research on nesting beach activity and the processes involved with eggs and hatchlings, with less research focused on sea turtles in feeding-grounds (Bjorndal and Bolten 2000). Most sea turtle research has also been limited to relatively small spatial scales, predominantly as a consequence of logistic and economic difficulties. However, there is growing recognition that sea turtle populations need management at much larger spatial scales than those for which research data are currently available (Musick 1999; Bjorndal and Bolten 2000). Information is required on the relative distribution and trends in abundance of sea turtles throughout their feeding-grounds (Dobbs 2001; TEWG 2000) and this information is required at scales that are suitable for management (Dobbs 2001).

Baseline data on sea turtle by-catch are important for several reasons: (i) to assess the scale of the fishing impacts on sea turtle sub-populations (Lutcavage *et al.* 1996); (ii) to improve our understanding of factors that influence sea turtle by-catch; (iii) to provide insight into the relative distribution of sea turtles; and (iv) to identify the most critical areas for sea turtle by-catch mortality. All these factors are fundamental to the comprehensive management of sea turtle by-catch through the monitoring and enforcement of TEDs in critical areas and to convince practitioners in culprit fisheries of the need mitigate their impact and the value of their efforts.

1.3.2 The Queensland East Coast as a Case Study

The Queensland east coast extends from Cape York (10.7°S, 142.5°E) to the state border with New South Wales (~28°S). This thesis examines the sea turtle by-catch in the Queensland East Coast Trawl Fishery as a case study for developing a comprehensive approach to sea turtle by-catch for the following reasons.

Significance to World Sea Turtle Conservation

The Queensland east coast supports some of the largest remaining sea turtle sub-populations in the world (Dr Colin Limpus, QPWS, personal communication 2000), by providing nesting-grounds for *C. mydas*, *C. caretta*, *E. imbricata* and *N. depressus* and feeding-ground for six of the world's seven species of sea turtles i.e., the above species plus *L. olivacea* and *Dermochelys coriacea* (leatherback turtles). All sea turtle species are listed by the World Conservation Union as threatened with very high or high risk of extinction in the wild in the immediate or medium-term future (IUCN 2000).

World Heritage Obligations

The Great Barrier Reef World Heritage Area (GBRWHA) occupies a significant proportion of the waters off the Queensland east coast. The Queensland and Australian Governments have an obligation under World Heritage listing to identify, protect, conserve, present and transmit cultural and natural heritage aspects of the GBRWHA of outstanding universal value to future generations (Valentine *et al*. 1997). This explicitly includes the significance of the Area to the continued survival of sea turtles (GBRWHA 1981). World Heritage properties are reviewed to determine if their values have remained intact, obliging the responsible agencies to identify and implement world's best practice in management and ensure that any resource use is ecologically sustainable.

Evidence of a Decline

The Queensland east coast has been the location of significant long-term research by the Queensland Turtle Research Group (QTRG). This research has provided major insights into the biology and ecology of sea turtles. The QTRG annually monitor trends in nesting numbers of *C. caretta* and *C. mydas.* A decline in the number of nesting females of 50 to 80% was observed for the east Australian sub-population of *C. caretta* over the past 25 years and whilst a number of anthropogenic activities impact on *C. caretta*, predation of nests by introduced foxes and demersal prawn trawling were likely to be the high impact contributors to the decline (Limpus and Reimer 1994). The decline in nesting numbers of *C. caretta* was a major reason for the introduction of TEDs in the Queensland East Coast Trawl Fishery (Robins and Dredge 2000).

Multiple fisheries

The Queensland East Coast Trawl Fishery is managed as a single fishery, but is actually comprised of multiple sectors that target ten species of penaeid prawn, two species of scallop, one species of fish and 10 species of non-target catch (Dredge and Trainor 1994; Robins 1995; Robins and Courtney 1999). Trawling occurs from shallow coastal waters (<10m deep) to the edge of the continental shelf (~200m deep) and includes three fishing techniques i.e., beam trawl, otter trawl and semi-pelagic fish trawl. Catch and effort are relatively well documented through a compulsory daily logbook that was introduced in 1988 by the Queensland Government through its fisheries management agency i.e., the Queensland Fisheries Service formerly the Queensland Fish(eries) Management Authority.

Enforcement challenge

TEDs were regulated into the Queensland East Coast Trawl Fishery in a stepwise process from 1999 to 2002 (Robins and Dredge 2000). Whilst TED legislation is in place, the challenge for fisheries management is to ensure that effective TEDs are used throughout the fishery in order to achieve a 95% reduction in sea turtle by-catch (QFMA 1998; EA 1998). As in other fisheries, not all fishers in the Queensland East Coast Trawl Fishery value the benefits of the compulsory use of TEDs and as such monitoring and enforcement will be necessary. However, the fishery is distributed across some 226,900 km^2 and has over 800 participants. Resources for enforcement are limited, therefore enforcement of TEDs should be targeted in areas with the greatest conservation benefit i.e., those areas with high catch or mortality rates of sea turtles. Identifying priority areas for TED enforcement and monitoring is investigated in Chapter 6 of this thesis.

History of alternative management measures

The Queensland East Coast Trawl Fishery has a strong history of management including limited entry, restricted fishing areas, and gear specifications. More recent management measures include restricted fishing time per vessel (i.e., days fished) and the use of Vessel Monitoring Systems (i.e., satellite tracking of individual vessels). The history and degree of regulation in this fishery permit alternative management measures to be considered, as they are both politically and practically possible. Enforcement of fishing regulations is a priority of the Queensland Government and is the primary jurisdiction of the Queensland Boating and Fisheries Patrol (QBFP), a government agency.

1.4 AIMS AND OBJECTIVES

This thesis aims to resolve deficiencies in the knowledge required to sustainably manage sea turtle by-catch through a comprehensive approach, which has the following objectives:

(v) Estimate the number and species composition of sea turtles caught and killed in a multiple sector prawn trawl fishery using spatial and temporal stratification;

(vi) Investigate the response of sea turtles to trawl capture to examine evidence of post-trawl mortality or altered behaviour that would lead to secondary mortality;

(vii) Examine factors that influence the distribution of sea turtles, from which relative densities of sea turtles can be estimated at broad spatial scales; and

(viii) Assess the spatial interaction between sea turtles and fishing effort to identify priority areas for TED enforcement or other sea turtle conservation management.

1.5 THESIS OUTLINE

Chapter		Details
1	General Introduction	Describes the rationale of the thesis, describes the structure of the thesis, and gives a brief outline of the methodology.
2	Literature Review	Reviews sea turtle biology and ecology that is relevant to issues associated with sea turtle by-catch.
3	Estimated Catch and Mortality	Estimates the catch and mortality of sea turtles in the Queensland East Coast Trawl Fishery and considers these estimates in the context of trends in sea turtle population size.
4	Responses of Sea Turtles to Capture	Compares the response of trawl-caught and rodeo-caught sea turtles and discusses the implications for post-trawl mortality.
5	Spatial Distribution of Sea Turtles	Estimates the relative density of sea turtles along the Queensland east coast using trawl capture and aerial survey data, predicts relative spatial distribution of sea turtles and identifies areas of high relative sea turtle density.
6	Assessment of Critical Areas and Management Implications	Identifies critical areas for sea turtle by-catch and considers the implications for fisheries management and sea turtle conservation.
7	Conclusions	Reiterates the research findings of the thesis and discusses state, national and international implications for sustainable fisheries management in general.
8	References	Lists the sources of information cited in the thesis.
	Appendices	Provides details of additional information referred to in various chapters

1.6 METHOLOGICAL APPROACH

1.6.1 Issues

The problem of scale

The jurisdiction of the Queensland East Coast Trawl Fishery encompasses about 226,900 km^2 of aquatic habitats between high-water mark and edge of the Australian continental shelf. A significant portion of this area cannot be trawled for legislative reasons (i.e., spatial closures) or practical purposes (i.e., coral or rocky reefs). The spatial extent of the fishery provides a diversity of habitats and trawling conditions, but sampling for any parameter relevant to sea turtle by-catch at this scale is difficult and requires a multi-facetted approach.

Observational versus experimental

Sea turtles are often a relatively infrequent by-catch of prawn trawling operations, with catch per unit effort averaging less than 0.05 sea turtles per hour of trawling (Henwood and Stuntz 1987, Poiner *et al.* 1990, Epperly *et al.* 1995a; Robins 1995). Low frequency of capture and ethical considerations limit the research of sea turtle by-catch to observational studies. High costs of vessel charter limit the use of dedicated research trawls to document the spatial and temporal distribution of sea turtle by-catch. Understanding sea turtle by-catch is approached most pragmatically through fishery-dependent sampling using an observer or a logbook program. Scientific observers are the most independent means of monitoring by-catch on commercial vessels but require substantial financial resources and good spatial and temporal coverage to obtain representative samples (Stobutzki *et al.* 2001b). Most Australian fisheries use compulsory logbooks to monitor the effort expended to catch commercial species (Kailola *et al*. 1993). I applied logbook methodology in a voluntary and selective form to obtain a wide spatial and temporal representation of sea turtle by-catch in the Queensland East Coast Trawl Fishery. Where available, I have compared sea turtle by-catch reported during commercial prawn trawling with that observed during research trawling (see Chapter 3).

1.6.2 Overview

Sampling sea turtle by-catch

Sea turtle by-catch was monitored by selected commercial fishers and reported upon through a research logbook, referred to as the 'sea turtle by-catch monitoring program'. The program ran from January 1991 to December 1996 and was targeted primarily at the Queensland East Coast Trawl Fishery and Torres Strait Prawn Fishery, but had several participants from the Northern Prawn Fishery. Sea turtle by-catch was matched to the corresponding effort for participating vessels that was retrieved from the compulsory catch and effort logbook managed by the Queensland Fisheries Service. The program and data are discussed in Chapter 3, sections 3.3.1 and 3.3.2.

Estimating annual sea turtle by-catch

Sea turtle by-catch (and 95% confidence interval) was estimated using C=RxT, where C is the estimated total sea turtle by-catch, R is the catch rate (i.e., CPUE) and T is the total fishing effort (Robins 1995, Poiner and Harris 1996). Catch rate (i.e., sea turtle catch per unit effort) was weighted by sampling effort and was stratified for fishing sector, year and month. Similarly, total fishing effort was stratified for fishing sector, year and season (i.e., high or low). Estimation procedures are discussed in greater detail in Chapter 3, section 3.3.3.

Responses of sea turtles to capture

Sea turtles caught in trawl nets and by rodeo capture were monitored after their release to determine their behaviour and their short-term mortality (i.e., one to three days post-release). The sea turtles were monitored using ultrasonic tracking equipment in combination with Temperature Depth Recorders. Dive profiles were examined to assess the extent of behavioural modifications (see Chapter 4).

Relative sea turtle density

Indices of relative sea turtle density were calculated from trawl catch rates and sightings from aerial surveys. Factors influencing the catch rates of sea turtles in trawled areas were assessed and the significant relationship between sea turtle catch, water depth and target species trawled was used to predict sea turtle CPUE for most areas of the Queensland continental shelf (see Chapter 5).

Identifying critical areas for sea turtle by-catch

The spatial distribution of effort for the Queensland East Coast Trawl Fishery in the year-2001 was integrated with the relative density of sea turtles to identify critical areas for sea turtle by-catch (see Chapter 6).

CHAPTER 2. BIOLOGY AND ECOLOGY OF SEA TURTLES

2.1 CHAPTER SUMMARY

Sea turtle by-catch is a significant issue that must be addressed by governments to achieve the ecologically sustainable development of fisheries resources and to meet international and national obligations to conserve sea turtle populations. The incidental capture of sea turtles in fishing operations, particularly demersal prawn trawl fisheries and oceanic long-line fisheries, causes significant mortalities and has contributed to major declines in sub-populations of some sea turtle species. The mandatory introduction of Turtle Excluder Devices (TEDs) has been assumed to address the problem of sea turtle by-catch in prawn trawl fisheries. However, TED use requires monitoring and enforcement to ensure the effective mitigation of sea turtle by-catch.

2.2 INTRODUCTION

Sea turtle by-catch is a significant global issue that has resulted in the widespread regulation of Turtle Excluder Devices (TEDs) into the prawn (=shrimp) trawl fisheries of many countries. The aim of regulating TEDs in these fisheries has been to minimise the number of sea turtles killed as a consequence of a forced submergence in trawl nets. However, regulation alone will not ensure the elimination of this impact and the subsequent recovery of sea turtle populations. A strong understanding of sea turtle biology and ecology is a prerequisite to interpreting the impact of by-catch mortality on populations of sea turtles and can assist by-catch management and conservation efforts to be focused in areas with the greatest conservation benefit.

2.2.1 Conservation status

There are seven extant species of sea turtle of which the world-wide populations have declined over the 20th Century, often as a consequence of the overexploitation of eggs and adults (Magnuson *et al.* 1990). At a global scale, all species of sea turtle are listed as Vulnerable, Endangered or Critically Endangered (Table 2.1) by the World Conservation Union (IUCN 2000). Declines in the numbers of nesting sea turtles are the basis for the World Conservation Union (WCU) listings, reflecting observed (Criteria A1) or suspected (Criteria A2) reductions in sea turtle populations within three generations. The size of the decline determines the listing (i.e., Critically Endangered represents an 80% decline, Endangered represents a 50% decline and Vulnerable represent a 20% decline).

Table 2.1 Conservation status of sea turtles

Species	WCU		Australia[1]	Qld[2]	NT[3]	WA[4]	NSW[5]
	Listing	Criteria*					
Chelonia mydas	EN	A1	VU	VU	VU	VU	VU
Caretta caretta	EN	A1	EN	EN	EN	EN	VU
Natator depressus	DD	A2	VU	VU	VU	VU	nl
Eretmochelys imbricata	CR	A1 & A2	VU	VU	VU	VU	nl
Lepidochelys olivacea	EN	A1	EN	EN	EN	EN	nl
Dermochelys coriacea	CR	A1	VU	EN	VU	VU	VU

VU = vulnerable, E = endangered, CR = critically endangered, DD = data deficient, nl = not listed. *WCU criteria – A1: observed reduction in population size within three generations, A2: suspected decline in population size within three generations. [1] Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*; [2] Queensland *Nature Conservation (Wildlife) Regulations 1994;* [3] no specific Northern Territory listing so Commonwealth listings adopted; [4] *Western Australian Wildlife Conservation Act 1950;* [5] New South Wales *Threatened Species Conservation Act 1995*.

The WCU listings imply that at a global scale, all species of sea turtle are at very high or high risk of extinction in the immediate and medium-term future. The critical status of most sea turtle populations is also recognised by the prohibition of their commercial trade under the Convention on International Trade in Endangered Species (CITES) of Wild Fauna and Flora (Edgar and Stephens 1993) as well as the Convention on Migratory Species. The WCU listings and the CITES convention aim to promote the conservation of sea turtles. The WCU (IUCN 2000) defines conservation as:

> "*The management of human use of the biosphere so that it may yield the greatest sustainable benefit to present generations, while maintaining its potential to meet the needs and aspirations of future generations*".

The Australian continental shelf an area where sea turtles have been subject to relatively minor exploitation from humans (i.e., direct harvesting) in comparison to places such as Indonesia, Mexico and Costa Rica (Marquez 1990; Chan and Liew 1996; Bjorndal *et al.* 1999; Suarez 2000; Suganuma *et al.* 2000). In Australia, protective legislation was introduced in the middle of the 20th Centaury (e.g., Queensland *Fisheries Act 1962*), with all six species of sea turtle that occur in Australian waters variously being protected by Commonwealth and State legislation (Table 2.1). This has prevented the human exploitation of Australian sea turtle populations in the past four decades, excepting indigenous harvest for non-commercial purposes[2]. However, prior to protection, it is likely that sea turtles were exploited by non-indigenous Australian's for eggs (all species) and meat (predominately *C. mydas*), which may have substantially reduced the population size of various species of sea turtles in northern Australia (Limpus *et al.* 2002).

Australian sea turtle species are listed as Vulnerable or Endangered under the relevant State and Commonwealth legislation (Table 2.1) and are protected as a migratory species by the Commonwealth *Environment Protection Biodiversity Conservation Act 1999*. Despite the protective legislation and the inclusion of large areas of sea turtle habitat in Marine Parks and World Heritage Areas (e.g., Great Barrier Reef), declines have occurred in some Australian sub-populations (= nesting assemblages, IUCN 2000). Of most concern, is the 80% decline in nesting numbers of the east Australian *Caretta*

caretta sub-population (Limpus and Reimer 1994). Declines in Australian sea turtle sub-populations are a consequence of a range of human activities that impact upon sea turtles throughout their life (Magnuson *et al.* 1990). These impacts are difficult to disaggregate to determine their relative importance. This increases the priority for fisheries impacts on sea turtle populations to be sustainable in light of other anthropogenic factors.

2.3 LIFE HISTORY

The generalised life cycle of sea turtles can be categorised into five ontogenetic stages (Musick and Limpus 1996): (i) eggs; (ii) hatchlings; (iii) juveniles; (iv) sub-adults; and (v) adults (Figure 2.1). Sub-adult and adult sea turtles reside in feeding-grounds for most of the year. Once every two to eight years, mature adult sea turtles migrate between feeding- and nesting-grounds that are from 10 to 2,600 km apart (Limpus and Nicholls 1988; Limpus *et al*. 1992). Courtship occurs during migration or in the immediate vicinity of the nesting-ground. Mated female sea turtles make nesting crawls up the beach to deposit eggs, nesting between two and eight times during one breeding season, depending on the species. Female sea turtles remain relatively close to the nesting beaches and do not feed during nesting (Limpus 1973; Forbes 1994). When nesting is complete, female sea turtles migrate back to their feeding-grounds and tag-recapture has shown strong fidelity of individual sea turtles to particular feeding-grounds in Queensland (Limpus and Limpus 2001).

Sea turtle eggs incubate for four to eight weeks, depending on temperature (Georges *et al.* 1993). Incubation temperature determines the sexual bias of the clutch, with warmer temperatures producing a greater proportion of females and cooler temperatures producing a greater proportion of males (Standora and Spotila 1985; Georges *et al*. 1993; Mrosovsky 1994). Therefore, the sand temperature in which most of the eggs are laid determines the primary sex ratio of the sub-population. Incubation temperature has consequences for the sub-population as a whole because different nesting beaches contribute a different proportion of males and females (Heppell *et al.* 1996). Impacts at female-biased beaches will have greater effects on sub-population trends than impacts at

[2] Indigenous hunting of sea turtles is allowed under Australian Native Title legislation, the Torres Strait Treaty between Australia and Papua New Guinea, and State and Commonwealth legislation.

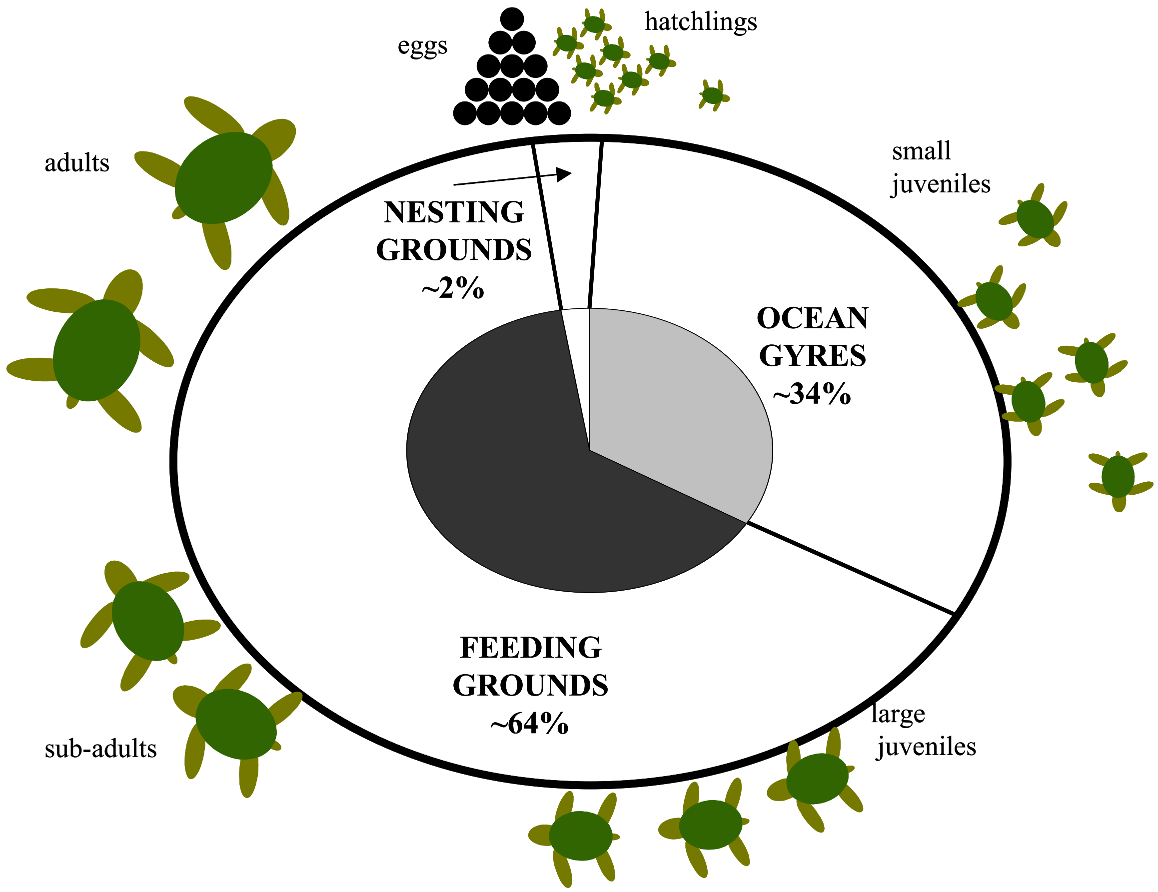
male-biased beaches, particularly where sub-population trends are measured by annual nesting indices (Richardson *et al.* 1978; Meylan 1981; Bjorndal *et al.* 1993).

Hatchlings swim away from the nesting beach and take up a pelagic existence, with all species except *Natator depressus* dispersing into the open ocean (Limpus *et al.* 1983a; Walker 1994; Musick and Limpus 1996). It is strongly suspected that early juvenile sea turtles live and feed in debris drift-lines of the major oceans (Witham 1980; Musick and Limpus 1996).

After two to 15 years in oceanic gyres (Limpus *et al.* 1994a; Chaloupka 1998; Bjorndal *et al.* 2000; TEWG 2000), juvenile sea turtles recruit to neritic habitats that are used as feeding-grounds until maturity, except for *Dermochelys coriacea*, which continues with a predominantly pelagic existence as an adult. It is suspected that some species remain at the initial feeding-ground to which they recruit e.g., *Chelonia mydas*, whilst others undertake a developmental migration between feeding-grounds e.g., *Eretmochelys imbricata* (Limpus 1992). Sea turtles return to breed at their natal nesting grounds (Miller 1996).

The generalised sea turtle life cycle can be split into three spatial categories that reflect the habitats used by various ontogenetic stages: (i) nesting-grounds (i.e., beaches and adjacent inter-nesting habitat); (ii) ocean gyres (for all species except *N. depressus*); and (iii) feeding-grounds (Figure 2.1). A theoretical example of the time spent in each category illustrates the spatial and temporal exposure of sea turtles to human impacts (Table 2.2). In general, sea turtles spend about two thirds of their life in feeding-grounds (Figure 2.1), suggesting that human impacts at feeding-grounds play a critical role in the fate of sea turtle populations. This generalisation is supported by elasticity analyses used to model sea turtle populations (Crowder *et al*. 1994; Heppell *et al.* 1996; Heppell *et al*. 1999). The survival rates of sub-adult and adult stages have a greater relative contribution to population growth than survival rates of eggs and hatchlings. Therefore, it is important to quantify the fishing by-catch mortality on these stages. This is addressed in Chapter 3 of this thesis.

Figure 2.1 Generalised life cycle of sea turtles



In reality, the proportion of time spent in each spatial category varies with species, geographic location and between individuals, reflecting genetic and environmental influences (e.g., food quality). If a species has a shorter oceanic phase i.e., eight to ten years (Chaloupka 1998; Bjorndal *et al*. 2000), a longer sub-adult phase i.e., 25 years (Heppell *et al*. 1996) or survives as a mature adult for greater than 20 years, then a greater proportion of an individual's life would be spent in feeding-grounds. In contrast, if a species has a longer oceanic phase or a shorter sub-adult phase then a smaller proportion of an individual's life would be spent in feeding-grounds. *N. depressus* are relatively more exposed to human impacts on feeding-grounds than other sea turtle species because this species does not have a oceanic phase (Walker 1994), being vulnerable to human impacts in continental feeding-grounds throughout the juvenile stages as well as sub-adult and adult stages.

Table 2.2 Spatial classification of a generalised sea turtle life cycle

Spatial habitat and ontogenetic stage	Duration[1] years x weeks = weeks	Portion of life cycle	Human impacts	Relative ability to manage human impacts
Nesting-grounds				
(v) Adults			Direct harvest	Known areas
• Courtship	4 yrs x 4 wks = 16	0.69%	Feral predation	Known seasons
• Nesting	4 yrs x 8 wks = 32	1.38%	Habitat loss	Land based
(i) Eggs	1 yr x 7 wks = 7	0.30%	Fishing by-catch	Difficulty with
(ii) Hatchlings	1 yr x 4 wks = 4	0.17%		remote locations
Sub-total	**59 weeks**	**2.5%**		**High**
Oceanic phase				Vast areas
(iii) Juveniles	15 yrs x 52 wks	2.24%	Fishing by-catch	International waters
		per year	Pollution	Poor knowledge base
Sub-total	**780 weeks**	**33.6%**		**Low**
Feeding-grounds			Direct harvest	Large areas
(iv) Sub-adults	10 yrs x 52 wks = 520	22.38%	Fishing by-catch	Often within an EEZ
(v) Adults			Pollution	Regulate human
• non nesting yrs	15 yrs x 52 wks = 780	33.58%	Habitat loss	activities
nesting yrs	4 yrs x 40 wks = 184	7.92%	Boat strike	
Sub-total	**1,481 weeks**	**63.9%**		**Medium**

[1]Total hypothetical life cycle is 2,323 weeks = 44½ years; numbers in Table 2.2 were derived from stage-based population models for female *C. caretta* (Heppell *et al.* 1996). The numbers for males would be similar but with slightly less time in the nesting-grounds and slightly more time in the feeding-grounds. Duration will also differ between species.

Human activities that impact upon sea turtles include the direct harvest of adults and eggs for human consumption, incidental capture or entanglement in fishing operations, entrainment in dredging operations, boat strikes, and the ingestion of plastic debris. Indirect effects include the loss or degradation of habitat (e.g., nesting beaches) through beach development and artificial lighting, depredation of nests by feral animals and exposure to oil pollution and other contaminants (Magnuson *et al.* 1990; Lutcavage *et al.* 1996). The relative impact of each human activity changes between sea turtle species and amongst different geographic locations. For example, the predicability, accessibility and density of sea turtles at nesting-grounds have contributed significantly to declines in sub-populations of *D. coriacea*, *C. mydas, Lepidochelys olivacea* and *L. kempii* (Limpus 1995; Lutcavage *et al.* 1996).

Managing all human impacts in all spatial habitat categories would be the preferred management intervention to assist in the recovery of depleted sea turtle populations. However, the effectiveness and practicalities of addressing impacts varies with the habitat category. Human impacts at nesting beaches have a relatively high degree of manageability through controlled intervention such as restricting human access and direct harvest or controlling introduced (i.e., feral) animals. This is a consequence of the predictable nesting patterns of sea turtles and the spatial concentration of mostly land-based impacts. This does not translate to the ‘easy’ management of human impacts at nesting beaches, as the solutions involved are often politically sensitive and have social ramifications. At the other extreme, managing human impacts in the open ocean has relatively low manageability, as a consequence of the remote and vast spatial scale over which the activities occur, often in international waters (Polovina *et al.* 2000). Between these two extremes is the ability to manage human impacts in feeding-grounds that often encompass large spatial scales, but usually occur within continental shelf waters of one or more nations. Managing human impacts on sea turtles in feeding-grounds should be a high priority because of the reasonable feasibility of management intervention and the significance of these impacts on population trends. However, effective management requires an understanding of the distribution of sea turtles in feeding-grounds and the impacts upon them. Developing a better understanding of the relative distribution of sea turtles in continental shelf waters is addressed in Chapter 5 of this thesis.

2.4 DISTRIBUTION

Sea turtles are predominantly distributed throughout tropical and sub-tropical waters of the world, but are not uniformly distributed throughout feeding-ground habitats. Whilst the general distribution of sea turtles is known, their relative density is poorly quantified, particularly at large spatial scales, such as the waters of the Queensland east coast. Insight into the probable distribution of sea turtles can be gained from knowledge of their preferred diets. The following account summarises the feeding strategies of the six species of sea turtle found in Australian waters and then reviews the known distribution of sea turtles in northern Australia. I have used this information, to speculate on the potential distribution of sea turtles in waters of the Queensland east coast.

Each species of sea turtle has specialised feeding habitats that influence its distribution (Hendrickson 1980; Bjorndal 1996). Sea turtles can be classified as primarily herbivorous (*C. mydas)*, omnivorous (*E. imbricata*), carnivorous (*C. caretta*, *L. olivacea*, and *N. depressus*) or gelantivorous (*D. coriacea*).

2.4.1 Inferring distribution from generalised feeding strategies

C. mydas is omnivorous during its pelagic oceanic stage, and then adopts a primarily herbivorous feeding strategy after settling onto benthic feeding-grounds (Bjorndal 1985; 1996). Its principle dietary items are seagrass, algae and mangrove fruits, although jellyfish and small crustaceans are occasionally included (Forbes 1994; Limpus *et al.* 1994a; Bjorndal 1996; Read and Limpus 2002). *C. mydas* is widely distributed in tropical and sub-tropical waters (Marquez 1990) being particularly abundant on coral and rocky reefs, seagrass meadows and algal turfs on sandy substrates.

E. imbricata is an omnivorous species, consuming reef-associated benthic organisms including sponges, tunicates and anemones (Bjorndal 1996). In general, the feeding-grounds of *E. imbricata* comprise coral reefs and other complex, hard substrate habitats (Hendrickson 1980; Meylan 1989). *E. imbricata* is considered to be predominantly tropical in its distribution (Marquez 1990).

The dietary preferences of *C. caretta* is better known than for the other carnivorous species i.e., *L. olivacea* and *N. depressus* (Dodd 1988; Plotkin *et al*. 1993; Limpus *et al.* 1994a; Bjorndal 1996; Limpus *et al.* 2001). Dietary items of carnivorous sea turtle species vary with location and individual, but in general include molluscs, crustaceans and other benthic fauna associated with soft-bottom habitats (Bjorndal 1996; Limpus *et* *al*. 2001). Prey items such as bivalve molluscs and benthic crustaceans occur commonly on penaeid trawl grounds making the carnivorous species susceptible to trawl capture (Shoop and Ruckdeschel 1982; Marquez 1990; Poiner *et al.* 1990; Robins 1995).

The carnivorous sea turtles appear to have similar dietary preferences but can be separated spatially on the basis of latitude preferences. *L. olivacea* and *C. caretta* are circumglobal in their distribution, but *L. olivacea* prefers tropical waters and *C. caretta* prefers sub-tropical waters (Hendrickson 1980; Marquez 1990; Musick and Limpus 1996). *N. depressus* is restricted in its distribution to the Australian continental shelf (Marquez 1990), and is more common in tropical waters (Limpus *et al.* 1983a). *N. depressus* and *L. olivacea* both prefer turbid benthic habitats in tropical waters (Harris 1994; Parmenter 1994), but appear to be separated by the preference of *N. depressus* for shallow waters (i.e., <20m deep, Limpus *et al.* 1983a) and of *L. olivacea* for deeper waters (i.e., 30 to 40m deep, Harris 1994). *N. depressus* in particular avoids reef habitats (Musick and Limpus 1996).

D. coriacea also occurs in Australian waters and is an obligate feeder of jellyfish and other gelatinous plankton (Bjorndal 1996). This species generally has a pelagic lifestyle in the open oceans but is known to forage close to shore and over the continental shelf (Limpus *et al.* 1984b).

The feeding-ground preferences of sea turtles can be synthesised to develop a framework of the probable distribution of each species (Table 2.3). It is not suggested that these preferences preclude the occurrence of a species in another type of feeding-ground, but the implication is that the greatest relative density of a species will be on the preferred feeding-ground habitats. This framework is useful in identifying the likely relative spatial distribution of sea turtles, particularly when considering their possible exposure to human impacts, which in the case of this thesis is demersal prawn trawling. It also assists in focusing research and management efforts such as TED enforcement, in areas that may have the greatest contribution to conserving sea turtle populations. The framework differentiates between tropical and sub-tropical areas, although it is unknown whether the changes in the relative density for those species that prefer one area is gradual between the tropics and sub-tropics or whether the change in relative density is distinct. This is a consequence of a lack of knowledge of the relative density of sea turtles in feeding-grounds across broad spatial scales.

Table 2.3 Hypothetical preferred feeding-grounds of sea turtles

Feeding-ground habitat type	Tropical	Reported occurrence	Sub-Tropical	Reported occurrence
Seagrass beds	*C. mydas* *N. depressus*	Y U	*C. mydas* *C. caretta*	Y Y
Rocky reefs	*C. mydas* *L. olivacea*	Y U	*C. mydas* *C. caretta*	Y Y
Coral reefs	*C. mydas* *E. imbricata*	Y Y	*C. mydas* *C. caretta*	Y Y
Soft-bottom habitat – reef lagoons			*C. caretta*	Y
Soft bottom habitat – inter-reef areas	*L. olivacea*	U		
Soft-bottom habitats – coastal areas (sand or mud bottoms) estuaries	*N. depressus shallow water* *L. olivacea 'deep' water*	Y Y	*C. caretta*	Y
Oceanic pelagic waters Oceanic coastal waters	*D. coriacea*	Y	*D. coriacea* *C. caretta*	Y Y

Y=confirmed in habitat, U=unknown

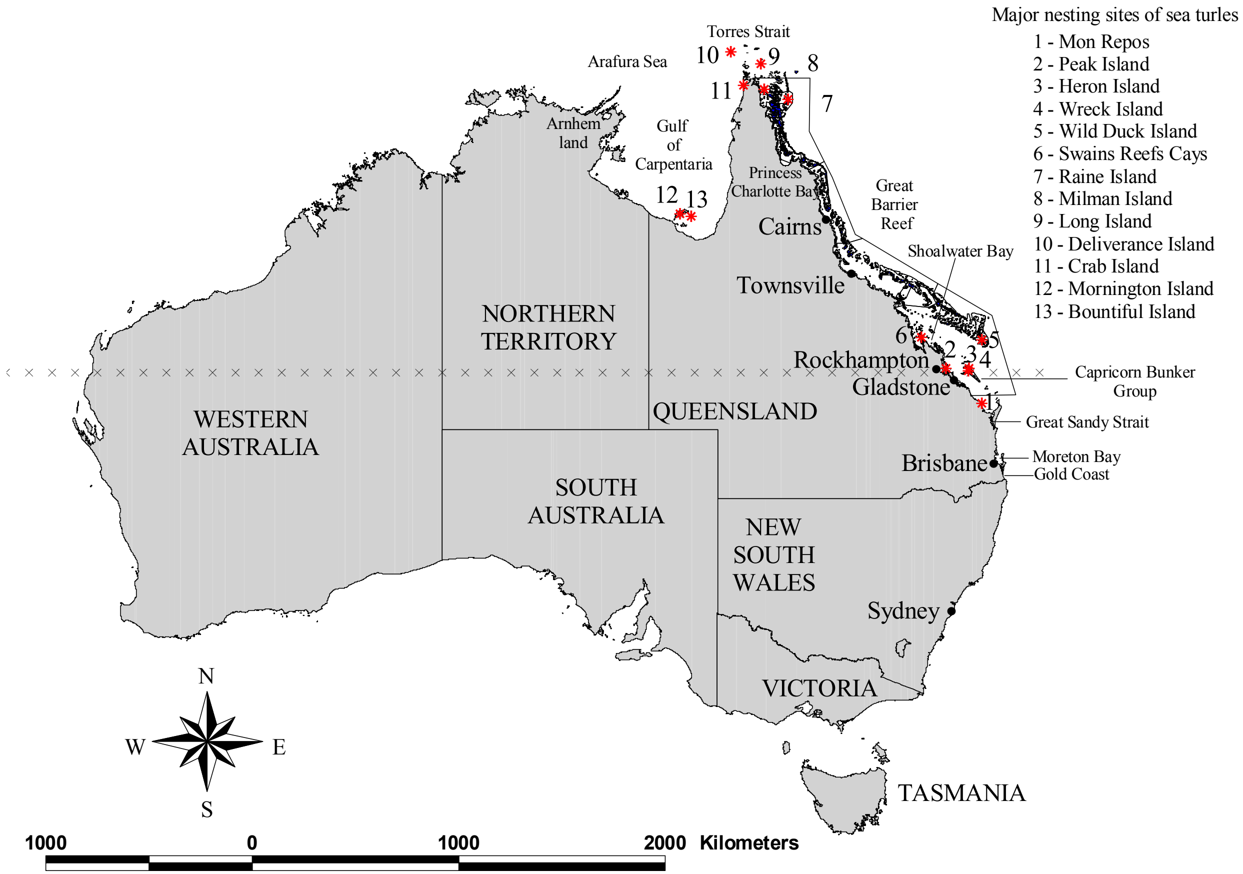
2.4.2 Distribution in Australia

C. mydas is numerically common, being abundant on reef habitats of the Great Barrier Reef (i.e., ~45 *C. mydas* per km^2, Chaloupka and Limpus 2001). *C. mydas* is also abundant in the major seagrass beds in northeastern Australia (Guinea 1994; Limpus 1981), such as those in Moreton Bay, Great Sandy Strait, Hervey Bay, Shoalwater Bay, Princess Charlotte Bay and Torres Strait. *C. mydas* populations in Queensland and the Northern Territory possibly represent the largest remaining stocks of *C. mydas* in the South Pacific basin (Dr Colin Limpus, QNPWS, personal communication 1998). There are three sub-populations (= nesting assemblages) in northeastern Australia (Figure 2.2): (i) a southern Great Barrier Reef aggregation (i.e., Capricorn Bunker Group) that draws individuals from feeding-grounds in the central and southern Great Barrier Reef and the eastern Pacific; (ii) a northern Great Barrier Reef aggregation (i.e., Raine Island and Moulter Cay) that draws individuals from feeding-grounds in the northern Great Barrier Reef and Torres Strait; and (iii) a Gulf of Carpentaria aggregation (i.e., Wellesley and Mornington Islands) that draws individuals from the Gulf of Carpentaria and the Arafura Sea (Limpus 1994).

E. imbricata is found most commonly on hard-bottomed habitats, such as the coral and rocky reefs of the Great Barrier Reef, Torres Strait, Western Australia and the archipelagos of the Northern Territory (Miller 1994). *E. imbricata* does not occur on feeding-ground in large numbers, with an estimated mean density on reefs of the southern Great Barrier Reef of ~3.3 *E. imbricata* per km^2 (Limpus 1992). *E. imbricata* is not caught commonly in Australian prawn trawl fisheries (Poiner *et al.* 1990; Robins 1995), suggesting that this species is unlikely to be abundant in inshore turbid waters of the southern and central Great Barrier Reef. Immature *E. imbricata* occur on reefs in the southern Great Barrier Reef, but few adults occur on reefs south of Cairns, suggesting a developmental migration from feeding-grounds in the southern Great Barrier Reef to those in the northern Great Barrier Reef and Torres Strait (Limpus 1992; Miller 1994). There are two major nesting aggregations of *E. imbricata* in eastern Australia (Figure 2.2): (i) on the northern Great Barrier Reef inner shelf cays (i.e., Milman and Johnson Islands); and (ii) in central Torres Strait (i.e., Long and Bet Islands, Limpus 1994). These breeding aggregations, which occur from January to April, draw individuals from feeding-grounds along the Queensland east coast and Arafura Sea (Limpus and Parmenter 1986).

C. caretta is most common in sub-tidal areas of sub-tropical Australia including coral and rocky reefs, seagrass meadows and large shallow bays and estuaries where molluscs and crabs are abundant (Limpus 1981; Limpus *et al.* 1994a). The mean density of *C. caretta* on coral reefs in the southern Great Barrier Reef is estimated at ~4.5 sea turtles per km^2 (Chaloupka and Limpus 2001). There is a single sub-population in eastern Australia (Limpus *et al.* 1984a; Limpus 1985; Gyruis and Limpus 1988) drawing individuals from feeding-grounds in New Caledonia to about the Arnhemland coast (i.e., western Gulf of Carpentaria) and from 35^{o}S (i.e., New South Wales) to the Solomon Islands (Limpus *et al.* 1992; Dr Colin Limpus, QNPWS, personal communication 2002). However, Limpus and Reimer (1994) report that most tag returns of *C. caretta* tagged at nesting beaches are concentrated between Gladstone and the Gold Coast (i.e., southern Queensland), suggesting a greater concentration of *C. caretta* in sub-tropical feeding-grounds (Limpus *et al*. 1994a, Chaloupka and Limpus 2001). The main nesting season for *C. caretta* is from late October to early February and is concentrated in the southern Great Barrier Reef at three locations (Figure 2.2): (i) the mainland coast (Mon Repos, Wreck Rock); (ii) the Capricorn Bunker Group of Islands (Wreck, Erskine and Tyrone Islands); and (iii) Swains Reef Cays (Pryce, Bylund Cays) (Limpus *et al.* 1992).

Figure 2.2 Nesting locations of sea turtle species in northern Australia



Major feeding-grounds for *N. depressus* occur in tropical waters of the Gulf of Carpentaria and the shallow inshore turbid habitats of the east Australian coast to about 25°S (Limpus *et al*. 1983a; Parmenter 1994). All tag returns of *N. depressus* tagged at nesting beaches have been reported from the inshore area between Rockhampton and Torres Strait (Limpus and Reimer 1994), supporting speculation that *N. depressus* are most abundant in tropical waters. There are three major nesting areas for *N. depressus* in northeastern Australia (Figure 2.2): (i) in the southern Great Barrier Reef lagoon (i.e., Peak, Wild Duck, Avoid, Curtis and Facing Islands); (ii) in the northeast Gulf of Carpentaria and eastern Torres Strait (i.e., Crab and Deliverance Islands); and (iii) the Wellesley Group in the south west Gulf of Carpentaria (i.e., Bountiful, Pisonia and Rocky Islands, Limpus *et al*. 1983a; Limpus *et al.* 1983b). Minor nesting occurs along the Queensland east coast between Townsville and Bundaberg.

L. olivacea is reported most commonly from the soft-bottom aquatic habitats of northern Australia, especially the Gulf of Carpentaria (Limpus *et al*. 1983b; Harris 1994) and along the Queensland east coast as far south as Cleveland Bay adjacent to Townsville (Harris 1994). *L. olivacea* has been reported from sub-tropical waters (e.g., Moreton Bay, Robins and Mayer 1998), although the frequency of reports is less common. There are no documented large nesting aggregations of *L. olivacea* in northern Australia, only sporadic nesting along the Gulf of Carpentaria, including Western Cape York Peninsula (Limpus 1994). As such, this species does not have a formal tag-recapture program and most of the data on Australian *L. olivacea* are derived from incidental capture in prawn trawling operations.

Dermochelys coriacea is often reported in the oceanic waters of southern Queensland, New South Wales, Western Australia and Tasmania, as well as from the shallow continental waters in the Gulf of Carpentaria (Limpus *et al.* 1984b; Marsh *et al.* 1999). Nesting of *D. coriacea* in Australia is recorded infrequently i.e., less than one individual per year along the Bundaberg coastline (Limpus and McLachlan 1994).

Despite relatively good general knowledge on the sea turtles in Australian waters, their spatial distribution is poorly quantified (Dr Colin Limpus, QNPWS, personal communication 1998) and there is insufficient information on the location of key feeding-grounds for effective management (Dobbs 2001). Broad scale maps of the relative density of sea turtles in northern Australia would provide insights into the location of significant feeding-grounds. Potential factors that contribute to an area being a critical feeding-ground could be identified from these maps, particularly for the lesser-known species such as *N. depressus* and *L. olivacea*. This information would be valuable to the conservation management of sea turtles. This thesis addresses this gap in the knowledge by developing a predicted relative spatial distribution of sea turtles in waters of the Queensland east coast (see Chapter 5).

2.5 POPULATION DYNAMICS

Sea turtles are long-lived species having relatively slow-growth, delayed maturity, low reproduction rates and naturally low adult mortality (Heppell *et al.* 1999). They are difficult animals to count directly to infer absolute population size because individuals of a sub-population can be widely dispersed across numerous feeding-grounds and conversely, localised feeding-grounds can contain a mixture of individuals from a number of sub-populations. Therefore, trends in the population size of sea turtles are often monitored through indices such as counts of nesting females (Bjorndal *et al.* 1993; Hopkins-Murphy and Murphy 1994), and the rate of recruitment of juveniles to neritic feeding-grounds (Chaloupka and Limpus 2001) or neophyte females (i.e., first time nesters) to nesting beaches (Limpus *et al.* 1994a).

2.5.1 Population models

The relative consequences of human impacts and management scenarios on sea turtle populations have been explored through deterministic age- or stage-structured matrix models (Richardson and Richardson 1981; Frazer 1986; Crouse *et al.* 1987; Crowder *et al.* 1994, 1995; Heppell *et al.* 1996). These models assume the non-impacted stable population is density independent and time invariant (Caswell 2000). All models except Heppell *et al.* (1996) have no spatial stratification to account for spatial heterogeneity in mortality (i.e., protected or refuge habitats versus high impact habitats). In general, the models have simulated declining trends in sea turtle populations as a result of human impacts such as direct harvesting or trawl by-catch mortality. Deterministic matrix population models have been criticised for their use of enumerated and life-table based survival estimates (Chaloupka and Limpus 2002). Like most simulation, model outputs are representative of the accuracy or validity of input parameters, but deterministic matrix models do not recognise the plasticity of parameters such as survival, age-at-first-maturity and fecundity, and do not accommodate any feedback mechanisms. As a consequence, any mortality added to the steady-state simulation results in the eventual extinction of the population. More recent matrix models of sea turtle population dynamics attempt to overcome some of the limitations of previous models by encompassing stochastic simulation of demographic parameters (Chaloupka 2002).

One of the main purposes of matrix population modelling has been to explore the elasticity of input parameters through sensitivity analyses and to assess different management options. Sensitivity analyses highlight the consequence of changes in parameters on population trends e.g., increasing survival through various management actions. The initial deterministic matrix models suggested that survival rates of late juveniles and adult stages were relatively more important in ensuring population maintenance and growth than mortality of eggs or fecundity of females (Crouse *et al.* 1987; Crowder *et al.* 1994; Heppell *et al.* 1996). Elasticity analyses derived from deterministic matrix models are usually single-element elasticity analyses. These have been criticised as a limited approach to assessing model sensitivities where environmental stochasticity is an important influence of demographic parameters, such as occurs in sea turtle populations (Chaloupka 2002). Demographic loop analyses that account for multi-stage effects have been used to suggest that fertility (i.e., breeding probability and fecundity) is also relatively important in ensuring population maintenance and growth (Chaloupka 2002). Population fertility is a difficult aspect for management by human intervention, although Chaloupka (2002) suggests that increasing the survival of eggs and hatchlings (i.e., headstarting) might be an alternative strategy for improving population fertility. Nonetheless, all modelling approaches indicate that minimising anthropogenic mortality (either direct or incidental) on adult and sub-adult sea turtles remains the most important strategy to prevent declines of sea turtle populations and to promote the recovery of already depleted populations.

2.5.2 Impact of sea turtle by-catch on population trends

Sea turtles have inherently low recovery capability (sensu Stobutzki *et al.* 2001a). Therefore, sea turtle populations have an inherently low capacity to sustain mortality associated with trawl capture (Crouse *et al.* 1987; Crowder *et al.* 1994; Crouse 1999; Heppell *et al.* 1999). However, it is difficult to translate qualitative statements on levels of trawl by-catch mortality into quantitative estimates of the number of sea turtles killed that would cause declines in sub-population size of the scales observed in eastern Australia. Heppell *et al.* (1996) estimated that a combined by-catch mortality in the prawn trawl fisheries of northern Australia in the low hundreds of individuals would be sufficient to cause a decline similar to that observed at nesting beaches of *C. caretta* (Limpus and Reimer 1994). This estimate was based on a hypothetical total population size of 946,000 *C. caretta* that was back-calculated from a hypothetical unimpacted population of 5,000 adult females. This assumed a population composition of 3.0:1.3:1.0 of sub-adults:pubescents:adults as reported for Heron Reef (Heppell *et al.* 1996). A gradual decline was simulated from the hypothetical population by annually removing (i.e., killing) 100 female sea turtles or 345 female and male, immature and adult sea turtles. Chaloupka and Limpus (1998) derived a similar estimate when simulating the population response of the east Australian *C. caretta* sub-population to a 90% fox-induced mortality on eggs and various levels of trawl by-catch mortality. The fox-induced egg mortality caused a major decline in sub-population numbers and trawl by-catch mortality in the low hundreds of individuals was sufficient to contribute to the observed decline.

Impacts of trawl by-catch mortality have been simulated primarily for *C. caretta* because of the availability of data, with some exploratory models for *Lepidochelys kempii* (TEWG 2000). The impact of by-catch mortality on other sea turtle species is less well understood. All sea turtles share a life-history strategy similar to that of *C. caretta* and it would be expected that by-catch mortality would contribute to sub-population declines. Assessment of the susceptibility of sea turtle species to trawl by-catch suggests that some species are more susceptible than others. Those species that display habitat preferences for the shallow, soft-bottom habitats that are typical of penaeid trawl grounds (i.e., *C. caretta*, *N. depressus*, and *L. olivacea*) should be more susceptible to capture than those species that favour only coral and rocky reef habitats (i.e., *E. imbricata*).

However, the relative significance of trawl-associated mortality has not been calculated for the species that occur in Australian waters other than *C. caretta* and *C. mydas* because current sub-population sizes are unknown. Speculating on the relative significance of by-catch mortality for each species is further complicated by other sources of anthropogenic mortality. For example, east Australian sub-populations of *C. mydas* are thought to be an order-of-magnitude larger than the sub-population of *C. caretta*. This is based on the estimated density of *C. mydas* on coral reef habitats (Chaloupka and Limpus 2001) and nesting female numbers, although fluctuation in nesting numbers of *C. mydas* is highly correlated with the El Nino phenomenon (Limpus and Nicholls 1988). Reef habitats are likely to provide a vast refuge for *C. mydas* from prawn trawl by-catch mortality. However, *C. mydas* in northern Australia is subject an indigenous harvest of 1,000s of individuals per year (Smith and Marsh 1989; EA 1998), although the harvest is comprised of numerous groups of indigenous hunters whose sea turtle take occurs on localised coral reefs when *C. mydas* is relatively easy to catch. In addition, *C. mydas* is susceptible to boat strikes, although the mortality as a consequence of boats strikes is thought to be in the order of tens-of-individuals per year (Haines and Limpus 2000).

Lack of knowledge complicates the assessment of the relative significance of the trawl by-catch of *N. depressus* and *L. olivacea.* However, the mitigation of trawl by-catch mortality on all species of sea turtles is consistent with a theoretical (i.e., using the susceptibility framework of Stobutzki *et al.* 2001a) and precautionary approach to managing the impacts of fishing.

2.6 DIVING ABILITIES

Sea turtles are air-breathing reptiles that can remain submerged for up to 98% of the time (Lutz and Bentley 1985). They are well adapted to undertake breathhold diving as a consequence of their respiratory physiology, blood physiology and cardiac system (Table 2.4). Sea turtles have an efficient system of oxygen transportation to maximise the use of limited oxygen stores. In addition, the vital tissues and organs of sea turtles can tolerate significant periods with negligible levels of oxygen (Lutcavage and Lutz 1996).

Table 2.4 Aspects of sea turtle physiology that enhance their diving capacity

Respiratory Physiology	• Inhales before diving [A] • Uses lungs as an oxygen store [B] • Exchanges the full lung volume each breath [C] • Possesses highly efficient lungs (i.e., compartmentalised giving a large surface area [D]
Blood Properties	• Possess blood with high O_2 affinity, enabling the O_2 stored in the lungs to be transferred into the blood as the dive progresses [D] • Releases glucose continuously from the liver keeping blood glucose levels stable during the dive [D] • Tolerates large changes in blood pH and blood gases [C]
Cardiac System	• Possesses a three chambered heart, which enables CO_2 saturated blood to be shunted past the lungs thereby keeping CO_2 in the tissues and blood, ensuring that all blood O_2 is used [D]
Tissue Tolerance	• Possesses tissues and vital organs able to withstand anoxic conditions [C] • Possesses mitochondrial respiratory enzymes (cytochrome a, a3) that are not damaged irreversibly when deprived of oxygen. [C] • Possesses a brain that remains functional under anoxic conditions [C]

[A] Berkson 1966; [B] Jackson 1985; [C] Lutz and Bentley 1985; [D] Wood *et al.* 1984

2.6.1 Natural dive patterns

Diving activity varies between species, life stages and individuals, indicating that submergence patterns are a function of the physiological abilities of an individual sea turtle as well as its ecological situation (Lutcavage and Lutz 1996).

Time submerged

Evidence from laboratory and field studies of voluntary diving in sea turtles, indicates that non-migrating sea turtles spend between 80% and 98% of their time submerged (Table 2.5). This varies with species, size of individuals and diurnal periodicity i.e., day or night. The lowest percent-time submerged reported in the literature is 65%, for an adult female *L. kempii* during a post-nesting migration (Gitschlag 1996).

Dive intervals

Sea turtles usually undertake dives that are relatively short, i.e., less than 45 minutes in duration (Lutz and Bentley 1985; Lutcavage and Lutz 1991; Byles and Dodd 1989). Dive interval also varies with species, size, diel periodicity and ecological situation (Table 2.5). Renaud and Carpenter (1994) report a complex picture of dive interval variability during satellite telemetry of *C. caretta* in the Gulf of Mexico. Mean dive interval varied with season, being shortest in summer, and varied between day and night, being shortest during the day (Table 2.5). Seasonal variation is likely to be related to water temperature as sea turtles are ectotherms and their metabolic rate is determined by their size and water temperature (Lutz and Bentley 1985). Dive intervals are also strongly related to activity type, being long and regular when juvenile *C. mydas* are stationary or resting, and short (i.e., ~10 minutes) when they are swimming or active (Brill *et al*. 1995). Observation of *C. caretta* in captivity suggests that daily behaviour patterns are equally divided between activity (e.g., swimming) and resting (Dodd 1988). van Dam and Diez (1997) separated dive patterns into foraging and resting dives. Foraging dives were defined as “dives with depth fluctuations greater than 0.5m within 2 minutes, during at least 50% of the submergence interval” and resting dives were defined as dives that were not foraging dives.

Most natural dives occur within the limits of the oxygen store of the sea turtle (Lutcavage and Lutz 1996), but this is not the case when sea turtles are forcibly submerged, such as when caught in a trawl net. Sea turtles frequently die as a consequence of trawl capture, despite their physiological tolerance to anoxic conditions (Kemmerer 1989).

Table 2.5 Parameters associated with the diving patterns of sea turtles

Species, size, diurnal period	Study Condition	Percent time submerged	Dive interval (mins)		Comments
			Routine	Max.	
Caretta. caretta, sub-adult	Open water	80% to 94%	19 - 30		Lutcavage and Lutz 1996 citing Soma 1985 and Byles 1988
C. caretta, 20kg	Laboratory tanks	98%	4.85 ± 3.0 (s.d.)	27.2	Method of determining time of voluntary dives not specified, but likely to be observation; Lutz and Bentley 1985
C. caretta, sub-adult	Laboratory tanks	86%	16.1	40.0	Ventilation measured by pneumotachography; Lutcavage and Lutz 1991
C. caretta, adult female, pnm[1]	Open water		25.7	40.0	Satellite telemetry; Byles and Dodd 1989
C. caretta – spring, day	Open water		29.8 ± 3.3 (s.e.)		Satellite telemetry; Renaud and Carpenter 1994
- spring, night			44.7 ± 3.4 (s.e.)		
– summer, day,	Gulf of Mexico	90%	11.6 ± 1.8 (s.e.)		
- summer, night			23.0 ± 2.7 (s.e.)		
– autumn, day			26.7 ± 2.8 (s.e.)		
- autumn, night			42.9 ± 2.9 (s.e.)		
– winter, day		95%	74.0 ± 9.0 (s.e.)		
- winter, night			156.4 ± 11.8 (s.e.)		
Chelonia mydas, 20kg,	Laboratory tanks		4.74 ± 2.0 (s.d.)	26.9	Method of determining time of voluntary dives not specified, but likely to be observation; Lutz and Bentley 1985
C. mydas	Open water, i.e., an embayment	96%	1.8 ± 0.8		Radio tracking; Renaud and Carpenter 1994; Williams and Renaud 1998
Eretmochelys imbricata, immature, day	Coral reef	86%	18.9 foraging 39.7 resting		Foraging dive during day, resting dives during night; TDRs[2], 8 second interval; van Dam and Diez 1997
E. imbricata, immature, night		85%, 96%			TDRs[2], 8 second interval; van Dam and Diez 1997
Lepidochelys kempii	Open water, i.e, an embayment,	93%	6.3 ± 0.2		Radio tracking; Renaud and Carpenter 1994; Williams and Renaud 1998
L. kempii, sub-adult	Open water	96%	12.7-18.1		Lutcavage and Lutz 1996 citing Soma 1885 and Byles 1988
L. kempii, two juveniles, one adult female, pnm	Open water	94% 65%			Diel difference with long dives at night; radio tracking; Gitschlag 1996

[1] pnm = post-nesting migration; [2] TDR = Temperature Depth Recorders

2.6.2 Effect of trawl-capture

Sea turtles caught in trawl nets are effectively subjected to a forced submergence that results in the sea turtle being in one of four conditions when removed from the net: (i) active; (ii) injured; (iii) comatose; or (iv) dead. The impact of the forced submergence varies for each sea turtle caught depending on how long the sea turtle has been trapped in the net (Lutcavage 1992), the size of the sea turtle (Hillestad *et al*. 1981), the water temperature (Lutz *et al.* 1989) and the species involved (Lutcavage 1992).

Sea turtles have been observed to actively avoid demersal trawl nets, either by moving out of the path of the net (Standora *et al.* 1994) or repeatedly trying to outswim the net (Ogren *et al.* 1977). The duration of avoidance behaviour varies between sea turtles, depending upon individual levels of stored oxygen and energy reserves. Bursts of swimming activity deplete the stored oxygen of the sea turtle more quickly than natural diving behaviour (Lutz and Dunbar-Cooper 1981).

Lutz and Bentley (1985) and Ahern (1994) noted that sea turtles forcibly submerged under laboratory conditions initially struggle vigorously, then are quiescent for long periods, interrupted by occasional short bursts of activity. This may be true of sea turtles forcibly submerged in demersal trawl nets, suggesting that oxygen and energy reserves will be depleted during the time the sea turtle is trapped in the net.

Laboratory studies have also demonstrated that sea turtles have an exceptional tolerance to prolonged periods of submergence, including those during which their blood and lungs are fully depleted of oxygen (Belkin 1963; Berkson 1966; Gatten 1987; Lutcavage 1992). Sea turtles forcibly submerged for up to 120 minutes demonstrated varying rates of recovery, with one *C. mydas* observed not to inhale until six hours after it was removed from a forced submergence (Berkson 1966). The recovery of forcibly submerged sea turtles, as measured by the time taken for lactic acid levels to return to base levels, is variable but occurs within 24 hours (Berkson 1966; Lutz and Dunbar-Cooper 1981).

2.6.3 Delayed mortality post-release

Most sea turtles caught in trawling operations are alive when landed and are released in a conscious and active condition (Henwood and Stuntz 1987; Poiner *et al*. 1990; Robins 1995). Some of these sea turtles are speculated to suffer delayed post-release mortality (Limpus and Reed 1985b), but the prevalence of such mortality is unknown. Some trawl-caught sea turtles have been observed subsequently at nesting beaches (Limpus *et al.* 1992) and therefore must have recovered. However, other sea turtles have been found dead after being released alive from a trawl-capture (Limpus *et al.* 1992). This delayed post-release mortality is possibly a result of acute internal injuries such as fractured skulls, water in the lungs (i.e., drowning) or irreversible changes to physiology (Limpus *et al.* 1992, Parmenter 1994).

Delayed fatal reactions to capture have been reported for other aquatic reptiles. For example, some Australian estuarine crocodiles (*Crocodylus porosus*) alive and active after capture in traps have later died (Seymour *et al*. 1989). Struggling by the crocodile during capture causes elevated levels of lactic acid as indicated by decreased blood pH. In this critical metabolic state, any additional period without oxygen may prove fatal (Seymour *et al.* 1989). Trawl-caught sea turtles also have decreased blood pH (Stabenau *et al*. 1991), indicating that lactic acid levels are raised and suggesting the possibility that conscious trawl-caught sea turtles may die at some later stage, like crocodiles. This type of delayed mortality may account for sea turtles that were caught in a demersal trawl net, released alive into the water, and then found as a carcass washed up on the beach adjacent to the trawl grounds within 12 hours (Limpus and Reimer 1994).

2.6.4 Modified behaviour post-release

Klima *et al*. (1988) and Dr Robert Shoop (University of Rhode Island, personal communication 1994) speculate that sea turtles in an active condition that are recovering from a trawl-capture would display modified behaviour such as slowed escape responses or floating at the surface to breathe. They suggest that these behaviours would make recovering sea turtles more likely to be hit by boats, attacked by sharks or captured in another trawl net. For sea turtles to be more susceptible to boat strike or shark attack, released sea turtles would need to spend a greater amount of time at the surface or in the water column than under normal circumstances. For sea turtles to be more susceptible to multiple capture, released sea turtles would need to remain in the vicinity of the trawling grounds and spend a greater amount of time on the sea floor than under normal circumstances.

Mortality rates of trawl-caught sea turtles have a large influence on the overall impact of trawl fisheries on sea turtle populations. If delayed or secondary mortality is a frequent event, then trawling could be having a much greater impact on sea turtle populations than currently estimated. This thesis investigates the post-trawl responses of sea turtles to capture (see Chapter 4), which may indicate whether additional mortality should be considered during assessments of the impact of sea turtle by-catch.

CHAPTER 3. SCALE AND COMPOSITON OF SEA TURTLE BY-CATCH

3.1 CHAPTER SUMMARY

Sea turtle by-catch data collected by selected commercial fishers during trawling operations were used to estimate the average annual number and species composition of sea turtles caught and killed in the Queensland East Coast Trawl Fishery prior to the regulation of Turtle Excluder Devices (TEDs). The work was undertaken to determine if the size and composition of sea turtle by-catch in this fishery was of the scale required to significantly contribute to observed declines in nesting numbers of the east Australian sub-population of *Caretta caretta*. About 5,900 sea turtles were estimated to be caught annually in the waters adjacent to the Queensland east coast, with about 50% being *C. caretta*. Between 60 and 80% of the sea turtle catch was of immature size classes, depending on species. Mortality rates were calculated for (i) dead sea turtles (i.e., observed direct mortality); (ii) dead plus comatose sea turtles (i.e., observed potential mortality); and (iii) the relationship between observed mortality and tow duration (i.e., expected mortality). The mortality rate of sea turtles (i.e., all species pooled) for the Queensland East Coast Trawl Fishery (i.e., all sectors pooled) was 1.3% and 5.7% for observed direct mortality and observed potential mortality respectively. These rates are low compared to other trawl fisheries in northern Australia and the USA. This is probably a function of tow duration associated with the various sectors of the Queensland fishery. Mortality rates of sea turtle by-catch should be reported as: (i) sector-specific to take account for operational characteristics of fishing in local areas (e.g., tow duration); and (ii) species-specific to account for submergence capabilities of different species. Other species impacted by the Queensland East Coast Trawl Fishery were *Chelonia mydas*, *Natator depressus* and *Lepidochelys olivacea*. The fishery had minimal impact on *Eretmochelys imbricata* and *Dermochelys coriacea*. The combined catch and mortality of sea turtles in the trawl fisheries of northern Australia (i.e., the Queensland East Coast Trawl Fishery, the Northern Prawn Fishery and Torres Strait Prawn Fishery) was of sufficient magnitude to have contributed to the observed declines in nesting numbers of east Australian *C. caretta*. These results support the mitigation of sea turtle by-catch through the mandatory use of TEDs.

3.2 INTRODUCTION

3.2.1 Why quantify sea turtle by-catch when TEDs are mandatory?

A review of threats to sea turtles concluded that prawn (=shrimp) trawling had the greatest anthropogenic impact on sea turtle populations (Magnuson *et al*. 1990). Six species of sea turtle occur in Australian waters and were threatened with trawl by-catch mortality prior to the regulation of Turtle Excluder Devices (TEDs) into the trawl fisheries of northern Australia between 1999 and 2002. Indeed, one of the main drivers of TED regulations was the decline in numbers of nesting females of the east Australian sub-population of *Caretta caretta* (loggerhead turtles). Declines in nesting indices of sea turtles probably represent long-term impacts from numerous anthropogenic activities throughout the range of a sub-population (Heppell *et al.* 1996). Limpus and Reimer (1994) speculated that the major causes of the decline of *C. caretta* were the deaths of immature and adult sea turtles in trawl nets and the predation of eggs by introduced foxes. More recently, Chaloupka and Limpus (1998) also considered the mortality of pelagic juvenile sea turtles in high seas longline fisheries as a significant threat to the east Australian *C. caretta* sub-population.

Several authors have speculated on the scale of historic (i.e., pre-TED) sea turtle by-catch in the trawl fisheries of northern Australia. Limpus and Reimer (1994, p45) speculated that the "order-of-magnitude of the kill of *C. caretta* in commercial fisheries in northern and eastern Australia is many hundreds, possibly greater than a thousand annually" and includes drowning in crab-pot buoys, gill nets and longlines, although these sources of mortality were considered to be minor in comparison with the trawl mortality. Dredge and Trainor (1994) speculated that sea turtle by-catch would be common in certain inshore sectors of the Queensland east coast and that mortality might vary as a function of tow duration. They did not speculate on the size of sea turtle by-catch in Queensland. The potential scale of sea turtle by-catch in trawl fisheries of northern Australia has been explored through population modelling (Heppell *et al.* 1996; Chaloupka and Limpus 1998). Heppell *et al*. (1996) simulated a gradual decline from a hypothetical population of 5,000 adult female *C. caretta* (or a total population of 946,000) based on the removal of 100 adult female sea turtles per year (or 345 juveniles/adults, males/females). Chaloupka and Limpus (1998) simulated the impact of medium (=250 sea turtles) and heavy (=1,000 sea turtles) fishing mortality on a hypothetical population of *C. caretta* also impacted by fox predation of eggs.

Speculation or computer simulation of potential trawl by-catch mortality are useful tools for raising awareness or assessing competing sources of mortality, but by themselves do not quantify the scale of by-catch occurring in a fishery. Baseline data on the location, scale, species composition, size composition and mortality rates of sea turtle by-catch are fundamental for the comprehensive management of trawl impacts and the assessment of possible population responses to the removal of by-catch mortality through the use of TEDs.

3.2.2 How is this information useful to by-catch management?

Once TEDs are introduced into a fishery, the main research and management concerns become: (i) the effectiveness of TEDs; and (ii) what changes (i.e., size-class frequency or abundance trends) would be expected in feeding- and nesting-grounds that could indicate the population response to TEDs.

Crowder *et al.* (1994) simulated the response of sea turtle populations to the introduction of TEDs in USA shrimp trawl fisheries. They suggest that an increase in sea turtle abundance should occur, but the timing and scale of the increase were highly dependent on the changes in the stage-specific mortality rates resulting from the use of TEDs. Historic estimates of the mortality of sea turtles caught in USA trawl fisheries (Henwood and Stuntz 1987) did not include information on the stage-specific mortality. This information is difficult to obtain once TEDs are regulated into a fishery. Thus, information on the stage-specific impacts of trawl fisheries is important, not only for determining the negative impacts on populations prior to TED regulation, but also for predicting the population response after TEDs are regulated into a fishery.

The most reliable method of assessing the stage of development of sea turtles is internal examination of the gonads by laparoscope because sea turtles display variable size at maturity (Miller 1996). Laparoscopy of sea turtle by-catch during commercial trawling operations is not practical. However, size can be used as an approximate indicator of maturity status for the purpose of assessing the proportion of sea turtle by-catch that is immature versus mature. This is useful information for understanding the impacts of trawling on sea turtle demography (Crowder *et al.* 1994).

3.2.3 Other benefits

There are several other benefits of quantifying sea turtle by-catch, particularly from data collected prior to TED regulations. These include the scale of the historic impact on all sea turtle species, as modelling of trawl impacts has focused on *C. caretta*. Historic trawl impacts in northern Australia are only partially documented for species other than *C. caretta* (Poiner and Harris 1996; Robins and Mayer 1998), particularly those that inhabit the turbid, coastal waters typical of trawl fisheries such as *Natator depressus* (flatback turtles) and *Lepidochelys olivacea* (Pacific Ridley turtles).

3.2.4 Aims of this chapter

In this chapter, I estimated the scale and species composition of sea turtle by-catch in the Queensland East Coast Trawl Fishery based on six years of reported sea turtle captures from selective logbook program. This builds on the results published in Robins (1995), which were based on two years of data. I also assessed the reported mortality rates of trawl-caught sea turtles. The assessment was expanded to species and sector specific mortality rates, which were not reported upon in Robins (1995). The results were considered in terms of the contribution of the Queensland East Coast Trawl Fishery to observed declines in nesting numbers of *C. caretta*.

3.3 MATERIALS AND METHODS

3.3.1 Recording of sea turtle by-catch

Sea turtle by-catch monitoring program

Data on sea turtle by-catch in the Queensland East Coast Trawl Fishery were derived from a selective logbook program that officially ran from January 1991 to December 1996. I initiated and managed the 'sea turtle by-catch monitoring program' and personally approached fishers to participate in the program. I discussed with potential participants the issues associated with sea turtle by-catch, the fisher's opinions on these topics, the research objectives and the data to be collected. Only those individuals who expressed and maintained keen interest in recording information were selected. I maintained regular contact (i.e., every three months) with participating fishers through phone calls, written material and personal visits to reinforce the need for honest recording.

Participating fishers were supplied with a sea turtle data kit that included standardised data sheets, a species identification chart, a flexible tape measure and guidelines on measuring the curved carapace length (CCL) of sea turtles[3]. Fishers were instructed to record the date, time, location, tow duration, tow depth, species and CCL of captured sea turtles. Fishers were requested to record the sea turtles as 'unidentified' if they were unsure of the species identification. From 1993 onwards, fishers recording more than five sea turtle captures per year were provided with disposable cameras so that the species identification of sea turtles could be validated. The physical condition of sea turtles upon capture was also recorded (Table 3.1). These classifications were derived from discussions with Dr Ian Poiner (CSIRO), Dr Aubrey Harris (BRS) and Dr Colin Limpus (QPWS).

Table 3.1 Classification of sea turtle condition upon capture

Physical condition	Signs and symptoms
Healthy	Moving, flapping aggressively
Injured externally	Wounded externally but otherwise healthy
Comatose	Dazed, few external movements, slight signs of breathing
Dead	No movement, head limp, extended and flops to ground, no signs of breathing, eyes do not respond to touch

Data screening

Sea turtle by-catch data were screened for reliability i.e., whether the data sheets were returned consistently and whether the datasheets contained fundamental information (i.e., date, location, species and condition). Discussions with Dr Colin Limpus (QPWS) suggested that records of trawl-caught sea turtles outside the size range recorded in studies by the Queensland Turtle Research Group (QTRG) should be treated with caution, as it was likely participating fishers had mis-identified the species or inaccurately measured the size. Reported sea turtle captures that were beyond the size limits recorded by the QTRG were treated as unidentified sea turtles of unknown size. (Recorded size limits are presented with the results in Table 3.8).

[3] Fishers were given the following instructions for measuring CCL: using the supplied flexible tape measure, measure along the midline of the shell of the turtle, from the very front of the shell, near the neck, over the shell to the rear edge of the shell. A diagram of measurement was also included on the back of every data sheet.

Classification of sea turtles caught into adults and immatures

As discussed earlier, size is not a reliable indicator of maturity status (Miller 1996), as all sea turtle species display variable size at maturity. However, assessing the proportion of sea turtle by-catch that is immature versus mature is useful from the context of understanding the impacts of trawling on sea turtle demography (Crowder *et al.* 1994). The approximate size at maturity was taken from the mean size at maturity reported in the literature and rounded to the nearest 5 cm size class because laparoscope studies indicate that *E. imbricata*, *C. mydas* and *C. caretta* "typically begin breeding at just less than the average breeding size of the population" (Miller 1996, p 54). Trawl-caught sea turtles were classified as either immature or mature depending on whether the reported CCL was less than or greater than the approximate size at maturity for each species.

3.3.2 Recording and allocation of effort

All fishers participating in the Queensland East Coast Trawl Fishery are required to complete a daily logbook of catch and effort and submit this information to the agency that manages the fishery, the Queensland Fisheries Service. The daily catch of target and non-target species (by weight or number) of each vessel is recorded at a scale of 30^2nm (referred to as 'CFISH grids'), 6^2nm (referred to as 'CFISH sites') or as a point estimate with latitude and longitude. Logbook data are not formally cross-validated with independent information sources, such as records from seafood wholesalers, making it difficult to assess the reliability of commercial catch and effort data. Anecdotal reports suggest that some mis-reporting of logbook information occurs, but the scale and direction are unknown[4]. This is an inherent potential source of error in the commercial catch and effort data. However, the scale of the fishery, both in terms of the spatial distribution of effort and the number of vessels participating, suggest that mis-reporting should at worst blur trends in the data rather than significantly bias the data.

Catch and effort information for commercial trawl fishers in the monitoring program (hereafter referred to as the 'sample fleet') were retrieved from the logbook database, as were the catch and effort information for all commercial trawl fishers (hereafter referred to as the 'total fleet') for the years 1991 to 1996 inclusive. Invalid or incomplete data

[4] Anecdotal reports suggest that catches of target species (i.e., prawns and scallops) are sometimes under-reported as a consequence of illegal sales and to under-represent income for taxation purposes. Around 1996, rumours amongst the fishing industry of potential spatial zoning and the likely restriction of effort based on historic fishing effort

(e.g., land-locked records of fishing effort or catch with no effort location) were removed prior to analysis.

Stratification

The Queensland East Coast Trawl Fishery can be subdivided into nine sectors based on target species (Table 3.2). Stratifying the Queensland East Coast Trawl Fishery by fishing sector is a useful way of partitioning the spatial and temporal complexities of the fishery (Robins and Courtney 1999) and encompasses differences in operating characteristics (e.g., net specifications, tow duration and speed) that are likely to influence the catchability of sea turtles (Dredge and Trainor 1994).

Table 3.2 Sector characteristics of the Queensland East Coast Trawl Fishery

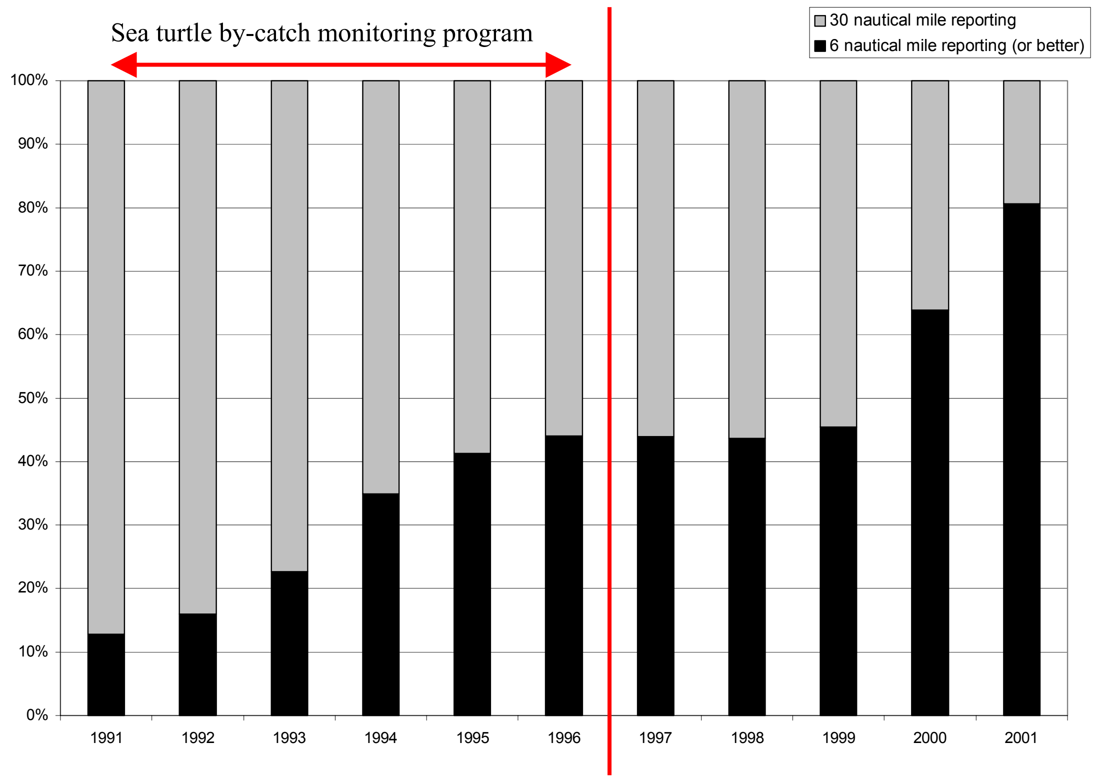
Sector and target species	**Main location**	**Main season**	**General characteristics** (Average tow ± s.d.)
Tiger prawn *Penaeus esculentus* *P. monodon*	North of 19°30'S Inshore areas	March to May	Night fishery Tows 129 ± 44 mins. 95% effort < 20 m deep
Endeavour prawn *Metapenaeus ensis* *Metapenaeus endeavouri*	North of 19°30'S Inshore areas	March to May	Night fishery Occurs in conjunction with tiger prawn sector
Red spot king prawn *Melicertus longistylus* *Melicertus latisulcatus*	Northwest of 23°S, 152°E Inter-reef areas	May to Sept.	Night fishery Tows 128 ± 51 mins. 85% effort > 20 m deep
Eastern king prawn *Melicertus plebejus*	Southeast of 23°S 152°E 1. Offshore waters up to 200m deep	Sept. to May	1. Night fishery Tows > 120 mins. 88% effort > 20m deep
	2.inshore	Sept. to May	2. Night fishery Tows < 90 mins. 12% effort <20 m deep
Banana prawn *Fenneropenaeus merguiensis*	Between 16°S and 25°S Inshore, adjacent to major rivers	Feb. to May	Day fishery Tows 55 ± 28 mins. 97% effort < 20 m deep
School prawn *Metapenaeus macleayi*	Southern Queensland, around 25°30'S, 153°E	Feb. to April	Not annual fishery sporadic, shallow waters
Moreton Bay *Metapenaeus bennettae* *P. esculentus* *M. plebejus* (juvenile)	Large embayment adjacent to Brisbane city, located at 27°S 153°E	Sept. to May	Day and night fishery Tows 76 ± 29 mins. All effort <30 m deep
Scallop *Amusium balloti* *A. pleuronectes*	Central Queensland, between 19°S and 25°S	Nov. to April	Night fishery Tows 155 ±49 mins. 85% effort > 30 m
Stout whiting *Sillago robusta*	South of Sandy Cape, Fraser Island	April to Dec.	Restricted entry fishery Five vessels during study

(QFMA 1996) suggested that fishers might be over-reporting their fishing effort i.e., days fished. The problem of inflated effort potentially affects the results in Chapter 3 for estimates of sea turtle by-catch in 1996.

Units of fishing effort

A day of fishing per CFISH grid ($30nm^2$) was selected as the unit of effort for estimating annual sea turtle by-catch because for 1991 to 1996 greater than 50% of the total fleet effort was reported at a spatial precision of 30^2nm (Figure 3.1). The spatial precision of reported effort has improved, mostly as a consequence of increased use of Global Positioning Systems.

Figure 3.1 Spatial precision of effort in the Queensland East Coast Trawl Fishery as reported in logbooks



Allocation to sector

Effort was nominated as having occurred in a fishing sector based on the target species making up the largest proportion by weight of the retained catch per day. This allowed each day of fishing effort to be allocated uniquely to a fishing sector, with the sum of effort in all fishing sectors equalling the total effort of the fishery. The fishing sectors used in the analysis were as per Dredge and Trainor 1994 (Table 3.2) with one modification and one exception. The School Prawn sector is sporadic between years and fewer than 400 boat days per year could be allocated to this sector in 1991 to 1996. Effort for this sector was incorporated into the Eastern King Prawn sector as it occurs in the same location. The Stout Whiting sector was not considered in the analyses because there were only five endorsed boats during the study, the number of days fished was relatively small (i.e., <1,000 days) and no sea turtle by-catch data were available for this sector (Williams 1997; Robins and Courtney 1999). As such, seven sectors were used to estimate sea turtle by-catch within the Queensland East Coast Trawl Fishery.

3.3.3 Estimation procedures

The main objective was to estimate the average annual sea turtle by-catch and associated 95% confidence interval[5]. Annual fleet effort, whilst being a known quantity, was treated as a random variable for the purposes of inferring future annual sea turtle by-catch. Annual sea turtle by-catch was estimated as the product of the two available variables, namely sea turtle catch per unit effort (sea turtle CPUE) and total fleet effort (in days fished). The product of two independent unbiased parameters gives an unbiased estimator of the total (Pollock *et al.* 1994).

Allocated daily effort per fishing sector for sample fleet sea turtle captures were summed into monthly values and used to calculate sea turtle CPUE over the six years 1991 to 1996. Monthly values were used in preference to individual daily records to minimise variability and reduce the dataset to a size amenable for analysis. Thus, sea turtle CPUE was stratified into seven fishing sectors by six years by twelve months within years. Each of these combinations is referred to as a stratum. Sea turtle CPUE tended to be skewed because of the presence of numerous true zero captures, with the degree of skewness varying between fishing sector. An analysis of variance (ANOVA) of sea turtle CPUE, weighted by sample fleet days fished, was used to determine the relative importance of each of the main strata. Total fleet effort was distributed approximately normally. The stratum main effects for this variable were determined by unweighted and untransformed parametric ANOVA.

These preliminary analyses demonstrated large differences and heterogeneous variances between fishing sectors for sea turtle CPUE and total fleet effort. For sea turtle CPUE, year and month effects were not large and were interpreted as indicative of random

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[5] Estimates of sea turtle by-catch were undertaken in collaboration with Dr David Mayer, Principal Biometrician, QDPI.

variation, giving 72 independent observations of sea turtle CPUE for each fishing sector. For total fleet effort, year and month effects were significant ($p<0.001$). The month effect within each fishing sector was reduced to a single degree-of-freedom contrast between 'high season' and 'low season'. Fishing seasons were derived from months in which the majority of the target species was caught for a fishing sector. Hence, the strata for estimating the total fleet effort of the Queensland East Coast Trawl Fishery consisted of seven fishing sectors by six years by two seasons within years (i.e., 84 strata), with six random observations within each stratum.

Bootstrapping (Efron and Tibshirani 1993) was considered to be an appropriate approach for estimating the mean annual sea turtle by-catch and associated confidence intervals because the sea turtle CPUE was highly skewed and contained a reasonable number of true zero CPUEs. For each replicated bootstrap, the total sea turtle captures for each of the 84 strata (i.e., fishing sector by year by season) were estimated by multiplying mean sea turtle CPUE by mean total fleet effort, with the number of resamplings (with replacement) for each being the number of observations available (Efron and Tibshirani 1993) i.e., six for total fleet effort and 72 for sea turtle CPUE. Bootstrap resamplings from the sea turtle CPUE were weighted according to the sampling fleet effort of each observation. DiCiccio and Efron (1996) recommend the use of 2,000 or more bootstrap replicates for the more difficult estimation of confidence intervals. The mean catch and associated distribution per strata were estimated from 5,000 replicates because of the variability in the data. Total annual sea turtle by-catch was estimated from the generated distributions by summing the 5,000 bootstrap estimates from each stratum. Non-parametric confidence intervals from these ordered replicates were estimated using the standard percentile method. This is a symptotically valid method of estimating the confidence limits of skewed data (Young 1994).

As confirmation of the bootstrap methodology, the weighted means and standard errors (using pooled variation from analyses within each fishing sector) were used to calculate parametric estimates of total sea turtle by-catch. Associated confidence limits about these estimates were calculated via the methods of Buonaccorsi and Liebhold (1988) and Poiner and Harris (1996). Independence between these means was assumed. The bootstrap means were virtually the same as the means from the weighted untransformed parametric analysis, indicating the overall estimates of sea turtle by-catch were quite stable. However, the confidence limits were notably different, as also found by Buonaccorsi and Liebhold (1988). The bootstrap 95% confidence intervals were tighter and non-symmetrical, supporting the choice of bootstrap analyses of the highly skewed sea turtle CPUE data (Appendix A).

3.3.4 Sea turtle by-catch mortality

Observed mortality rates based on reported condition upon capture

Studies of sea turtle by-catch in other prawn trawl fisheries have estimated the number of sea turtles killed as a result of a capture from observed dead sea turtles (Poiner *et al.* 1990). This has been criticised as a minimum estimate of trawl-related mortality because comatose sea turtles are not included (Murphy and Hopkins-Murphy 1989). Kemmerer (1989) suggested that comatose sea turtles returned to the water after a trawl capture probably die and should be included in mortality estimates. Therefore, I calculated two estimates of observed mortality rates: (i) direct mortality = dead sea turtles/total sea turtles captured, representing a minimum estimate of trawl by-catch mortality based on reported dead sea turtles; and (ii) potential mortality = (dead sea turtles + comatose sea turtles)/total sea turtles captured, representing an estimate of trawl by-catch mortality assuming all comatose sea turtles die.

Establishing a relationship between sea turtle by-catch mortality rates and tow duration

Tow duration can be used as an alternative method of estimating sea turtle by-catch mortality (Henwood and Stuntz 1987). Watson and Seidel (1980) and Henwood and Stuntz (1987) used simple linear regressions to describe the relationship between tow duration and mortality of trawl-caught sea turtles. However, a standard linear regression did not adequately represent the relationship between tow duration and observed direct mortality for tows less than 60 minutes where mortality rates were negligible (Henwood and Stuntz 1987). Mean tow duration in the Banana Prawn and Moreton Bay sectors of the Queensland East Coast Trawl Fishery are around 70 to 80 minutes (Dredge and Trainor 1994), which is near the limit of the applicability of the relationship described by Henwood and Stuntz (1987). Adequately describing the relationship between tow duration and mortality for short tows is important because a significant proportion of sea turtle by-catch was likely to have occurred in fishing sectors of the Queensland East

Coast Trawl Fishery where tow durations are short (i.e., <75 minutes, Dredge and Trainor 1994).

As an alternative to a standard linear regression, the relationship between tow duration and mortality was analysed using a conditional weighted bent-stick linear regression for direct mortality and potential mortality. This analysis is similar to the linear regression applied to trawl-mortality by Watson and Seidel (1980) and Henwood and Stuntz (1987), but a threshold must be reached before the linear regression is valid. Analyses were conducted using GENSTAT™ (2000). Sufficient data were available to analyse the relationship for all species pooled and for individual species except *E. imbricata*. Data were grouped into 15-minute tow duration intervals (Kemmerer 1989). Data for tows longer than 240 minutes (i.e., six hours) were pooled because of the limited number of tows of longer than six hours. The significance of the bent-stick relationship was tested using sums of squares corrected for the mean.

Expected mortality rates based on tow duration

Expected mortality rates were estimated using a variety of data sources to explore the possible scale of annual sea turtle by-catch mortality for each fishing sector of the Queensland East Coast Trawl Fishery. Expected sea turtle mortality was derived from mean tow duration per fishing sector and: (i) the bent-stick relationship calculated from the Queensland east coast data; and (ii) the significant linear relationship calculated by Henwood and Stuntz (1987) i.e., $Y=0.00165X-0.03$, where Y is mortality rate and X is mean tow duration. This offered a comparison of various scenarios of expected sea turtle mortality in the Queensland East Coast Trawl Fishery.

3.3.5 Verification of the data

An important aspect of any sampling program is to ensure that the sample is representative of the total population. In this case, the sample is the fishers in the sea turtle by-catch monitoring program and the total population is the total fleet of Queensland East Coast Trawl Fishery. Previous sea turtle by-catch studies offer little guidance in the verification of these types of datasets (Henwood and Stuntz 1987; Poiner *et al.* 1990; Poiner and Harris 1996). Some studies have compared limited aspects of their sample fleet data with total fleet data. Henwood and Stuntz (1987) compared average depth and tow duration between fleets but found no probable difference.

The spatial and temporal distribution of sample fleet effort was compared to total fleet effort between fishing sectors over months and years using an analysis of variance (ANOVA), which showed a reasonably constant sampling fraction across all strata. Stratifying for fishing sector encompassed differences in the distribution of effort between sample and total fleets. In addition, the sample fleet was compared to total fleet for: (i) effort within fishing sector; (ii) catch weight by target species e.g., king prawns, tiger prawns, scallops; (iii) mobility via an index representing the extent that individual vessels travelled throughout the fishery based on the number of CFISH grids fished on average per year; and (iv) boat length and hull units, which are measures of the operational capacity of a vessel (i.e., its ability to fish).

3.3.6 Assumptions and inherent difficulties of these methods

The Queensland East Coast Trawl Fishery occurs over a vast geographic area (about 226,900 km^2) and fishing occurs in all seasons of the year. Low frequency of sea turtle by-catch and ethical considerations limited my research of sea turtle by-catch to an observational study. Monitoring fisheries through logbook reporting is a standard method for collecting catch and effort information in most Australian fisheries (Hilborn and Walters 1992; Kailola *et al.* 1993). Observers were not used to collect sea turtle by-catch data because of limited funds and the difficulty of achieving representative observer coverage in a fishery where >85,000 nights were fished per year by about 1,000 vessels over an area of about 226,900 km^2, much of which is remote from human settlement. A fishery-dependent logbook program using selected voluntary fishers was the most feasible method, in terms of cost and coverage, to obtain information on sea turtle by-catch in this fishery. The sea turtle by-catch and mortality estimates assume that: (i) all sea turtles caught by the sample fleet were reported accurately; and (ii) all fishing effort was reported accurately.

An inherent criticism of fishery-dependent sampling is the possibility of bias from small or unrepresentative sampling and if based on logbooks, inaccurate reporting by the fishers involved (Murphy and Hopkins-Murphy 1989). A strength of the sea turtle by-catch monitoring program was the participation of numerous individuals (~100). The number and diversity of participants suggest that it would take a concerted effort by the majority of fishers to mis-report sea turtle captures in order to significantly affect the accuracy of the data and subsequent estimates. However, if fishers did inaccurately report details of sea turtle by-catch, then estimates presented here are likely to be a minimum estimate of the annual by-catch and mortality of sea turtles in the Queensland East Coast Trawl Fishery. Sea turtle CPUE has been recorded during research trawling by the Queensland Department of Primary Industries (QDPI) for benthic community surveys, prawn tagging, TEDs trials and research observations of commercial trawling. These offer an independent source of sea turtle CPUE, although the information is limited in time and space. Mean research sea turtle CPUE, weighted by the number of days fished, was calculated where data were available and compared against reported sea turtle CPUE for each fishing sector of the Queensland East Coast Trawl Fishery.

3.4 RESULTS

Sea turtle by-catch data were obtained from 96 vessels, representing the involvement of about 10% of the Queensland trawl fleet. About 1,500 sea turtles were reported caught during ~24,000 days of fishing by vessels in the sample fleet from 1991 to 1996. Catch and effort for the sample fleet covered 122 CFISH grids, while the total fleet covered 226 CFISH grids. Nine fishers returned sea turtle by-catch data for six years, two fishers participated for five years, six fishers participated for four years, 14 for three years, 23 for two years and 42 for one year.

3.4.1 Comparison of fleet characteristics

Overall, the sample fleet displayed similar distributions (i.e., number of vessels per category) in fleet characteristics to the total fleet (Table 3.3). There were no significant differences between the distributions of the sample and total fleet for annual effort per vessel (χ^2=8.68, d.f.=4, p=0.070) and mobility (χ^2=6.18, d.f.=5, p=0.289).

Table 3.3 Mean annual effort per vessel from 1991 to 1996

Annual effort per vessel			Mobility		
Days fished per year	Sample fleet[A]	Total fleet[B]	No of grids fished	Sample fleet[A]	Total fleet[B]
> 200	3.1%	5.3%	25-30 (highly mobile)	1.1%	0.2%
150 to 200	12.5%	20.2%	20-25 (highly mobile)	4.2%	2.3%
100 to 150	22.9%	28.1%	15-20	5.3%	6.7%
50 to 100	32.2%	21.5%	10-15	16.8%	19.8%
0 to 50	29.2%	24.9%	5-10	37.9%	29.3%
			1-5 (localised)	34.7%	41.7%

[A] Sample fleet = 96 vessels; [B] Total fleet = 985 vessels.

The mean annual catch of the sample fleet (as a proportion of the total fleet catch) varied between sectors (Table 3.4), being highest in the Banana Prawn sector and lowest in the Moreton Bay sector. The mean annual effort of the sample fleet (as a proportion of the total fleet effort) was also highest in the Banana Prawn sector and lowest in the Moreton Bay sector (Table 3.4). The mean annual catch and effort of the sample fleet reflects the overall sampling fraction obtained. Sample fleet catch and effort per fishing sector fluctuated between years as a consequence of fishers joining and leaving the sea turtle by-catch monitoring program.

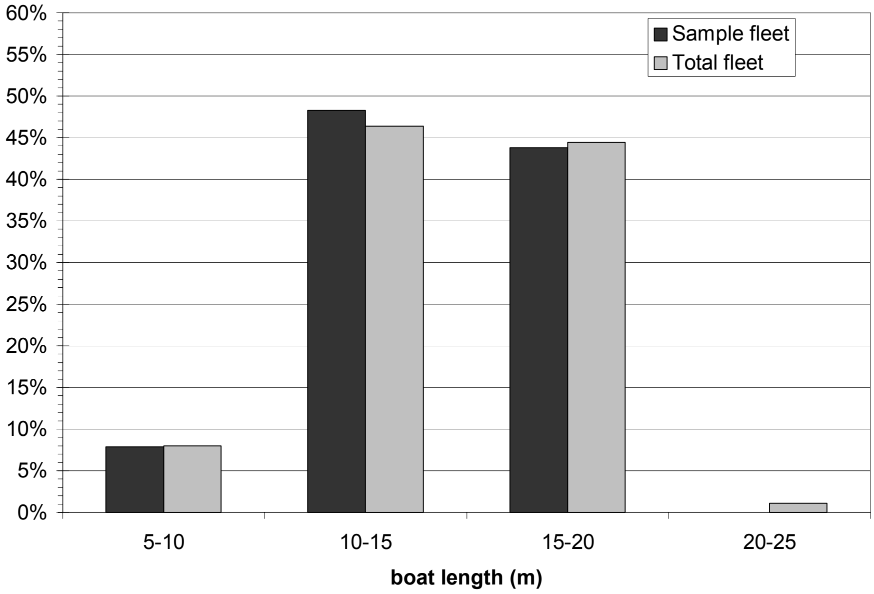
Table 3.4 Mean annual commercial catch and effort of the sample fleet as a percentage of the total fleet

Fishing sector	Commercial catch	Effort
Tiger Prawn	4.7%	4.6%
Endeavour Prawn	7.9%	7.4%
Red spot king Prawn	5.6%	4.4%
Eastern king Prawn	2.9%	3.7%
Moreton Bay	2.4%	2.2%
Banana Prawn	10.8%	8.7%
Scallop	3.9%	6.0%
All sectors pooled	5.0%	4.7%

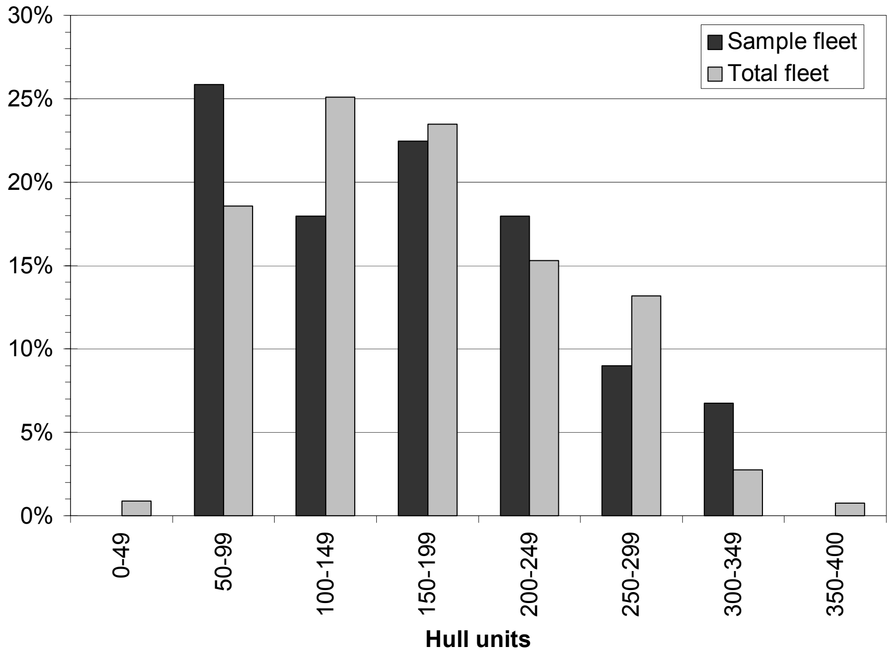
The vessels in the sample fleet were of similar size (i.e., boat length) to those in the total fleet except that vessels >20m in length were not represented in the sample fleet (Figure 3.2a). These vessels comprised 1.1% of the total fleet and there was no significant difference between the distribution of lengths of vessels in the sample fleet and total fleet (χ^2=1.08, d.f.=3, p=0.782). The sample fleet encompassed the range of hull units in the total fleet except for vessels with <50 or >350 hull units (Figure 3.2b). There was no significant difference between the distribution of hull units of vessels in the sample fleet and total fleet (χ^2=10.82, d.f.=7, p=0.146).

Figure 3.2 Comparison of the operational characteristics of the sample and total fleet

a) Boat length



b) Hull units



3.4.2 Sea turtle CPUE per fishing sector

Sea turtle CPUE was not consistent across fishing sectors (Table 3.5). The highest sea turtle CPUE occurred in Moreton Bay, where catch rates of *C. caretta* were 0.203 sea turtles per day fished and those of *C. mydas* were 0.055 sea turtles per day fished. Other fisheries with high sea turtle CPUE were the Tiger and Endeavour Prawn sectors, which often overlap in time and space, and the Banana Prawn sector. The research sea turtle

CPUE was in the same order-of-magnitude as the reported sea turtle CPUE for the Tiger Prawn and Banana Prawn sectors (Table 3.5), being ~1.3 and ~1.1 times the reported sea turtle CPUE respectively. No sea turtles were caught during research trawling in the Scallop and Eastern King Prawn sectors. This was lower than the fishery-dependent sea turtle CPUE and is consistent with anecdotal reports of infrequent sea turtle captures in these sectors. Research sea turtle CPUE was an order-of-magnitude lower than the reported sea turtle CPUE for the Moreton Bay sector i.e., ~0.3 times the reported sea turtle CPUE. Research catch rates of sea turtles were likely to be influenced by the limited spatial and temporal sampling regimes in which research trawls were conducted compared with the areas in which commercial trawls occurred.

Table 3.5 Sea turtle CPUE derived from commercial and research trawls

	Sea turtle CPUE (sea turtles caught per day fished)						
	Reported from commercial trawls (=fishery-dependent)						Research
Fishing sector	*N. depressus*	*C. caretta*	*L. olivacea*	*C. mydas*	*E. imbricata*	All species	All species[A]
Tiger Prawn	0.0240	0.0060	0.0090	0.0230	0.0020	0.0645	0.0854 (82)
Endeavour Prawn	0.0260	0.0070	0.0050	0.0130	0.0008	0.0498	-
Red Spot King Prawn	0.0120	0.0050	0.0030	0.0050	0.0006	0.0213	-
Eastern King Prawn	0.0020	0.0090	0.0010	0.0070	0.0003	0.0155	0.0000 (137)
Moreton Bay	0.0020	0.2030	0.0020	0.0550	0.0016	0.2754	0.0733 (150)
Banana Prawn	0.0110	0.0260	0.0030	0.0280	0.0005	0.0682	0.0714 (84)
Scallop	0.0040	0.0060	0.0010	0.0040	0.0000	0.0159	0.0000 (213)

[A] (n) = The total number of days fished from which the weighted research sea turtle CPUE was derived.

3.4.3 Estimated sea turtle catch

Overall

Between ~5,200 and ~6,600 sea turtles were estimated to be caught annually in the Queensland East Coast Trawl Fishery, being comprised of ~50% *C. caretta*, ~27% *C. mydas*, ~16% *N. depressus*, ~6% *L. olivacea* and ~1% *E. imbricata* (Table 3.6).

Table 3.6 Mean annual sea turtle by-catch in the Queensland East Coast Trawl Fishery

	Mean	Percent composition	95% Confidence Interval
N. depressus	968	16.4%	770 to 1,165
C. caretta	2,938	49.8%	2,390 to 3,487
L. olivacea	323	5.5%	240 to 406
C. mydas	1,562	26.5%	1,223 to 1,902
E. imbricata	80	1.4%	42 to 119
All species[A]	5,901	100.0%	5,199 to 6,604

[A] Includes sea turtles not identified to species.

By fishing sector

Estimated sea turtle captures were not evenly distributed across fishing sectors (Table 3.7). In particular, Moreton Bay accounted for 54% of estimated sea turtle captures, the Tiger Prawn sector accounted for 23% of estimated captures and the Banana Prawn sector accounted for 6% of estimated captures. These fishing sectors were associated with inshore waters and occur close to the Queensland coastline. The other four sectors of the Fishery each caught less than 5% of total estimated captures. Most of these sectors (i.e., Eastern King Prawn, Scallop and Red Spot King Prawn) were associated with waters that are offshore or deep.

Table 3.7 Mean annual sea turtle by-catch by species and fishing sector

Sector	N. depressus	*C. caretta*	*L. olivacea*	*C. mydas*	*E. imbricata*	All species[A]
Tiger Prawn	502	126	188	481	42	1,350
Endeavour Prawn	149	40	29	75	5	286
Red Spot King Prawn	155	65	39	65	8	276
Eastern King Prawn	32	143	16	111	5	246
Moreton Bay	23	2,358	23	639	19	3,199
Banana Prawn	55	130	15	140	3	342
Scallop	51	76	13	51	0	203

[A] Includes sea turtles not identified to species.

3.4.4 Size class of reported sea turtle by-catch

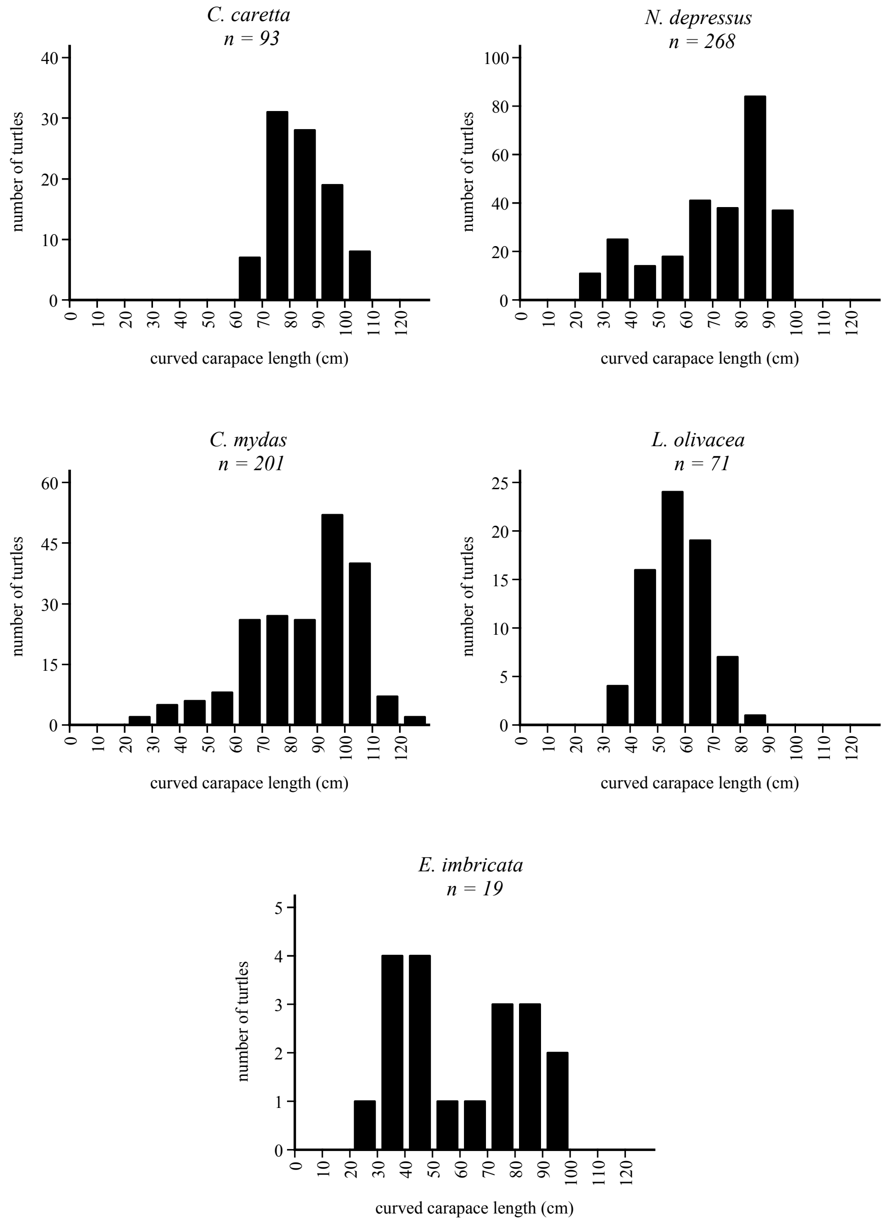
A diverse size range of sea turtles was reported caught in the Queensland East Coast Trawl Fishery (Figure 3.3). More than 60% of sea turtles caught were likely to be immature based on approximate size at maturity (Table 3.8).

Table 3.8 Proportion of immature and mature sea turtles

	CCL limits		Approx. size	Sample	Percent	Percent
	Lower	Upper	at maturity	size*	immature	adult
N. depressus	5cm	100cm	>90cm [A]	268	80.1%	19.9%
C. caretta	70cm	110cm	>90cm [B]	93	69.9%	30.1%
L. olivacea	40cm	85cm	>65cm [C]	71	77.8%	22.2%
C. mydas	35cm	130cm	>95cm [D]	201	61.7%	38.3%
E. imbricata	35cm	100cm	>75cm [E]	19	63.2%	36.8%

[A] Limpus *et al*. 1983b; [B] Limpus *et al*. 1994a; [C] Harris 1994; [D] Limpus *et al*. 1994b; [E] Limpus 1992;
* Includes sea turtles caught where size information was reported.

Figure 3.3 Size distribution of reported sea turtle by-catch in the Queensland East Coast Trawl Fishery



Trawl-caught *C. caretta* ranged in size from 65 to 110 cm CCL (Figure 3.3) About 70% of individuals were ≤90 cm CCL and were probably immature(Table 3.8). This excludes 39 *C. caretta* caught mostly in Moreton Bay that were treated as unidentified and of uncertain size because their reported CCL was <65 cm. Reported captures of *C. mydas* ranged in size from 27 to 124 cm CCL, with most individuals being of a medium to large size (Figure 3.3). About 62% of trawl-caught *C. mydas* were likely to be immature

(Table 3.8). *E. imbricata* caught in trawl nets ranged in size from 28 to 92 cm CCL (Figure 3.3). The minimum size of recruitment of *E. imbricata* to coral reefs is estimated at 35 cm CCL (Limpus 1992, Chaloupka and Limpus 1997). However, a 28 cm CCL *E. imbricata* was reported caught by a fisher who was competent in species identification as confirmed by photographic validation. The sample size of trawl-caught *E. imbricata* with size information was small (n = 19), but suggested that ~63% of trawl-caught *E. imbricata* were probably immature (Table 3.8). Trawl-caught *N. depressus* ranged in size from 22 to 100 cm CCL (Figure 3.3). The 80 to 90 cm size class dominated reported catches. About 80% of *N. depressus* caught by the Queensland East Coast Trawl Fishery were probably immature (Table 3.8). Five sea turtles reported as *L. olivacea* had CCLs greater than 85 cm and were treated as mis-identified. The remaining *L. olivacea* ranged from 37 to 83 cm CCL (Figure 3.3), with ~78% being immature (Table 3.8).

3.4.5 Observed mortality rates

Observed mortality rates were based on the reported physical condition upon capture (Table 3.9). The categories ‘healthy’ and ‘externally injured’ were pooled because in all cases of sea turtles reported externally injured, the external injuries described were not the result of immediate trawl capture, but were scars or damage from previous events. The condition of two *C. caretta* captured was not specified, so as a conservative approach, these individuals were assumed dead. In addition, seven trawl-caught sea turtles were reported in various stages of decomposition and were assumed to have died prior to capture. Epperly *et al.* (1995a) also reported the capture of sea turtle carcasses that were dead prior to capture in the summer flounder trawl fishery off North Carolina. Like Epperly (*et al*. 1995a), rotting carcasses were not included in calculations of sea turtle CPUE, estimated captures or mortality rates.

Observed direct mortality: overall

Observed direct mortality rates were calculated as the number of dead sea turtles/total sea turtles captured (see section 3.3.4). Pooled across all species and all fishing sectors, more than 94% of sea turtles caught were reported as healthy when first landed on the vessel. About 4% were reported as comatose and about 1% were reported dead (Table 3.9).

Table 3.9 Reported condition upon capture and observed mortality rates of sea turtle by-catch

Stratification		**Healthy %**	**Comatose %**	**Dead %** (Direct mortality)	**Dead+Comatose %** (Potential mortality)
Overall [A] (n = 1,500*)		94.3	4.4	1.3	5.7
By species i.e., pooled across sectors					
N. depressus	(n = 314*)	95.9	2.2	1.9	4.1
C. caretta	(n = 573*)	94.0	4.4	1.6	6.0
L. olivacea	(n = 78*)	85.9	11.5	2.6	14.1
C. mydas	(n = 429*)	93.9	5.4	0.7	6.1
E. imbricata	(n = 23*)	91.3	8.7	0.0	8.7
By species and fishing sector					
N. depressus	Tiger Prawn	93.8	3.4	2.7	6.1
	Endeavour Prawn	97.1	1.4	1.4	2.8
	Red Spot King Prawn	100.0	0.0	0.0	0.0
	Eastern King Prawn [B]	100.0	0.0	0.0	0.0
	Moreton Bay [C]	83.3	16.7	0.0	16.7
	Banana Prawn	96.8	0.0	3.2	3.2
	Scallop	100.0	0.0	0.0	0.0
C. caretta	Tiger Prawn	77.5	10.0	12.5	22.5
	Endeavour Prawn	94.7	5.3	0.0	5.3
	Red Spot King Prawn	92.9	0.0	7.1	7.1
	Eastern King Prawn	95.2	0.0	4.8	4.8
	Moreton Bay	96.7	3.3	0.0	3.3
	Banana Prawn	90.8	6.6	2.6	9.2
	Scallop [C]	77.8	22.2	0.0	22.2
L. olivacea	Tiger Prawn	82.9	14.6	2.4	17.0
	Endeavour Prawn	88.2	11.8	0.0	11.8
	Red Spot King Prawn [C]	100.0	0.0	0.0	0.0
	Eastern King Prawn [B]	33.3	33.3	33.3	66.6
	Moreton Bay [B]	100.0	0.0	0.0	0.0
	Banana Prawn [C]	100.0	0.0	0.0	0.0
	Scallop [B]	100.0	0.0	0.0	0.0
C. mydas	Tiger Prawn	91.0	7.2	1.8	2.0
	Endeavour Prawn	90.6	7.5	1.9	9.4
	Red Spot King Prawn	90.9	0.0	0.0	0.0
	Eastern King Prawn	92.3	7.7	0.0	7.7
	Moreton Bay	93.6	6.4	0.0	6.4
	Banana Prawn	99.1	0.9	0.0	0.9
	Scallop	100.0	0.0	0.0	0.0
E. imbricata	Tiger Prawn [C]	87.5	12.5	0.0	12.5
	Endeavour Prawn [B]	100.0	0.0	0.0	0.0
	Red Spot King Prawn [B]	100.0	0.0	0.0	0.0
	Eastern King Prawn [B]	0.0	100.0	0.0	100.0
	Moreton Bay [B]	100.0	0.0	0.0	0.0
	Banana Prawn [B]	100.0	0.0	0.0	0.0
	Scallop [B]	100.0	0.0	0.0	0.0

[A] Includes sea turtles not identified to species; [B] <5 individuals caught; [C] <10 individuals caught; * n = sample size and includes sea turtle captures where condition upon capture was reported.

Observed direct mortality: by species

Pooled across all fishing sectors but separated into species, the percent of sea turtles reported as healthy when first landed on the vessel was >91% for all species except for *L. olivacea* where 85.9% of individuals caught were reported as healthy, 11.5% were reported as comatose and 2.6% were reported dead (Table 3.9). No direct mortality was observed for *E. imbricata* possibly as consequence of the small sample size (n = 23). However, it is unlikely that *E. imbricata* were not drowned in the Queensland East Coast Trawl Fishery. Therefore, caution is required in extrapolating the direct mortality rates, particularly for species with a small sample size.

Observed direct mortality: by species per fishing sector

When stratified by species and by fishing sector, the highest proportion of sea turtles reported dead (i.e., observed direct mortality) occurred in the Eastern King Prawn sector, where 33.3% of *L. olivacea* were dead when landed on the vessel (Table 3.9). However, this estimate was calculated from three captures of *L. olivacea* in this fishing sector (with one being dead) and should be viewed with caution. High rates of observed direct mortality occurred in the Tiger Prawn sector, where 12.5% of *C. caretta* caught in the sector (n = 40) were dead when landed on the vessel.

Observed potential mortality

Observed potential mortality rates were calculated as the number of [dead and comatose] sea turtles/total sea turtles captured (see section 3.3.4). Overall, observed potential mortality was 5.7% (Table 3.9). Pooled across fishing sector, but stratified by species, observed potential mortality was 4.1% for *N. depressus*, 6.0% for *C. caretta*, 14.1% for *L. olivacea* and 6.1% for *C. mydas.* Observed potential mortality varied amongst species and fishing sector (Table 3.9). Mortality rates derived from small sample sizes should be viewed with caution (Poiner and Harris 1996) as they are likely to mis-represent mortality rates.

Observed mortality by species by maturity status

The physical condition upon capture for each species was tabulated separately for immature and adult size classes to explore possible differences in mortality rates between small and large sea turtles (Table 3.10). Ideally, these data should also be stratified by fishing sector to take into account operating characteristics of the seven fishing sectors. However, the sample size of sea turtle by-catch data that included species, condition-upon-capture and size information was too small to be further stratified by fishing sector.

The percentage of immature and adult sea turtles that were comatose or dead upon capture varied between species (Table 3.10). Small sea turtles are speculated to have higher metabolic rates than larger sea turtles and should be more susceptible to drowning in trawl nets (Lutcavage 1992). Observed mortality rates of immature size classes were higher than those of adult size classes for *N. depressus, L. olivacea* and *C. mydas*, but for *C. caretta* the observed mortality rate of immature size classes was lower than that of adult size classes (Table 3.10). Despite these trends, there were no significant differences between the mortality rates of immature and adult sea turtles based on size class for any species (Table 3.10). Sea turtle by-catch was dominated by immature size classes i.e., >60% for every species. Therefore, mortality rates for all size classes combined, which include all mortality rates in Chapter 3 except those in Table 3.10, are influenced to a greater degree by mortality rates of immature sea turtles than adult sea turtles.

Table 3.10 Observed mortality rates and maturity status of sea turtle by-catch

Species	Maturity status	Sample size*	Healthy	Comatose	Dead (Direct mortality)	Dead+Comatose (Potential mortality)
N. depressus[A]	Immature	(n = 214)	94.4%	2.8%	2.8%	5.6%
	Adult	(n = 53)	98.1%	1.9%	0.0%	1.9%
C. caretta[B]	Immature	(n = 65)	89.2%	6.1%	4.6%	10.7%
	Adult	(n = 28)	82.1%	10.7%	7.1%	17.8%
L. olivacea[C]	Immature	(n = 56)	82.2%	14.3%	3.6%	17.9%
	Adult	(n = 16)	93.8%	6.2%	0.0%	6.2%
C. mydas[D]	Immature	(n = 124)	91.9%	6.5%	1.6%	8.1%
	Adult	(n = 77)	93.5%	6.5%	0.0%	6.5%
E. imbricata[E]	Immature	(n = 12)	88.3%	16.7%	0.0%	16.7%
	Adult	(n = 7)	100.0%	0.0%	0.0	0.0

* Includes reported sea turtle captures with species, condition upon capture and size information recorded; [A] χ^2=1.68, d.f.=2, p=0.431; [B] χ^2=0.89, d.f.=2, p=0.642; [C] χ^2=1.41, d.f.=2, p=0.494; [D] χ^2=1.25, d.f.=2 p=0.534; [E] χ^2=1.2, d.f.=2 p=0.550.

3.4.6 Tow duration versus mortality

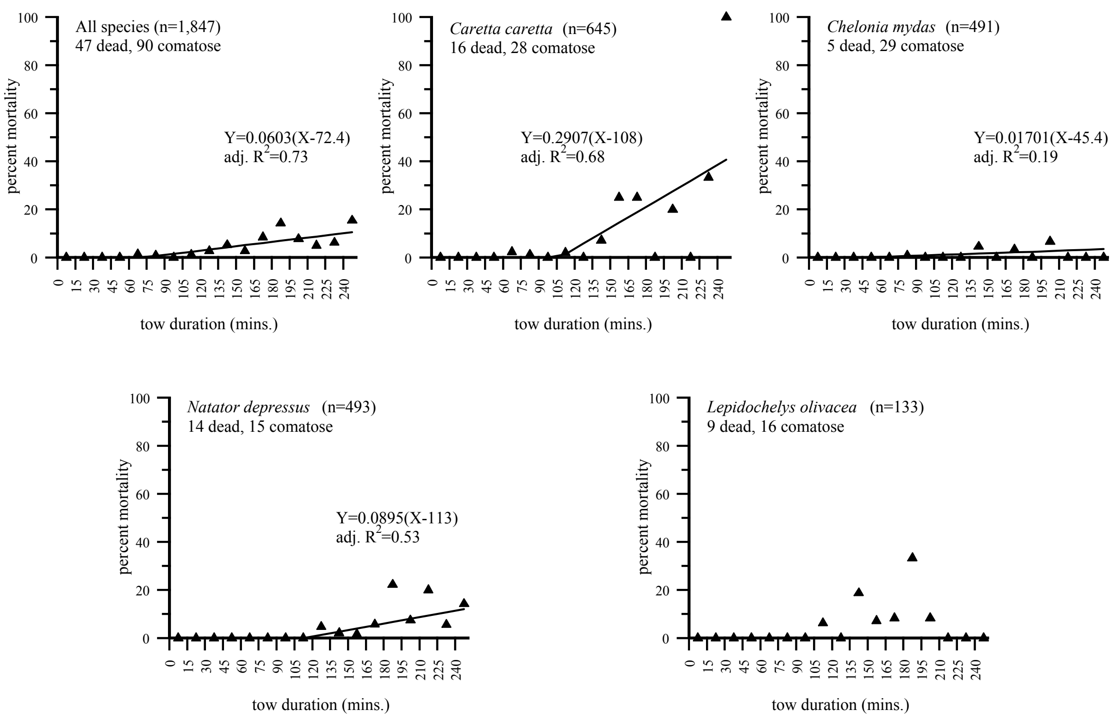
Sea turtles were caught in tows ranging in duration from 10 to 285 minutes, but most captures (~77%) occurred in tows of less than 135 minutes. Only two of the 1,500 sea turtles reported caught lacked information on tow duration. Of the remaining 1,498 trawl-caught sea turtles, 20 were reported as dead and 66 as comatose. The analysis of mortality versus tow duration is important because it has been suggested that some species may have greater tolerance to trawl capture than others (Poiner and Harris 1996). To increase the sample size, data recorded by additional fishers during the sea turtle by-catch monitoring program were incorporated into the tow-time versus mortality analysis. This included data from seven fishers from the Queensland East

Coast Trawl Fishery for 64 sea turtles caught in 1997, 11 fishers from the Torres Strait Prawn Fishery for 170 sea turtles caught between 1993 and 1997 and four fishers from the Northern Prawn Fishery for 106 sea turtles caught between 1995 and 1997. Pooling the data increased the sample size to 1,847 captures with a total of 47 being reported dead and 90 comatose. Data are presented for each species except *E. imbricata*, of which only 36 individuals were reported caught (32 healthy, two comatose and two dead). The resulting relationships between tow-time and mortality should be interpreted with some caution as the sample sizes were still relatively small, although larger than those used by Henwood and Stuntz (1987) and Poiner and Harris (1996).

Expected direct mortality

A conditional weighted bent-stick linear regression of tow-time against direct mortality (i.e., dead only) was statistically significant for all species pooled ($p<0.001$), *C. caretta* ($p<0.001$), *C. mydas* ($p=0.043$) and *N. depressus* ($p<0.001$), but was not significant for *L. olivacea* ($p=0.286$). The regression lines accounted for over half of the variance for all species pooled, *C. caretta* and *N. depressus* (Figure 3.4).

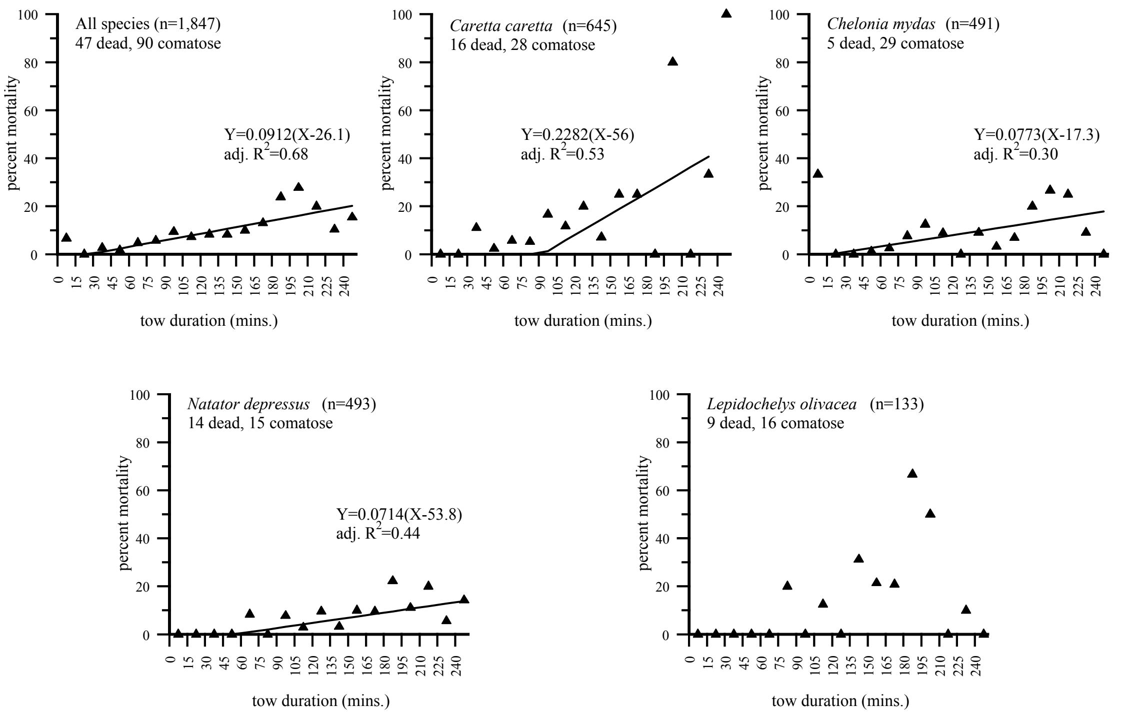
Figure 3.4 Direct mortality of trawl-caught sea turtles as a function of tow duration (Non-significant regression for *L. olivacea* not shown)



Expected potential mortality

A conditional, weighted bent-stick regression of tow-time against potential mortality (i.e., dead plus comatose) was statistically significant for all species pooled (p<0.001), *C. caretta* (p<0.001), *C. mydas* (p<0.013), *N. depressus* (p=0.002) and *L. olivacea* (p=0.045). The fitted regression lines for potential mortality accounted for a greater proportion of the variance than the fitted regression lines for direct mortality for *C. mydas* and *L. olivacea*. Adjusted R^2 values were 68%, 53%, 30% and 44% for all species pooled, *C. caretta*, *C. mydas* and *N. depressus* respectively (Figure 3.5).

Figure 3.5 Potential mortality of trawl-caught sea turtles as a function of tow duration



3.4.7 Estimates of expected mortality based on tow duration

The mean tow duration reported during the sea turtle by-catch monitoring program was similar to the estimates of Dredge and Trainor (1994) for four sectors of the Queensland East Coast Trawl Fishery (Table 3.11). For the Eastern King Prawn, Red Spot King Prawn and Scallop sectors, reported mean tow duration during the present study was shorter than the estimates of Dredge and Trainor (1994). Two estimates of expected mortality were calculated for these sectors (Table 3.11) based on: (i) reported tow duration during the present study; and (ii) the estimates of Dredge and Trainor (1994).

Expected mortality based on the linear relationship reported for USA shrimp fisheries

The overall expected direct mortality of sea turtles in the Queensland East Coast Trawl Fishery, based on the USA relationship, was 14%, being less than 10% in the Moreton Bay and Banana Prawn sectors and greater than 20% in the Tiger Prawn, Endeavour Prawn and Scallop sectors (Table 3.11, USA shrimp fisheries). These rates are higher than that reported by commercial fishers for the Queensland East Coast Trawl Fishery (Table 3.9). This may reflect the bias in the linear relationship between tow duration and mortality derived for USA shrimp fisheries where *C. caretta* comprised >90% of sea turtle by-catch. The response of that species to trawl capture would dominate the linear relationship reported by Henwood and Stuntz (1987). In the current Queensland study, *N. depressus* and *C. mydas* were caught in significant numbers and the ability of these species to tolerate forced submergence would influence observed direct mortality rates. Henwood and Stuntz (1987) commented that sea turtle mortality rates based on tow duration did not conform to the expected linear model for tows less than 60 minutes where mortality rates were <1%. The Moreton Bay and Banana Prawn sectors of the Queensland East Coast Trawl Fishery have mean tow durations close to 60 minutes and the linear relationship identified by Henwood and Stuntz (1987) provides a poor indication of sea turtle by-catch mortality in these sectors. Therefore, expected mortality rates were estimated from significant bent-stick relationships between tow duration and mortality recorded in the present study for all species combined and for *C. caretta*. Expected mortality rates were calculated for *C. caretta* captures for comparison with USA estimates.

Expected mortality based on tow duration relationship for the Queensland east coast

ALL SPECIES POOLED

The expected direct mortality rate of sea turtles was 1.8% for all sectors pooled based on a mean overall tow duration of 103 minutes (Table 3.11). Expected direct mortality was <5% in most sectors and was negligible in the Moreton Bay and Banana Prawn sectors (Table 3.11). If comatose sea turtles were assumed to die, then the expected potential mortality was 6.5% for all sectors pooled, ~4% for Moreton Bay and Banana Prawn sectors and ~10% for Tiger Prawn, Endeavour Prawn and Scallop sectors.

Table 3.11 Expected mortality rates of trawl-caught sea turtles based the relationship between mortality and tow duration

Sector	Tow duration mean ± s.e. (mins)	Expected mortality(%)				
		USA shrimp fisheries	Northern Australia prawn fisheries			
		Direct All species [B] Y=0.165(X–18.18)	Direct *C. caretta* [C] Y=0.2907(X-107.7)	Potential *C. caretta* [C] Y=0.2282(X-56.1)	Direct All species [C] Y=0.0603(X–72.4)	Potential All species [C] Y=0.0912(X-26.1)
Tiger prawn	144 ± 2.7	20.8	10.6	20.1	4.3	10.8
Endeavour prawn	146 ± 3.5	21.1	11.1	20.5	4.4	10.9
Red spot king prawn	111 ± 4.4	15.3	1.0	12.5	2.3	7.7
	128 [A]	18.1	5.9	16.4	3.4	9.3
Eastern king prawn	92 ± 3.9	12.2	0.0	8.2	1.2	6.0
	>120 [A]	16.8	3.6	14.6	2.9	8.6
Moreton Bay	76 ± 0.7	9.5	0.0	4.5	0.2	4.6
Banana prawn	71 ± 2.1	8.7	0.0	3.4	0.0	4.1
Scallop	113 ± 6.1	15.6	1.5	13.0	2.4	7.9
	155 [A]	22.6	13.8	22.6	5.0	11.8
All sectors pooled	103 ± 1.2	14.0	0	10.7	1.8	7.0

[A] Taken from Dredge and Trainor 1994; [B] linear relationship from Henwood and Stuntz (1987) converted into the same form as the Queensland relationship; [C] conditional weighted bent-stick linear relationship (this study, see Chapter 3, section 3.4.6).

C. CARETTA ONLY

The expected direct mortality of *C. caretta* was negligible for all sectors pooled, Moreton Bay and Banana Prawn sectors, but was between 10% and 14% for the Tiger Prawn, Endeavour Prawn and Scallop sectors (Table 3.11). The expected potential mortality of *C. caretta* was ~11% for all sectors pooled, between 3% and 5% for the Moreton Bay and Banana Prawn sectors and >20% for Tiger Prawn, Endeavour Prawn and Scallop sectors. Expected mortality rates derived from the significant bent-stick regressions often changed to a greater degree for slight changes in tow duration than the expected mortalities derived from the linear regression of Henwood and Stuntz (1987). This is a consequence of the threshold point in the bent-stick relationship.

The main purpose of calculating expected mortality rates was to determine a range of possible sea turtle by-catch mortality and apply these to the estimated annual sea turtle by-catch to derive a range of estimates of the number of sea turtles killed as a consequence of incidental trawl capture in the Queensland East Coast Trawl Fishery as outlined below.

3.4.8 Estimated annual kill

Based on observed mortality

The observed direct and potential mortality rates (Table 3.9) were applied to the estimated annual catch (Tables 3.6, 3.7) to calculate the annual kill of sea turtles in the Queensland East Coast Trawl Fishery (Poiner and Harris 1996) prior to the regulation of TEDs.

OBSERVED MORTALITY: ALL SPECIES POOLED

About ~60 sea turtles (95% C.I. 68 to 86) were estimated to drown annually in trawl nets of the Queensland East Coast Trawl Fishery based on observed direct mortality percentages (Table 3.12). If comatose sea turtles were included, then ~320 sea turtles (95% C.I. 296 to 376) were estimated to die annually as a consequence of being caught in a trawl net of the Queensland fleet (Table 3.12). These estimates were within the same order-of-magnitude as the preliminary estimates of Robins (1995).

Table 3.12 Estimated annual kill of sea turtles in the Queensland East Coast Trawl Fishery

Stratification		Number directly killed		Number potentially killed	
		Observed	Expected based on tow duration	Observed	Expected based on tow duration
Overall [A]		61	104	319	407
By species i.e., pooled across sectors					
N. depressus		18	22	41	57
C. caretta		31	36	149	193
L. olivacea		9	11	43	32
C. mydas		10	16	101	109
E. imbricata		0	2	10	8
By species and fishing sector					
N. depressus	Tiger prawn	14	14	31	32
	Endeavour prawn	2	4	4	10
	Red spot king prawn	0	2	0	8
	Eastern king prawn [B]	0	0	0	2
	Moreton Bay [C]	0	0	4	0
	Banana prawn	2	0	2	1
	Scallop	0	2	0	4
C. caretta	Tiger prawn	16	13	28	25
	Endeavour prawn	0	4	2	8
	Red spot king prawn	5	4	5	11
	Eastern king prawn	7	5	7	21
	Moreton Bay	0	0	78	107
	Banana prawn	3	0	12	4
	Scallop [C]	0	10	17	17
L. olivacea	Tiger prawn	5	8	32	20
	Endeavour prawn	0	1	3	3
	Red spot king prawn [C]	0	1	0	4
	Eastern king prawn [B]	4	0	8	1
	Moreton Bay [B]	0	0	0	1
	Banana prawn [C]	0	0	0	1
	Scallop [B]	0	1	0	2
C. mydas	Tiger prawn	9	8	43	47
	Endeavour prawn	1	1	7	7
	Red spot king prawn	0	1	0	6
	Eastern king prawn	0	1	9	9
	Moreton Bay	0	3	41	29
	Banana prawn	0	1	1	6
	Scallop	0	1	0	5
E. imbricata	Tiger prawn [C]	0	2	5	5
	Endeavour prawn [B]	0	0	0	1
	Red spot king prawn [B]	0	0	0	1
	Eastern king prawn [B]	0	0	5	0
	Moreton Bay [B]	0	0	0	1
	Banana prawn [B]	0	0	0	0
	Scallop [B]	0	0	0	0

OBSERVED MORTALITY: BY SPECIES

However, unlike previous estimates (Robins 1995; Robins and Mayer 1998), the estimates presented here include mortality that was also stratified by fishing sector and species (Table 3.9). About 30 *C. caretta,* ~20 *N. depressus*, ~10 *C. mydas* and ~10 *L. olivacea* were estimated to die per year as a consequence of direct mortality in trawl nets, pooled across all sectors of the Queensland East Coast Trawl Fishery (Table 3.12). Considering observed potential mortality (i.e., dead + comatose sea turtles), the estimated annual number of sea turtles which potentially died as a consequence of capture in the Queensland East Coast Trawl Fishery were ~150 *C. caretta*, 40 *N. depressus*, ~100 *C. mydas*, ~40 *L. olivacea* and ~10 *E. imbricata* (Table 3.12).

Based on expected mortality rates derived from mean tow duration

The expected mortality rates (Table 3.11) were applied to the estimated annual sea turtle catch (Tables 3.6, 3.7) to calculate the annual kill of sea turtles in the Queensland East Coast Trawl Fishery prior to the regulation of TEDs as expected from the mean tow duration in sectors of the fishery (Henwood and Stuntz 1987).

EXPECTED MORTALITY: ALL SPECIES POOLED

If expected mortality rates were based on the USA shrimp fishery relationship between tow duration and sea turtle by-catch mortality (Henwood and Stuntz 1987) and are applied to each sector, then ~830 sea turtles (95% C.I. 728 to 924) were estimated to drown annually in the Queensland East Coast Trawl Fishery. This was estimated to have been comprised of ~320 *C. caretta*, ~190 *N. depressus*, ~230 *C. mydas*, ~60 *L. olivacea*, and ~10 *E. imbricata*.

However, if the expected mortality rates were based on the significant relationship between tow duration and sea turtle by-catch mortality for the Queensland East Coast Trawl Fishery, then ~100 sea turtles (95% C.I. 96 and 122) were estimated to drown annually (Table 3.12). If comatose sea turtles were assumed to die, then ~410 sea turtles (95% C.I. 365 to 463) were estimated to die annually as a consequence of being caught in the trawl nets of the Queensland fleet (Table 3.12).

EXPECTED MORTALITY: BY SPECIES

Based on mean tow duration in each sector and the significant bent-stick relationship between tow duration and mortality rates for each species (see section 3.4.6), the estimated annual kill of sea turtles in the Queensland East Coast Trawl Fishery was ~40 *C. caretta*, ~20 *N. depressus*, ~20 *C. mydas*, ~10 *L. olivacea* and <5 *E. imbricata* (Table 3.12). If potential mortality rates were used (i.e., comatose sea turtles were assumed to die), then ~190 *C. caretta*, ~60 *N. depressus*, ~110 *C. mydas, ~30 L. olivacea* and ~10

E. imbricata were estimated to die annually as a consequence of trawl captures in the Queensland East Coast Trawl Fishery (Table 3.12). The data underlying the estimates of catch and mortality were not of sufficient accuracy for the presented values to be considered as point estimates, but rather they should be considered as order-of-magnitude estimates. The numbers presented above were based on average annual catch and effort of the Queensland East Coast Trawl Fishery from 1991 to 1996 and represent mean annual estimates of sea turtle by-catch and mortality in this fishery.

3.5 DISCUSSION

3.5.1 Estimated sea turtle by-catch

Sea turtle CPUE

Sea turtle catch rates were highest in fishing sectors that occur in shallow, inshore waters (i.e., Moreton Bay, Tiger Prawn and Banana Prawn) and were lowest in fishing sectors that occur in deep or offshore waters (i.e., Eastern King Prawn, Red Spot King Prawn and Scallop). This confirms speculation on the likely frequency of encounter of sea turtles in sectors of the Queensland East Coast Trawl Fishery (Dredge and Trainor 1994) and concurs with anecdotal reports from commercial trawl fishers. Similar trends in sea turtle by-catch have been reported in other penaeid trawl fisheries, where most sea turtle captures occur in waters less than 30m deep (Henwood and Stuntz 1987; Poiner and Harris 1996).

Species composition

The main species caught were *C. caretta*, *C. mydas* and *N. depressus*. *C. caretta* dominated sea turtle captures of sectors operating in sub-tropical waters and *N. depressus* dominated sea turtle captures of sectors operating in tropical waters. Captures of *C. mydas* were distributed throughout in tropical and sub-tropical waters. Similar trends in species composition have been reported for other trawl fisheries. The main species caught in trawl fisheries were carnivorous sea turtles i.e., *C. caretta* and *Lepidochelys kempii* in the USA trawl fisheries (Henwood and Stuntz 1987) and *N. depressus* and *L. olivacea* in the tropical Northern Prawn Fishery (Poiner and Harris 1996). Carnivorous sea turtle species favour benthic prey associated with soft-bottom habitats that also are typical of penaeid trawl grounds (Marquez 1990; Limpus *et al.*

2001). The low capture rate of *L. olivacea* in the Queensland East Coast Trawl Fishery probably reflects the relative low density of this species in waters of the Queensland east coast.

An unusual feature of sea turtle by-catch in the Queensland East Coast Trawl Fishery was the significant catch of the herbivorous *C. mydas*, which accounted for ~27% of sea turtles caught. *C. mydas* was also a significant by-catch in the Torres Strait Prawn Fishery (~22%, Robins and Mayer 1998) and the Northern Prawn Fishery (~8%, Poiner and Harris 1996). In contrast, *C. mydas* comprised less than 4% of sea turtle by-catch in the southeastern USA (Henwood and Stuntz 1987). The significant trawl capture of *C. mydas* in eastern Australia is probably the consequence of numerous large seagrass beds that provide feeding-ground habitat for *C. mydas*, as well as nursery habitat for tiger prawns (i.e., *Penaeus esculentus* and *P. semisulcatus*). In Australia, trawling for tiger prawns occurs in areas adjacent to seagrass beds, and exposed *C. mydas* to potential capture in these fisheries, prior to the regulation of TEDs. *E. imbricata* is known to be present throughout the Great Barrier Reef (Limpus 1992), but few *E. imbricata* were caught and killed by the Queensland fleet. This is likely to be a consequence of the preference of *E. imbricata* for reef habitats that are not suitable for trawling for penaeid prawns.

Size composition

The size-based maturity classification was used to assess the probable life stage of sea turtle by-catch in the Queensland East Coast Trawl Fishery. The allocation of maturity status based on size should be considered as approximate because the size of a sea turtle does not consistently reflect its age or maturation stage (Miller 1996; Musick and Limpus 1996). However, information on the likely proportion of immature and adult sea turtles impacted by trawl by-catch can assist in understanding the demographic consequences of by-catch mortality. The sizes of sea turtle caught by the Queensland East Coast Trawl Fishery suggested that trawling most significantly impacted immature sea turtles.

C. CARETTA

About 70% of trawl-caught *C. caretta* were immature, based on approximate size to maturity. This was comparable to the proportion of laparoscope-assessed immature *C.*

caretta reported for feeding-grounds in the southern Great Barrier Reef (68%, Chaloupka and Limpus 2001) and Moreton Bay (69%, Limpus *et al.* 1994a). This implies that immature and adult *C. caretta* probably have the same exposure to trawl capture, although because of the greater abundance of immature size classes, trawl by-catch impacts upon immature *C. caretta* twice as upon heavily as upon adult *C. caretta*.

C. MYDAS

Immature *C. mydas* comprised about 63% of the reported *C. mydas* by-catch, which is lower than the proportion of laparoscope-assessed immature *C. mydas* reported for feeding-grounds in the southern Great Barrier Reef (75%, Chaloupka and Limpus 2001) and Moreton Bay (92%, Limpus *et al.* 1994b). Limpus *et al.* (1994b) cautioned that the sample of sea turtles from Moreton Bay (a sub-tropical seagrass feeding-ground) was representative of the shallow sub-tidal and inter-tidal feeding habitats sampled, and that larger and mature sea turtles are more likely to occur in the adjacent waters up to 30 m deep, where the trawl captures occurred. If the assumed approximate size at maturity for *C. mydas* was increased to >100cm CCL, then proportion of immature sea turtles in the trawl by-catch was the same as that reported by Chalopuka and Limpus (2001) for reef feeding-grounds in the southern Great Barrier Reef.

N. DEPRESSUS

About 86% of *N. depressus* caught in trawl nets of the Queensland east coast were probably immature, based on approximate size at maturity. This is the first estimate of the proportion of immature and adult *N. depressus* in feeding-grounds dispersed along the Queensland east coast, as there are no comparative feeding-ground studies for this species. *N. depressus* had the highest proportion of immature individuals of any species of sea turtle, based on the sizes of sea turtles reported caught in the Queensland East Coast Trawl Fishery. This possibly reflects the greater exposure of *N. depressus* to human impacts on the continental shelf because this species does not have an oceanic dispersal phase (Walker 1994).

OTHER SPECIES

L. olivacea and *E. imbricata* were the other species where the proportion of immature sea turtles was estimated, being 77% and 65% respectively. This is the first estimate of the proportion of immature and adult *L. olivacea* in feeding-grounds along the

Queensland east coast. The estimated proportion of immature *E. imbricata* has limited application beyond the current estimates of sea turtle by-catch because it is based on a sample size of only 19 individuals and trawl grounds are not the preferred feeding-grounds of *E. imbricata*.

Overall, it appeared that between two-thirds and three-quarters of sea turtle by-catch in the Queensland East Coast Trawl Fishery was comprised of immature individuals. This is an important observation about sea turtle by-catch because it permits by-catch mortality to be proportioned between immature and adult stage classes. This is useful for: (i) assessments of the historic impact (i.e., pre-TED regulation) of the Queensland East Coast Trawl Fishery on sea turtle populations of north eastern Australian (e.g., Heppell *et al.* 1996; Chaloupka and Limpus 1998); and (ii) for simulated sub-population response to the use of TEDs (e.g., Crowder *et al*. 1994).

3.5.2 Mortality rates

Tow time versus mortality

One of the primary purposes of modelling sea turtle by-catch mortality with tow duration was to develop expected mortality rates of trawl-caught sea turtles, given the species caught and the tow duration of a fishing sector. However, this relationship is complex because the condition of a trawl-caught sea turtle is influenced by the oxygen reserves the sea turtle had when it became caught in the net, how long the sea turtle had been struggling within the net, whether the sea turtle was recovering from previous captures and whether the sea turtle was under biological stress such as from nesting (Kemmerer 1989; Tucker *et al*. 1995). As such, it would be unreasonable to expect a linear regression to have a close fit to the data unless these factors could be quantified and incorporated into the analysis. This is confirmed by Henwood and Stuntz (1987), who report that mortality rates did not conform to the expected linear model in tows less than 60 minutes duration in southeastern USA prawn-trawl fisheries, where mortality is less than 1%. They suggested a logistic model might be most appropriate, but considered the linear model to be adequate over the tow durations of interest in the fisheries of the southeastern USA.

The bent-stick analysis was used in the current study because of the apparent 'threshold' in tow duration before sea turtle mortalities were reported in the Queensland East Coast Trawl Fishery (see Figure 3.4 and 3.5); a result in accordance with those of Watson and Seidel (1980) and Henwood and Stuntz (1987). In the Queensland East Coast Trawl Fishery, thresholds for expected direct mortality were ~70 minutes for all species pooled, ~110 minutes for *C. caretta*, ~45 minutes for *C. mydas* and ~115 minutes for *N. depressus,* with no significant relationship detected for *L. olivacea*. The slope of the bent-stick regressions suggested that once a threshold in tow duration was reached, direct mortality rates increased rapidly. This was particular the case for *C. caretta*.

General conclusions that can be drawn from the mortality versus tow duration analyses suggest that for most species, there was a positive correlation between tow duration and mortality, as reported by Watson and Seidel (1980) and Henwood and Stuntz (1987). Lutcavage and Lutz (1996) speculated that mortality rates of trawl-caught sea turtles would differ between geographic areas and between species because of physiological capacities and size differences. No mortalities were reported for 48 trawl captures of tagged *N. depressus* (Limpus and Reimer 1994). Similarly, Poiner and Harris (1996) noted that *N. depressus* had the lowest mortality rate of trawl-caught sea turtles in the Northern Prawn Fishery, although sample sizes for species other than *N. depressus* were relatively small. Mortality rates from the current study support speculation that *N. depressus* may have a greater tolerance to trawl-capture.

Sea turtles that are repeatedly caught in trawl nets over a short time (i.e., < 24 hours) may be more likely to die than sea turtles that have experienced only a single trawl capture. Lutcavage (1992) suggests that the strong positive correlation between sea turtle mortality and tow duration is evidence that multiple recaptures increase by-catch mortality rates. However, sea turtles that die as a result of repeated trawl captures would probably die even if tow times were short, confounding the correlation. Anecdotal reports from fishers suggest that multiple recaptures of individual sea turtles do occur, but the possibility that multiple recapture leads to increased mortality remains speculative.

Mortality rates

Observed mortality rates were highly variable between fishing sectors of the Queensland East Coast Trawl Fishery, as speculated by Dredge and Trainor (1994), and were also highly variable between species as suggested by Poiner and Harris (1996). The contrast in observed mortality rates derived from data stratified only by fishing sector (i.e., all species pooled) or species (i.e., all sectors pooled) with that stratified by fishing sector and species suggests that ideally, mortality rates of sea turtle by-catch should be reported as: (i) sector-specific to take into account operational characteristics of fishing in local areas (e.g., tow duration); and (ii) species-specific to take into account submergence capabilities of different species (Lutcavage 1992; Poiner and Harris 1996).

However, overall mortality rates are the most common comparison between fisheries. The observed mortality rate for the Queensland East Coast Trawl Fishery for all sectors pooled was considerably lower than that reported by selected volunteer fishers in the Northern Prawn Fishery (Poiner and Harris 1996) or estimated for the southeastern trawl fisheries of the USA (Henwood and Stuntz 1987). Mortality rates in the Tiger and Endeavour Prawn sectors of the Queensland East Coast Trawl Fishery were less than half those reported for fishing operations targeting the same penaeid species in the Northern Prawn Fishery (Poiner and Harris 1996). The Tiger and Endeavour Prawn sectors of the Queensland East Coast Trawl Fishery catch a similar mix of sea turtle species (i.e., *N. depressus*, *L. olivacea* and *C. mydas*), but have slightly shorter tow durations than in the Northern Prawn Fishery (i.e., 144 minute tows compared to 180 minute tows). Either this difference in tow duration was sufficient to reduce by-catch mortality or mortality was under-reported by fishers participating in the sea turtle by-catch monitoring program of the Queensland East Coast Trawl Fishery.

Observed direct mortality rates in sectors where the mean tow duration was less than 76 minutes were similar to mortality rates reported in trawls less than 90 minutes in the USA (Watson and Seidel 1980).

Observed direct mortality rates of sea turtles caught in the Queensland East Coast Trawl Fishery were significantly lower than the 30% mortality rates reported by Limpus *et al.* (1992) for post-nesting *C. caretta* caught in trawls from a variety of fisheries in northeastern Australia (i.e., the Northern Prawn Fishery to the New South Wales Oceanic Prawn Fishery). Observed direct mortality rates of sea turtles caught in the Queensland East Coast Trawl Fishery were also lower than the 11% mortality rate reported for 62 *L. olivacea* caught during trawling operations off Townsville (Harris 1994). Possible explanations for the differences in mortality rates include the aggregation of mortality rates across fisheries (Limpus *et al.* 1992), the mis-identification of comatose sea turtles i.e., considering comatose individuals as dead (Poiner and Harris 1996) or the deliberate under-reporting of dead sea turtles by fishers participating in the sea turtle by-catch monitoring program for the Queensland East Coast Trawl Fishery. To address the uncertainty in reporting, mortality rates have been estimated for a range of scenarios (i.e., direct and potential, observed and expected) including worst-case estimates (Gribble *et al.* 1998). Mortality rates were used to derive a range of estimates of the annual kill of sea turtles in the Queensland East Coast Trawl Fishery that could be compared to simulated impacts from sea turtle population models.

3.5.3 Impact of by-catch on sea turtle populations

Trawling was one of a multitude of anthropogenic activities that impacted on sea turtle populations. Results from the current study indicate that the Queensland East Coast Trawl Fishery impacted most heavily upon *C. caretta* and *N. depressus,* particularly the immature size classes of these species and potentially has a significant impact on *C. mydas* if comatose sea turtles die as a consequence of trawl-capture.

Impacts on C. caretta

Numbers of *C. caretta* nesting in eastern Australia have declined by 50 to 80% over the past 25 years and this species is now listed as Endangered under Australian Commonwealth and State legislation (see Chapter 2, Table 2.1). Limpus and Reimer (1994, p45) speculated that the “order-of-magnitude of the kill of *C. caretta* in commercial fisheries in northern and eastern Australia is many hundreds, possibly greater than a thousand annually”, with the majority of deaths being attributed to trawl by-catch mortality. The east Australian sub-population of *C. caretta* is drawn from feeding-grounds located from the Arnhemland coast to southern New South Wales (see Chapter 2, Figure 2.2, Limpus *et al.* 1992). Trawl fisheries in this area encompass the Gulf of Carpentaria section of the Northern Prawn Fishery, the Torres Strait Prawn

Fishery, the Queensland East Coast Trawl Fishery and the oceanic and estuarine trawl fisheries of New South Wales. Estimates of the annual catch and kill of sea turtles are available for three of these fisheries (Table 3.13). An estimated ~3,200 *C. caretta* were caught annually in trawl fisheries of northeastern Australia with ~100 *C. caretta* dying during capture (i.e., direct mortality). If comatose sea turtles also died, then ~250 *C. caretta* died annually as a consequence of trawl capture in the trawl fisheries of northeastern Australia prior to TED regulations (Table 3.13).

Table 3.13 Estimated kill of sea turtles in trawl fisheries of northeastern Australia under various by-catch mortality scenarios

Species	Fishery	Number of sea turtles killed					
		Mean estimated catch	Observed direct mortality [A]	Observed potential mortality [B]	Expected direct mortality [C]	Expected potential mortality [D]	Expected direct mortality USA[E]
C. caretta	QECTF [F]	2,938	31	149	36	193	324
	TSPF [G]	85	11	16	9	17	18
	GOC [H]	175	59	59	37	49	47
	Total	3,198	101	224	82	259	389
N. depressus	QECTF [F]	968	18	41	22	57	188
	TSPF [G]	400	4	8	11	26	83
	NPF [I]	3,085	337	1,016	186	278	824
	Total	4,453	359	1,065	219	361	1,095
C. mydas	QECTF [F]	1,562	10	101	16	109	231
	TSPF [G]	145	4	16	2	14	30
	NPF [I]	402	48	120	9	51	107
	Total	2,109	62	237	27	174	368
L. olivacea	QECTF [F]	323	9	43	11	32	61
	TSPF [G]	18	0	0	1	2	4
	NPF [I]	643	80	202	42	90	172
	Total	984	89	245	54	124	237
E. imbricata	QECTF [F]	80	0	10	2	8	14
	TSPF [G]	6	0	0	0	1	1
	NPF [I]	241	64	170	16	34	64
	Total	327	64	180	18	43	79
All species [J]	QECTF [F]	5,901	61	319	104	407	826
	TSPF [G]	652	7	26	28	70	136
	NPF [I]	5,238	943	2,043	340	735	1,214
	Total	11,791	1,011	2,388	472	1,212	2,176

[A] Observed direct mortality = (dead sea turtles/total sea turtles captured) by QECTF vessels (see Table 3.9); [B] Observed potential mortality = ((dead+comatose sea turtles)/total sea turtles captured) by QECTF vessels (see Table 3.9); [C] Expected direct mortality = based on percentage of sea turtles estimated to be dead when landed using the bent-stick relationship between observed direct mortality and mean tow duration of QECTF vessels (see Table 3.11); [D] Expected potential mortality = based on percentage of sea turtles estimated to be dead or comatose when landed using the bent-stick relationship between observed potential mortality and mean tow duration of QECTF vessels (see Table 3.11); [E] Expected direct mortality USA = based on percentage of sea turtles estimated to be dead when landed using the linear relationship between expected direct mortality and mean tow duration derived by Henwood and Stuntz (1987) using data from shrimp fisheries in the southeastern USA (see Table 3.11); [F] QECTF = Queensland East Coast Trawl Fishery; [G] TSPF = Torres Strait Prawn Fishery, estimates from Robins and Mayer 1998; [H] GOC = Gulf of Carpentaria, estimates from Poiner and Harris 1996, assuming on 87.5% of *C. caretta* captures occurring in the GOC; [I] NPF = Northern Prawn Fishery, estimates from Poiner and Harris 1996, [J] includes unidentified sea turtles and all *C. caretta* captured in the NPF.

These are minimum estimates because of the voluntary nature of the underlying data and do not include sea turtles that were apparently healthy when released but might have died subsequently as a consequence of internal injuries (Limpus and Reed 1985b) or capture-related stress (Tucker *et al.* 1995). This is the same order-of-magnitude of by-catch mortality suggested by Heppell *et al.* (1996) to have caused the 50 to 80% decline in the eastern Australian sub-population of *C. caretta.* The estimated level of direct mortality from the trawl fisheries of northeastern Australia is similar in size to the "medium" level of impact surmised by Chaloupka and Limpus (1998) during simulations of the population dynamics of southern GBR *C. caretta* (i.e., "medium" = 250 immature/adult sea turtles killed per year for 16 years from 1978). Therefore, the estimated level of *C. caretta* by-catch mortality presented here confirms speculation based on hypothetical population dynamic models and further supports the use of TEDs to mitigate trawl by-catch impacts, particularly upon the endangered *C. caretta*.

Impacts on other species

It is difficult to surmise the significance of estimated levels of by-catch mortality on the sub-populations of the other species of sea turtle that inhabit Australian waters because of the lack of context in which to place the estimated annual kill (Table 3.13). Theoretically, any sea turtle by-catch mortality is of concern because of the low recovery capability (sensu Stobutzki *et al.* 2001a) of sea turtle populations (as discussed in Chapter 1, section 1.2.2). By-catch mortality of *N. depressus* warrants concern because of this species is endemic to Australasian waters and is restricted to the Australian continental shelf for all its life stages. As a consequence, this species had relatively greater exposure trawl by-catch mortality than other sea turtle species, although Limpus *et al.* (2002) reported that the eastern Australian population of *N. depressus* (i.e., that nesting in the southern Great Barrier Reef lagoon, see Chapter 2, section 2.4.2) appears to be stable over the last three decades. Estimated by-catch mortality for *C. mydas* was an order-of-magnitude less than the estimated indigenous harvest of northern Australia (EA 1998; Dr Colin Limpus, QPWS, personal communication 2003)[6]. However, concern over the status of the two sub-populations of

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[6] The mean annual by-catch mortality of *C. mydas* in "northern Australia" (= northern + southern GBR stock + GOC & NT) was estimated to be in the order of 62 to 368 individuals compared to the mean annual direct harvest of *C. mydas* in northern Australia which was reported in the Draft Recovery Plan for Marine Turtles in Australia (EA 1998) to be 500-1000 (southern GBR) + 1000s (northern GBR) + 1000s (GOC & NT stock). Taking the worst-case scenario, trawl by-catch mortality for *C. mydas* is in the hundreds of individuals per year compared to a direct harvest in the thousands of individuals per year.

C. mydas in eastern Australia (Chaloupka 2002) supports the removal of trawl by-catch mortality on this species through the use of TEDs.

Validation of estimates

A broad scale, labour intensive observer program in the Queensland East Coast Trawl Fishery would have been required before the mandatory use of TEDs in the fishery to validate the estimates of the current study. Young *et al.* (1993) suggests that observer coverage should be between 20 and 35% of fishing activity for by-catch estimates to be considered statistically reliable. If this estimate were applied to the Queensland East Coast Trawl Fishery, then between 17,000 and 29,750 observer days would be required to validate the sea turtle CPUE estimates. The fishery has an annual value of ~$AUD 120M and an appropriately sized observer program would cost between $AUD 2M and $AUD 5M.[7] This is about 2% to 4% of the landed value of the Queensland East Coast Trawl Fishery. However, such expenditure is not politically or practically feasible in the Queensland East Coast Trawl Fishery at present because of the economic situation of most participants in the fishery, lack of resources by the managing agency (the Queensland Fisheries Service), the lack of physical space on most Queensland trawlers to accommodate a fishery-independent observer and the mandatory requirement for TEDs to be used at all times.

Stratification of observer sampling by fishing sector and fishing season might reduce the number of days required by such an observer program and could be explored through power analysis of sampling designs. The sea turtle CPUEs recorded in the current study provide a strong basis for such a power analysis. Relaxation or exemption of the regulated use of TEDs in the Queensland East Coast Trawl Fishery would need to be negotiated in order for observers (fishery-independent or fishery dependent) to monitor sea turtle by-catch. This is an unlikely scenario because of the social and political issues associated with sea turtle by-catch in general, and intense scrutiny of fishing practices in the Great Barrier Reef World Heritage Area.

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[7] Based on an observer program with each observer spending 200 days at sea per year and considering wages only costs of $AUD30,000 per observer.

3.6 CONCLUSIONS

Sea turtle populations are particularly sensitive to anthropogenic impacts because sea turtles are long-lived, have delayed sexual maturity, and high survivorship of large immatures and adults (Crouse 1999). The results presented in this chapter fill the "conspicuous absence of data on location, catch rate and species composition of the incidental turtle catch from the Queensland East Coast Trawl Fishery" identified by Dredge and Trainor (1994, p141) and Limpus and Reimer (1994). The estimated pre-TED impact of trawl fisheries on the east Australian sub-population of *C. caretta* was within the order-of-magnitude proposed on the basis of hypothetical population models (Heppell *et al.* 1996) to have caused the 50% to 80% decline in nesting numbers of *C. caretta*. The results of this chapter support the mandatory use of TEDs in the demersal prawn trawl fisheries of northeastern Australia. Sea turtle by-catch information presented here has value in providing estimates of the relative impact of by-catch mortality on immature and mature sea turtles on a species basis. This information is essential to modelling potential population responses to the use of TEDs throughout a fishery (e.g., Crowder *et al.* 1994). The data have also provided the first estimates of the proportion of immature and adult sea turtles in Australian feeding-grounds for *N. depressus* and *L. olivacea*. Greater understanding of the distribution and demographics of sea turtles in feeding-grounds, such as those of the Queensland east coast, are essential if anthropogenic impacts are to be effectively managed and sea turtle populations monitored to ensure that management measures achieved the desired outcomes.