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ARTICLE

The Apparent Change in Population Structure of Green Turtles (*Chelonia mydas*) at a Northern Great Barrier Reef Foraging Site Over Three Decades and an Evaluation of Potential Causes

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ABSTRACT

Green Island lies approximately 11 km off the north Queensland coast of Australia. The associated 710-ha reef flat supports a *Chelonia mydas* foraging population, which has been monitored by the Queensland Government's Threatened Species Program since 1988. Population census data for this area show a significant adult age-class-specific population decline for *C. mydas*. Preliminary evaluation indicates the decline in adult age-classed turtles is unlikely to be caused by incidental capture, habitat degradation, pollution, change of foraging sites or climate change. Traditional take (locally or during reproductive migrations) provides a plausible explanation. A continuation of this trend may threaten the existence of *C. mydas* in this part of the Great Barrier Reef World Heritage Area.

1 | Introduction

The green turtle (*Chelonia mydas*) is considered vulnerable to extinction under the *Nature Conservation Act* 1992 (Qld) (NCA) and the *Environment Protection and Biodiversity Conservation Act* 1999 (Cth) (EPBC Act) and internationally listed as Endangered (IUCN 2023). South Pacific, *C. mydas* populations are threatened by feeding and nesting habitat loss and degradation, including chemical, plastic and light pollution (Duncan et al. 2021), unsustainable take (Dethmers and Baxter 2011), feral predators (Whytlaw, Edwards, and Congdon2013), incidental capture in active and discarded fishing gear (Wilcox et al. 2015) and the direct effects of climate change (e.g., feminization and low hatchling production (Jensen et al. 2018; Stuart-Smith et al. 2018; Becker, Brainard, and van Houtan 2019).

Green Island Reef is part of a regionally important feeding area for *C. mydas* within the Great Barrier Reef (GBR), Australia (Limpus, Couper, and Read 1994; Fuentes, Gyuris, and Lawler 2006). After hatching and an approximate 10-year oceanic phase, *C. mydas* recruit to inshore areas where they typically remain associated with, high-site fidelity and a restricted home range, feeding

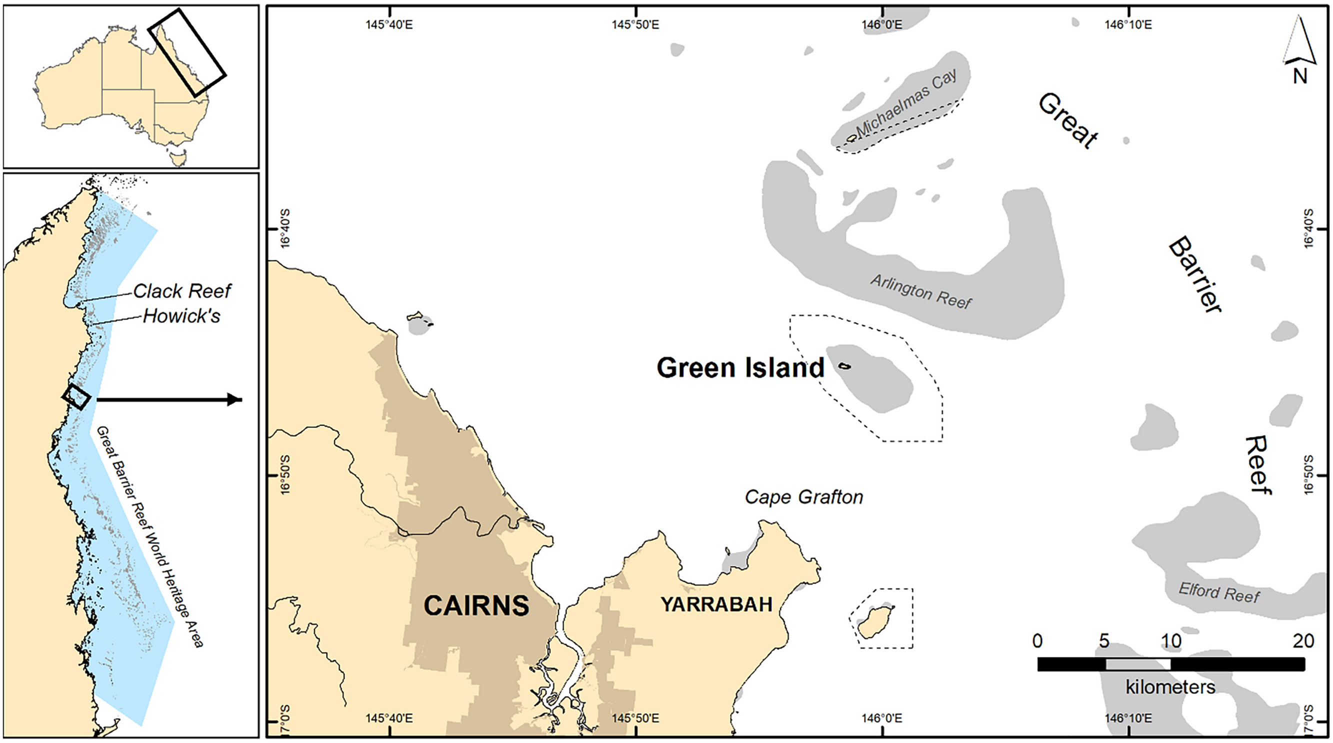

FIGURE 1 | Location of Green Island within the Great Barrier Reef World Heritage Area and the proximity of Cairns City. Dashed lines delineate boundary of Gunggandji Traditional Use of Marine Resources Agreements (TUMRA).

area (Limpus, Couper, and Read 1994; Chaloupka, Limpus, and Miller 2004) (Hamann et al. 2006; Shimada et al. 2014, 2016; Palaniappan and Hamid 2017; Bell et al. 2019; Shimada et al. 2020; Pillans et al. 2021, 2022). Although opportunistically omnivorous for the first few years of their life (Limpus 2007), once recruited to a foraging area, GBR *C. mydas* remain primarily herbivorous and spend the majority of their time feeding in waters less than 4-m depth (Lanyon, Limpus, and Marsh 1989; Read 1991; Forbes 1996; Brand-Gardner, Lanyon, and Limpus 1999; Hays et al. 2002; Prior, Booth, and Limpus 2016).

Although *C. mydas* occasionally nest on Green Island cay, this is unlikely to contribute meaningfully to population viability; however, the reef flat provides a high-quality foraging area. Foraging areas of high quality and abundance facilitate faster growth rates and reduce the period between reproductive migrations, which in turn promotes population size and resilience (Meylan et al. 2022). The Green Island Reef platform is an important foraging resource to support growth of *C. mydas* juveniles and subadults, and adult's acculation of sufficient body condition to sustain successful breeding migration and nesting activity.

Here, we describe the changing age structure of the resident *C. mydas* population on Green Island Reef over a 32-year period and explore potential causes of the decline in adult age classes, including seagrass cover, forage choice, anthropogenic impacts and climate change.

2 | Methods

Between 1988 and 2022, the Queensland Department of Environment and Science recorded population census data for *C. mydas* caught as part of an ongoing monitoring project on the reef surrounding Green Island. Quarterly seagrass monitoring was carried out from 2001 through to 2022, and the ingesta of a subsample of turtles was analysed for diet choice in 2017, 2019 and 2021. This collection of records has enabled a temporal study of seagrass availability, food preference and *C. mydas* population trends on Green Island Reef over 34 years.

2.1 | Study Site

Green Island (−16.7588°S; 145.9735°E) is a 12-ha forested coral cay situated atop a midshelf platform reef, located 27 km northeast of Cairns, Australia (Figure 1), and has been a popular tourist destination for over 100 years (GBRMPA 2014). Because the island attracts approximately 300,000 visitors annually, it is an important regional economic resource (QPWS 2003). The reef surrounding Green Island covers approximately 710 ha (QPWS 2003) and is made up of sand flats, coral reefs and seagrass meadows (Figure 2). The reef habitat is classified as an 'Exposed Mid Shelf Reef' under the Reef Bioregions of the GBR classification scheme (GBRMPA 2009). The reef immediately surrounding Green Island is classified as a Marine National Park ('green' zone), which prohibits take of natural resources. Indigenous Australians retain customary hunting rights, within their recognized traditional country, throughout the Marine Park.

2.2 | Turtle Data

Turtles were captured during systematic searches of the reef flat foraging habitat around Green Island. Turtles were captured by

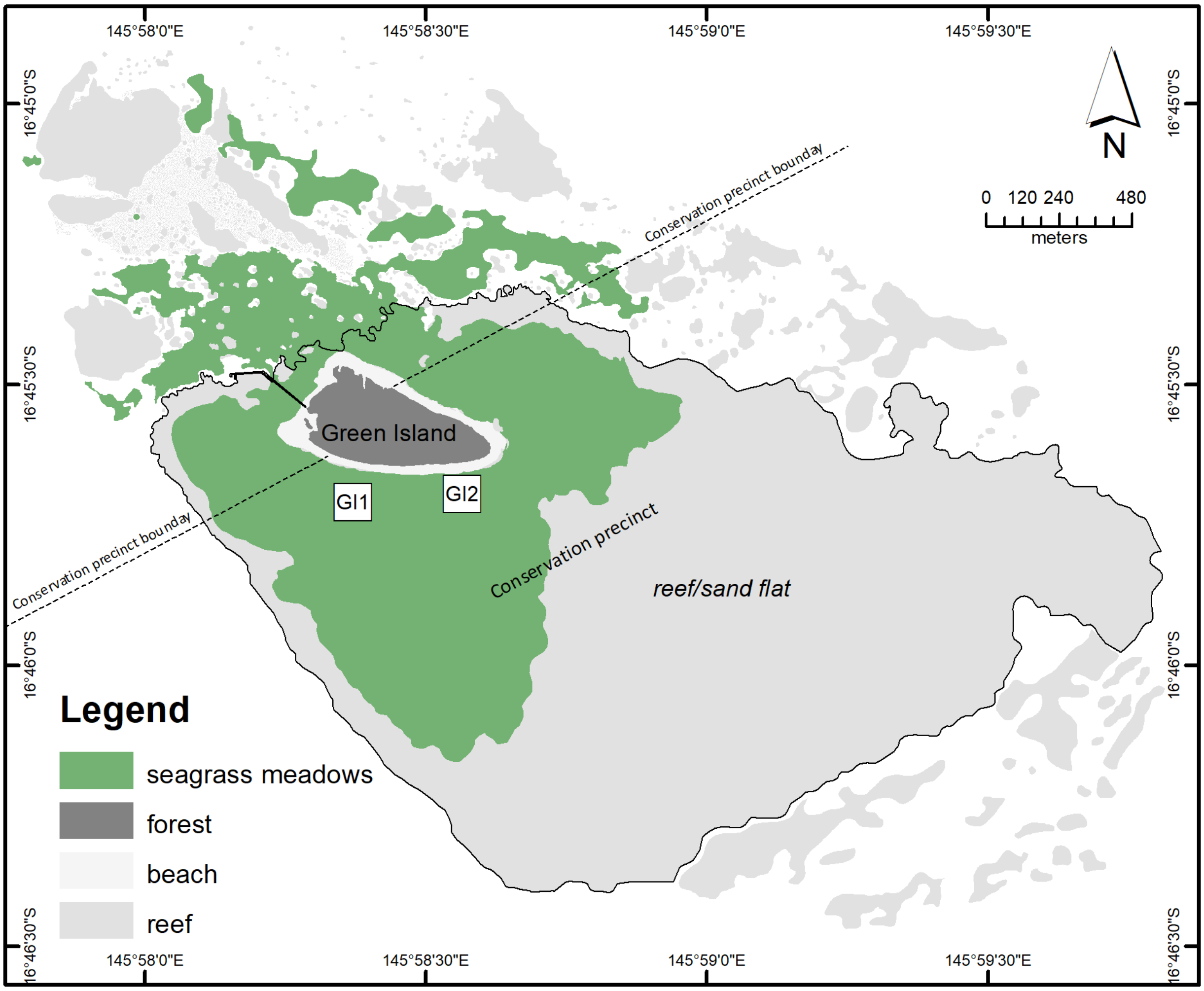

FIGURE 2 | Map of Green Island showing the location of reef flat, seagrass meadows (2003 extent; McKenzie, Yoshida, and Unsworth 2014) and seagrass long-term monitoring sites G11 and G12.

day using the turtle rodeo and beach jump methods (Limpus and Reed 1985). Whereas daytime rodeo methods were typically conducted, during suitable tide times, nighttime turtle rodeo captures were also undertaken adjacent to Green Island, using a hand-held 12-V floodlight. Occasionally, daytime beach jumps were attempted on the reef flat at low tide or during nighttime during high tides.

Following capture, the turtle's left fore-flipper was tagged with self-locking titanium flipper tags (Stockbrands Company, Pty. Ltd., Western Australia) (Limpus 1992). The curved carapace length (CCL) was measured (±0.1 cm) along the midline from the junction of the skin and carapace above the neck to the midline junction of the supracaudal scutes using fibreglass tapes (Limpus, Couper, and Read 1994). Fibreglass tape calibration was undertaken regularly against steel rules. Any large barnacles on the carapace that were likely to interfere with CCL measurement were removed. Turtle mass was measured (±0.2 kg) using calibrated scales. Turtle sex was assigned to adults based on tail length past the carapace edge (TLC), with males assigned if TLC > 25 cm (Limpus 1993). Turtles were typically released at their point of capture within 1 h. All turtles recaptured from past surveys were released with at least two secure titanium tags, one on each front flipper. All turtle tagging, capture and observational data were recorded in the Queensland Government database.

Morphometric data, collected from turtles captured in the Howick Group of islands over the same period of time, were compared with Green Island foraging turtles to see if a similar decline in the larger part of the population was also occurring at other reefs.

2.3 | Seagrass Sampling

Long-term monitoring of seagrass on the main turtle foraging area of Green Island reef flat was conducted at 3-to-6-month intervals between November 2001 and April 2022. Initially, only a single site (5.5 ha) was established (GI1). However, a second site (GI2) was introduced in April 2005. Both sites were located on the southern side of the cay in the declared conservation precinct, away from vessel and tourist recreational activities (Figure 2). Sites were permanently marked, ensuring that the same area was assessed each sampling event (approximately each January, April, July and October). Observers used a 50 cm × 50 cm quadrat (not anchored to the substrate) to record estimates of per cent cover of seagrass shoots originating in the quadrat and seagrass species composition as per globally standardized Seagrass-Watch protocols (McKenzie, Campbell, and Roder 2003; McKenzie et al. 2000). Geotagged photo-quadrats of every quadrat examined in the field were captured for quality control. Seagrass percentage cover estimates were validated postsurvey against a set of standardized percentage cover measures to ensure accuracy and consistency between sampling events and observers. Species composition estimates were similarly validated postsurvey using the photo-quadrats to verify species identification.

2.4 | Forage Preference

In 2017, 2019 and 2021, a subsample of turtles underwent gastric lavage to collect a sample of their most recent feeding event. Samples were obtained from the mouth, oesophagus and crop using the stomach flush technique (Forbes and Limpus 1993). Seagrass and algae were identified to genus under a dissecting microscope (Australian Instrument Services Model ST-36C-2LGO), and relative frequency was quantified in triplicate subsamples from the gastric lavage sample examined on a 25-point grid (75 data points per sample).

2.5 | Data Analyses

CCL for captured juvenile *C. mydas* on Green Island reef, for five nonconsecutive years (2012, 2017, 2019, 2020 and 2022), was analysed as described in Bolten (1999). These data were compared with the CCL from turtles using a remote, undisturbed foraging site some 300 km to the north to test for variation in the ability of forage areas to support good body condition of resident turtles (Figure 3).

Individuals were allocated to a size class based on CCL as follows: adult-sized (> 86 cm CCL), subadults (65- to 86-cm CCL), juveniles (38- to 64-cm CCL) and posthatchlings (> 5.5- to 37-cm CCL) (Chaloupka and Limpus 2005; Limpus, 2007). The size of the smallest nesting female varies with different populations (Limpus et al. 2003; Limpus 2007), so the adult-sized category may include both adult and immature individuals.

Kernel density estimators were used to smooth the length frequency histogram to compare CCLs of the first sampling year, where the number of turtles caught exceeded 45 individuals (1989) with subsequent sampling years. The significance of permutation tests was analysed using a one-tailed *t* test. In addition, a quantile additive regression model (qgam) for the upper CCL size (quantile = 0.75) of *C. mydas* measured between 1988 and 2019 was used to look at changes in CCL over the course of the data collection. Both analyses were carried out in the statistical package R (Studio) ver. 4.0.0.

We used generalized additive mixed-effect models (GAMM) in R 4.2.2 (mgcv package) (Wood 2020; R Core Team 2022) to show the variation of seagrass cover over time. A logit transformation was applied to the per cent cover data, and the logit was treated as normally distributed (Lessafre, Rizopoulos, and Tsonaka 2007) using the quasibinomial data distribution option of the mgcv package. For each response variable (logit of seagrass cover), a smoothness selection was fitted by maximum likelihood through the Laplace approximation (Wood 2010). Two separate models were produced: first, an overall model with the sites pooled and, second, a model including two different smoothers for each site (GI1 and GI2). The models include random effects to the site and quadrat levels.

Trend analysis were conducted to determine if there was a significant trend (reduction or increase) in seagrass cover at a particular site and sites pooled (averaged by sampling event) over all time periods. A Mann–Kendall test was performed in R 3.2.1 (R Core Team 2014, fume package) to detect overall trends over time. Because the test assumes independence between observations, we first checked visually for potential autocorrelation for each analysis through spline correlogram plots of lme model residuals (Zuur, Ieno, and Elphick 2010),

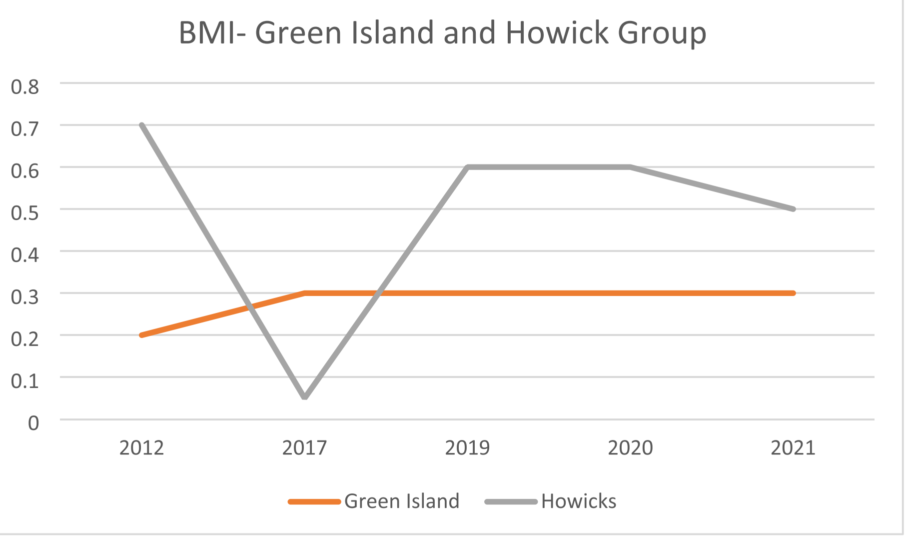

FIGURE 3 | Body mass index comparison between turtles caught at Green Island and the Howick Group of islands.

TABLE 1 | Summary of *Chelonia mydas* captures between 1988 and 2021 on Green Island Reef flat.

Year	P	ISR	WSR	Juv	SA	A
1988	27	0	0	11	14	2
1989	93	3	0	53	37	3
1990	108	13	0	64	41	3
1996	6	0	0	0	3	3
1997	24	1	0	20	3	1
1998	8	1	0	8	0	0
1999	2	0	0	2	0	0
2000	3	0	0	2	1	0
2002	2	0	0	0	0	2
2003	15	0	0	14	1	0
2004	97	5	0	92	5	0
2005	19	7	0	18	1	0
2007	17	3	0	17	0	0
2012	101	0	1	92	9	0
2013	10	6	0	10	0	0
2016	45	5	7	44	1	0
2017	58	22	7	53	5	0
2019	30	8	0	25	5	0
2020	50	9	0	51	1	0
2021	29	14	1	29	0	0
Total	744	97	16	605	127	14

Abbreviations: A, adult sized; CCL, curved carapace length; ISR, inter seasonal recapture; Juv, juvenile; P, primary; SA, subadult; WSR, within season recapture.

but no *p*-value adjustment was required because autocorrelation was not significant.

2.6 | Turtle Harvest

The Department of Environment and Science has maintained a full-time Queensland Parks and Wildlife Service Ranger Base on Green Island since 1985. During this study, Indigenous hunting was recorded when observed directly and opportunistically during sea turtle monitoring activities and via conversations with hunters, boaters and the Queensland Parks and Wildlife Rangers.

3 | Results

3.1 | Turtle Demographics

Between 1988 and 2022, the *C. mydas* population on Green Island reef was sampled in 18 of the 34 years with a total of 744 captures, of which 74 turtles were recaptured once and 2 turtles were recaptured twice. Fifteen turtles were caught within a

TABLE 2 | Number of individuals (*N*) and curved carapace length (CCL; mean, standard deviation and range [cm]) for all primary (first-time) captures for years where $N > 10$.

Year	1988	1989	1990	1997	2003	2004	2005	2007	2012	2016	2017	2019	2020	2021
No	27	90	95	23	15	92	11	14	100	45	58	30	51	29
Mean	68.54	63.56	63.03	53.44	51.06	49.87	49.75	46.01	49.75	49.4	48.44	53.59	49.0	48.9
SD	17.11	12.57	14.05	8.59	8.02	7.16	9.64	3.52	9.07	6.03	7.06	10.76	7.49	4.91
Range	41.0–95.5	38.5–107.5	40.1–100.0	41.6–68.6	45.5–75.6	38.9–69.7	41.6–70.4	39.8–51.8	40.7–88.4	0.40.0–67.7	39.3–75.3	40.1–79.9	36.7–71.1	41.6–71.9

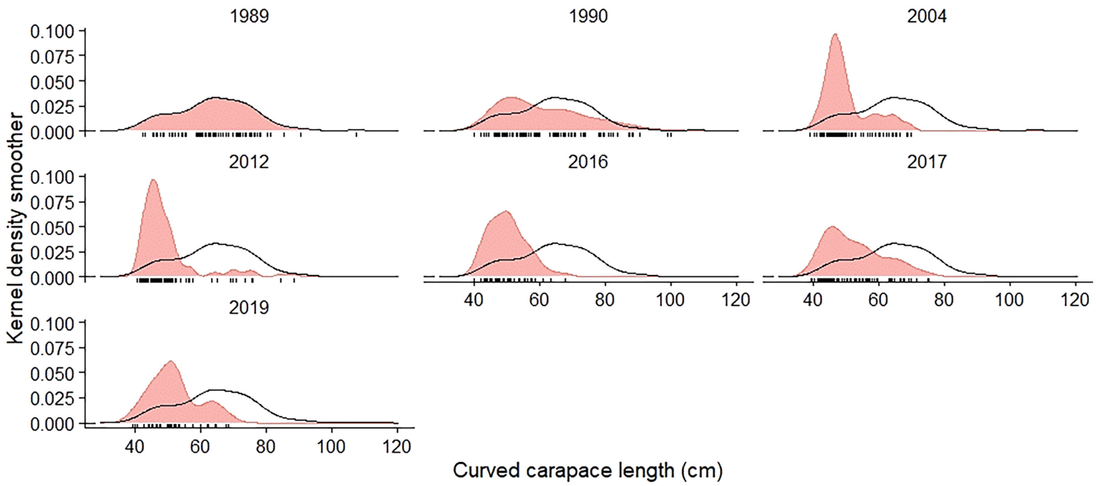

FIGURE 4 | The distribution of curved carapace length (cm) of foraging *Chelonia mydas* on Green Island Reef. The first sampling year where the number of turtles caught exceeded 45 individuals (1989) is used as a reference (black line) to compare subsequent sampling years. Kernel density estimators were used to smooth the length frequency histogram.

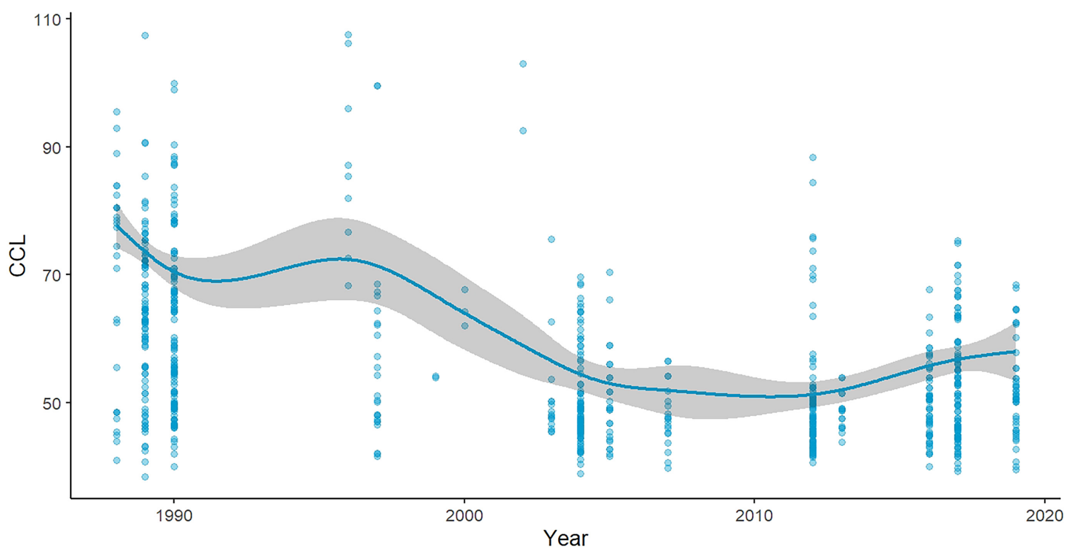

FIGURE 5 | A quantile additive regression model (qgam) for Curved Carapace Length (quantile = 0.75) of *Chelonia mydas* measured between 1988 and 2019. The shaded area represents the 95% confidence interval, and semitransparent points represent individual samples.

single year's sampling effort (within season recapture). Ninety-eight per cent of captures was juvenile and subadult individuals (Table 1). Individual CCL ranged from 38.5 to 107.5 cm, and annual mean CCL varied from 48.44 to 68.54 cm (Table 2).

A decline was observed in the mean CCL of *C. mydas* on Green Island reef over the project duration (Figures 4 and 5; Table S1). Given the catch of juveniles was relatively constant, CCL decline is likely resulting from a loss of adult- and pubescent-sized individuals from the population. On monitoring trips between 1988 and 1997, out of a total of 258 captures, 12 were adult-sized individuals (5%); in contrast, between 1998 and 2019, there were two adult-sized animals caught out of a total of 407 (0.05%) (Table 1). Similarly, between 1988 and 1997, out of a total of 258 captures, 98 (38%) were subadult size, whereas in the period 1998 to 2019, only 28 of 407 (7%) captures were subadults. This decline contrasts with turtle size recorded at the Howick Group of islands. During this time, there was no significant change in mean CCL at the Howick Group ($r^2 = 0.0261$; $df = 5$; $p = 0.9557$). Furthermore, the annual mean CCL for turtles at the Howick Group ranged from 78.37 to 92.87 cm, which was considerably larger (48.44 to 68.54 cm) than CCLs recorded at Green Island.

3.2 | Seagrass Surveys

Over the 21 years of seagrass monitoring (2001–2022), seagrass cover fluctuated within and between years. Total seagrass abundance (per cent cover) significantly increased prior to 2005 and then progressively declined from 2006 until 2012 at both sites (Figure 6 and Table S2). In 2013, the onset

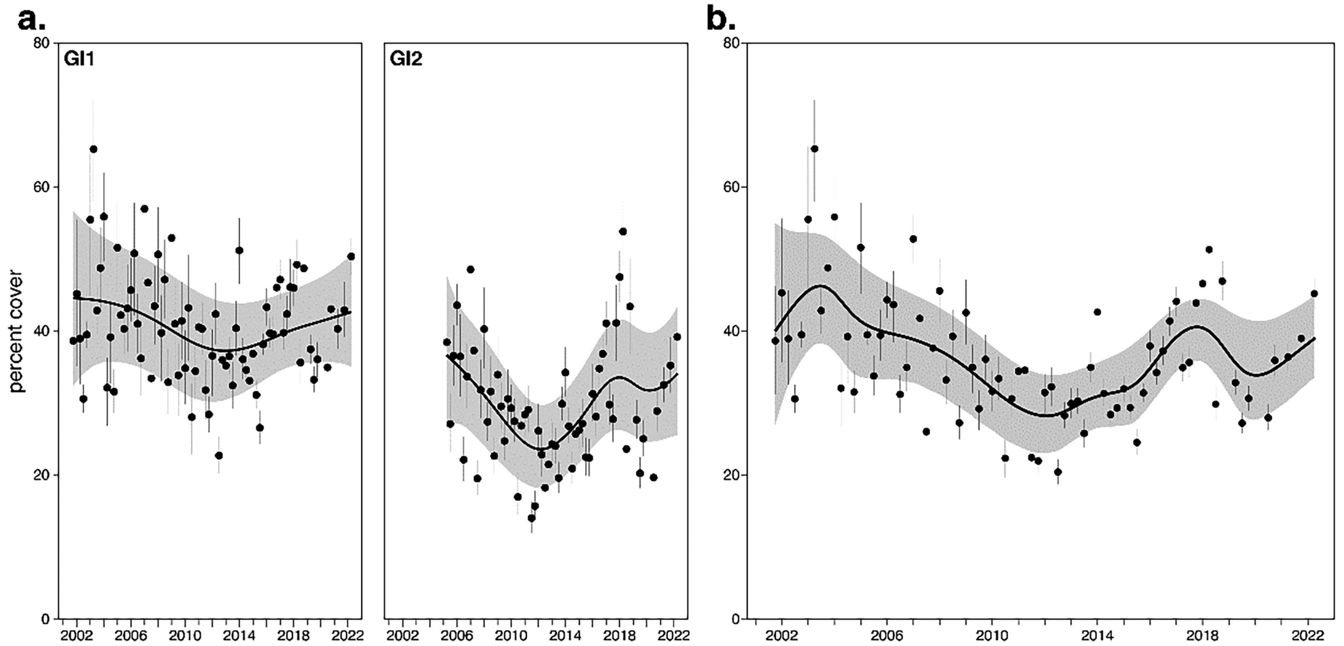

FIGURE 6 | Trends in seagrass abundance (per cent cover, ± SE) at G11 and G12 sites (a) and sites pooled (b) represented by a GAMM plot. Trends are solid lines with shaded areas defining the 95% confidence intervals of those trends.

of seagrass recovery occurred, with total percentage cover gradually increasing (Figure 6). Between 2018 and 2020, abundance declined at GI2 but recovered in 2021 and 2022. A long-term declining trend in total seagrass cover (approximately 4% from 2001 to 2022) was detectable at one of the two sites (GI1) (Table S3). Five seagrass species were observed within the long-term monitoring sites, the most common being *Cymodocea rotundata*, followed by *Thalassia hemprichii*, and to a lesser extent by, *Halodule uninervis*, *Halophila ovalis* and *Oceana serrulata* (previously *Cymodocea serrulata*). *C. rotundata* dominated both sites, particularly at GI1 with an average contribution of 0.76 compared to 0.55 at GI2. The proportion of *C. rotundata* fluctuated over the years and, although declined significantly at both sites, remained the dominant species over the study period (2001–2022) (Table S3). The least common species was *O. serrulata*, observed at GI1 on three separate occasions in 2022, 2014 and 2019.

3.3 | Diet Preference

The proportion of seagrass to algae ingested by the group of turtles sampled varied from 67% ($n = 20$) in 2017 to over 92% in 2019 ($n = 10$) and 93% in 2021 ($n = 10$). Of the individual turtles sampled in 2017, 2019 and 2021, 55%, 80% and 80%, respectively, had a predominantly seagrass diet, whereas the remainder had a mixed diet, with the exception of one turtle captured in 2017, which had only consumed algae.

3.4 | Turtle Harvest

The *C. mydas* population foraging on Green Island reef was subjected to hunting pressure throughout the monitoring period. Rangers collected multiple, anecdotal observational data of turtle hunting occurrence over Green Island reef. In addition, scientists working on the turtle monitoring project observed hunting directly on several occasions. Hunting specifically targeted adult and larger subadult turtles, particularly females.

4 | Discussion

Marine turtle body size (CCL) is an important correlate of turtle biology and ecological health (Peters 1983; Calder 1984). A trending reduction in average body size over time can be indicative of a population in decline (Bhupathy and Saravanan 2006; Fenberg and Roy 2008; Henrich et al. 2010; Eisemberg et al. 2011) and a cause of conservation concern for the Green Island reef *C. mydas* population. This is especially worrying as turtles take a long time to mature and turn around such a trend. The southern GBR genetic stock is still a recovering population, decimated due to the turtle cannery on Heron Island in the 1940s (Limpus and Reed 1985).

A local population decline has far-reaching implications though, as foraging and nesting stocks of turtles are interdependent, despite the large distances between them. Turtle T32477 was recaptured at Green Island reef after being tagged while nesting approximately 900 km south, at Northwest Island in the Capricorn-Bunker Group of islands off central Queensland in December 1987 (TurtleNet database). Loss of adult-sized *C. mydas* from the Green Island reef will likely contribute to a reduction in nesting density at natal rookeries in the future and must be addressed to conserve the *C. mydas* populations in this region.

Between 1988 and 2022, the number of adult-sized *C. mydas* foraging at Green Island Reef declined, resulting in a population strongly biassed towards immature-sized turtles (< 85 cm)

(Table 1). This highly skewed age class abundance contrasts with population structures seen in other southern GBR genetic stock feeding sites (Bell et al. 2019). A similar population structure shift has not been recorded elsewhere within the northern GBR. On the contrary, between 1997 and 2006, when mean turtle size declined most dramatically at Green Island, six annual *C. mydas* population surveys were undertaken at the Howick Group, and no shift in demographic structure was recorded. As a proportion of the total catch, adult-sized *C. mydas* at the Howick Group varied from 37% to 73%. In contrast, at Green Island reef, for those years when more than 20 turtles were captured, the proportion of adult-sized turtles varied from 0% to 7%, with adult *C. mydas* only being caught on one of the last 10 sampling surveys (Table S3).

4.1 | Foraging Resource Availability

Despite seasonal and long-term fluctuations in abundance occurring over the course of the study, the turtles' primary food source (seagrass) expanded from 151.6 ha in 2003 to 316.6 ha in 2020 (McKenzie, Yoshida, and Unsworth 2014; McKenzie, Langlois, and Roelfsema 2022). This suggests that Green Island reef flat had sufficient seagrass resources to support mature *C. mydas* grazing pressure during the period their age-class declined.

The species of seagrass encountered on Green Island reef are all consumed opportunistically by *C. mydas* (Scott, York, and Rasheed 2020). Furthermore, the diet analysis of the subsample of captured turtles indicated the capability of *C. mydas* to forage opportunistically on both seagrass and/or algae highlighting their ability to switch diets (Prior, Booth, and Limpus 2016), in response to foraging habitat changes. However, during the period of seagrass monitoring since 2001, it appeared that seagrass resources were not a restricting factor.

4.2 | Anthropogenic Impacts

Multiple anthropogenic impacts, including pollution (Wilson and Verlis 2017), commercial fishing practices (Evans, Bax, and Smith 2017) and traditional take (Delisle et al. 2018) are all likely to have had a significant negative impact on marine turtle stocks.

4.2.1 | Pollution

The ingestion of plastic by marine turtles is now reported for all species (Duncan et al. 2021). Plastic presents a threat through ingestion, entanglement, the degradation of key habitats and wider ecosystem effects. However, due to feeding preferences of juveniles, they are thought to be more at risk, compared to adult turtles (Duncan et al. 2021). Although plastic ingestion has been linked to marine turtle morbidity and mortality, it would be difficult to accurately attribute the loss of adults, due to the lack of associated pathology (Bjorndal, Bolten, and Laguex 1994).

4.2.2 | Fishing

Significant management actions to mitigate the negative impacts of commercial fishing on turtle stocks in Queensland did not occur until implementation of the Trawl Management Plan in 2000 and the rezoning of the GBR Marine Park to increase protected areas in 2004 (Brodie and Waterhouse 2012). As part of these conservation strategies, Turtle Excluder Devices (TEDs) were mandated in commercial trawl operations, and no fishing zones were established around Green Island reef mitigating the likelihood of adult turtle bycatch mortality occurring adjacent to Green Island for several decades.

4.2.3 | Turtle Harvest

The loss of large turtles in a population has been reported for *C. mydas* in the Gulf of Manner, India, and for foraging hawksbill (*Eretmochelys imbricata*) turtles at a site in Cuba. In both cases, the cause of the disappearance was reported as an unsustainable take of especially adult females while nesting or foraging (Bhupathy and Saravanan 2006; Carrilo, Webb, and Manolis 1999). Excessive take has also been documented for other species of turtle, resulting in a substantial reduction in mean body size (Close and Seigal 1997; Eisemberg et al. 2011). This similarity in the pattern of population reduction of adult turtles indicates that traditional hunting may be contributing to the decline in the average body size of *C. mydas* on Green Island Reef. The observed hunting of large turtles by several witnesses supports this theory.

Indigenous peoples with recognized Native Title Rights can legitimately hunt *C. mydas* in Australia for communal, noncommercial, purposes (Native Title Act 1993 [Cth] [NTA]). Many Green Island Traditional Owners, the Guru Gulu Gunggandji, reside in the Aboriginal community of Yarrabah, located on the mainland approximately 19 km southwest of Green Island with a population of approximately 2505 (Australian Bureau of Statistics 2021).

Today, many people in the Gunggandjii and Yirrganydji groups claim historical or traditional ownership and use of the area, which includes hunting. A previous estimate of the number of *C. mydas* taken by residents of Yarrabah was approximately 260 turtles per year (Limpus 2007), many of which could have been taken from Green Island and the surrounding reefs.

Conflicts between the Green Island tourism industry and hunters (Marszalek 2017) resulted in the signing of a Traditional Use of Marine Resource Agreement (TUMRA) between the Gunggandjii and the Australian and Queensland governments (GBRMPA 2016). Under this agreement, no hunting of turtle or dugong is allowed in three key areas east of Cairns, including Green Island. This TUMRA was in place for 5 years (2016–2021), with one aim being that the Great Barrier Reef Marine Park Authority (GBRMPA) would provide compliance assistance to Gunggandji Traditional Owners to implement the agreement. The apparent increase in turtle size for the 2019 sampling records (Figure 4) provides hope that the moratorium on hunting is slowly allowing the turtle population on Green Island to recover.

5 | Climate Change

Climate change's current and future impact (see Jensen et al. 2019) will likely have profound implications for marine turtle survival (Hawkes et al. 2009). Rising temperatures affect the sex ratios of hatchlings because the incubation temperature of turtle nests determines the sex of the offspring (Jensen et al. 2018). Warmer sand temperatures, currently producing nearly 100% female hatchlings in the northern GBR (Jensen et al. 2018) in combination with associated sea-level rise, also threaten nesting beach productivity by eroding or flooding nests and reducing the available nesting habitat for turtles (Fuentes et al. 2010; Rivas et al. 2023). The decline observed in the *C. mydas* population near Green Island reef does not appear to be in the younger part of the population, as would be expected from the above scenarios. The decline of average CCL over the past 30 years is due to a loss of adults from the population, which rules out climate change as a contributing influence.

6 | Conclusion

Of all the factors that could have initially reduced and subsequently caused a low-density adult-sized *C. mydas* population found foraging on Green Island Reef, we consider selective hunting practices that specifically targeted adult-sized turtles to be the most likely. Although examples of overharvesting are limited, the recovery rate following a wholesale take of nesting turtles on Heron Island to supply a commercial turtle cannery suggests it may take a total closure of hunting in this region and possibly several more decades to reestablish a 'normal' age-class population structure of turtles on the Green Island Reef.

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Ethics Statement

The turtle research methodology used in this manuscript was conducted under the auspices of the Queensland Turtle Research Project's *Standard Operating Procedures* under the Great Barrier Reef Marine Park Permit G19/38535.1. The project received ethics approval clearance (EPA/2006/12/19) from the Queensland Government's Animal Ethics Committee.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

Once the manuscript is accepted for publication, we will deposit the data in the open access Queensland Government data portal, which can be accessed here: Dataset - Open Data Portal | Queensland Government. The data are deposited in the Environment, Tourism, Science and Innovation folder and searchable under the 'Green Island Green Turtle Project'.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.