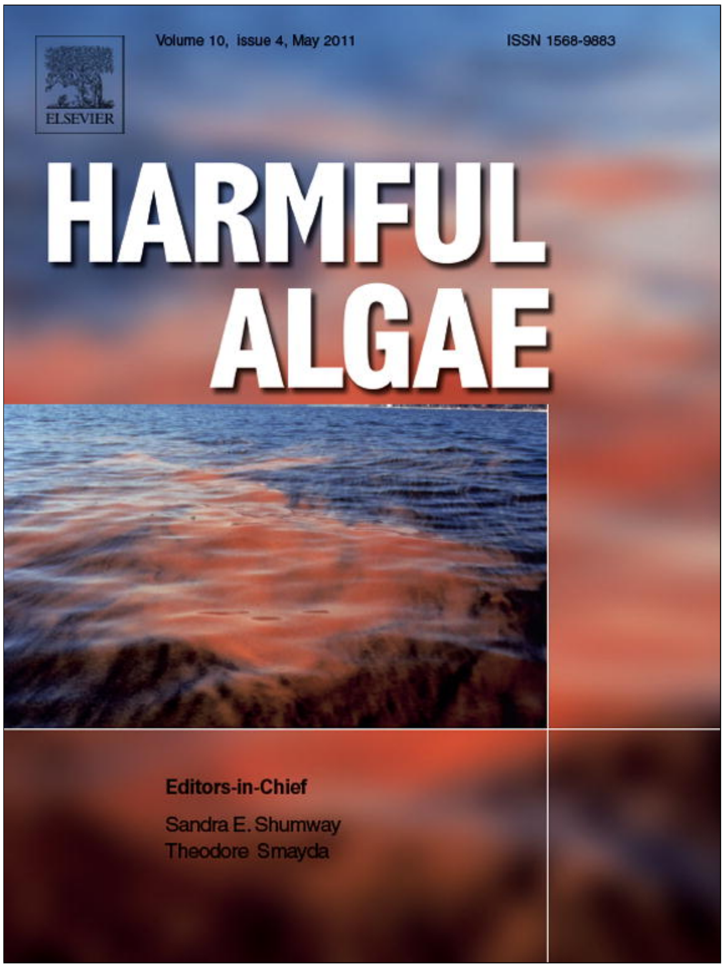
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Ciguatera poisoning in Rarotonga, southern Cook Islands

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ABSTRACT

The central Pacific Ocean has arguably more ciguatera poisoning than any other place on earth. Here we tested two competing hypotheses that outline the primary causes of ciguatera outbreaks: (1) the 'new surface hypothesis' and (2) the 'climate oscillation hypothesis'. Our findings indicated that in Rarotonga, from 1994 to 2010, the annual incidence of ciguatera poisoning ranged from 204 to 1,058 per 10,000 population per year. We found that the widest reefs of Rarotonga elicited the most cases of ciguatera poisoning, but found no relationship between ciguatera outbreaks and reef exposure. We also found strong correlations between cases of ciguatera poisoning and (i) the positive phase of the Pacific Decadal Oscillation, (ii) El Niño years, and (iii) periods with frequent disturbances. Yet, most disturbances occurred during the above-mentioned climate phases. This study links the two supposedly, mutually exclusive hypotheses. Moreover, as predicted by the 'climate oscillation hypothesis', the Pacific Ocean is now, in 2010, experiencing a negative phase of the Pacific Decadal Oscillation, and Rarotonga is reporting few cases of ciguatera poisoning.

1. Introduction

Ciguatera occurs in reef fishes and some invertebrates that have accumulated toxins produced by dinoflagellates (e.g., *Gambierdiscus toxicus*). These dinoflagellates are typically epiphytic to macroalgae (see Cruz-Rivera and Villareal, 2006). In humans, ciguatera poisoning occurs after ingesting the toxin. The symptoms include acute gastrointestinal and neurological disorder, and in extreme cases death. The global economy has increased the extent of ciguatera poisoning beyond the geographic range of natal fishes. For example, fishes caught in the tropical Pacific Ocean are frequently exported to Europe, Asia, and the USA (see van Dolah, 2000; Wong et al., 2005; Dickey and Plakas, 2010). Therefore, ciguatera poisoning affects at least 50,000–500,000 people per year worldwide (Fleming et al., 1998).

Ciguatoxins bioaccumulate as they pass through the marine food web, consequently concentrating in large, carnivorous fishes (Lewis, 2006). While *Gambierdiscus toxicus* has been implicated as the main dinoflagellate that causes ciguatera, other dinoflagellates have been also implicated (Faust, 1995; Holmes, 1998; Parsons and Preskitt, 2007; Rhodes et al., 2009). In addition, some studies have linked outbreaks of cyanobacteria to ciguatera-poisoning-like symptoms (Habekost et al., 1955; Dawson et al., 1955; Hashimoto et al., 1976; Laurent et al., 2008). Still, definitive links between ciguatera poisoning and the densities of microscopic life is tenuous, making predictions difficult.

Causes of ciguatera-poisoning outbreaks can be grouped into two broad categories. First, just as human-disease outbreaks are bio-indicators of ecosystem health (Spiegel and Yassi, 1997; Cook et al., 2004), ciguatera-poisoning outbreaks have been linked to degraded coral-reef ecosystems. The 'new surface hypothesis' (Randall, 1958) suggests that disturbances (i.e., cyclones, tsunamis, coral bleaching, *Acanthaster planci* outbreaks, dredging, boat channel construction, boat anchorage, and shipwrecks) provide freshly denuded surfaces for macroalgae to serve as substrate for toxic dinoflagellates (Cooper, 1964; Banner, 1976; Bagnis et al., 1988; Kohler and Kohler, 1992; Bagnis, 1994; Chinain et al., 1999). However, this rationale, although generally accepted, remains theoretical and correlations have been weak (see Kaly and Jones, 1994; Brusle et al., 1998; see Bienfang et al., 2008).

Alternatively, the 'climate oscillation hypothesis', suggests that ciguatera poisoning is primarily a consequence of specific phases of climate oscillations. For example, high, anomalous sea surface temperatures, associated with the inter-annual cycle of the El Niño Southern Oscillation (ENSO), have been frequently cited as the major factor inducing ciguatera-poisoning outbreaks (Tosteson et al., 1988; Epstein et al., 1993; Hales et al., 1999; Chateau-Degat et al., 2005; Tester et al., 2010). By contrast, Llewellyn (2009) proposed that the predicted scenario of high temperatures may instead suppress the incidence of ciguatera poisoning in tropical regions. Most recently, Rongo et al. (2009a) proposed that ciguatera events might be linked to larger-scale inter-decadal cycles, such as the Pacific Decadal Oscillation (PDO). Notably, El Niño tends to bring cool sea surface temperatures to the western and central Pacific

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whereas La Niña episodes promote warm temperatures and high rainfall. In the southern Cook Islands, conditions are cool during El Niño and warm during La Niña, while the opposite is observed in the northern Cook Islands (Baldi et al., 2009). There is also a coupling effect between ENSO and PDO; El Niño activities are frequent during the positive phase of PDO while La Niña activities are frequent during the negative phase (Verdon and Franks, 2006).

Ciguatera poisoning has been problematic in east Polynesia since the 1960s (Bagnis et al., 1979). To date, much of our understanding of ciguatera poisoning and its social impact stem from hospital records (e.g., Bagnis et al., 1979; Lewis, 1986; Chateau-Degat et al., 2007). Such results appear to underestimate the severity of ciguatera poisoning because only around 20% of cases are reported (Lewis, 1986; Dalzell, 1991). No ciguatoxic fishes were reported in the 1970s in Rarotonga, in the southern Cook Islands (Lewis, 1979; Fig. 1), where 67% of the Cook Islands population resides. Ciguatera poisoning only became a problem in the late 1980s to early 1990s. From 1993 to 2006, hospital cases showed an average incidence rate of 17.6 per 1,000 population per year (Rongo et al., 2009a). The impact of ciguatera poisoning in Rarotonga has had both economic and social repercussions. For example, Hajkowicz (2006) reported that the effect of ciguatera poisoning resulted in the avoidance of reef fishes in the diet of 71% of the resident population of Rarotonga. The loss of revenue has been estimated at NZD $534,000 annually. In addition, Rongo et al. (2009a) proposed that ciguatera poisoning may have contributed to the recent migration of Cook Islanders to New Zealand and Australia.

Therefore, predicting ciguatera poisoning is essential over both the short and long term, particularly in Rarotonga. We were able to test both the 'new surface hypothesis' and the 'climate oscillation hypothesis' by examining ciguatera poisoning in Rarotonga through time, assessing the types of fishes involved, and monitoring the reef condition. More specifically, we examined (1) the spatial distribution of ciguatera poisoning incidents, (2) the frequency of disturbances and its influence on ciguatera-poisoning events, and (3) whether there were any links between incidents of ciguatera poisoning and climatic cycles (i.e., ENSO and PDO). Rarotonga is an ideal location to examine the links between disturbances and climate oscillations because most of the human population practice subsistence fishing (Solomona et al., 2009), especially since alternative protein sources are expensive (Rongo et al., 2009a).

2. Materials and methods

2.1. Questionnaire survey

A questionnaire survey was conducted from December 2008 to January 2010. The survey targeted individuals aged 15 and over because adults generally avoid feeding reef fishes to children. Using the total resident population of 2006 (10,226), 626 individuals were interviewed at random (i.e., over 6%; whereas 5% seems sufficient for a representative sample according to Zar, 1999). Information collected included: (i) date of poisoning, (ii) identity of fish or invertebrate implicated in ciguatera poisoning (to the lowest taxonomic level possible), (iii) location where each fish was caught, (iv) symptoms experienced, (v) whether ciguatera poisoning was reported to health officials, and (vi) current fish consumption preference (including reef fishes considered 'safe').

Hospital records on cases of ciguatera poisoning were obtained from the Cook Islands Ministry of Health, which is responsible for compiling information from the only public hospital in Rarotonga. While the reporting of cases of ciguatera poisoning to the public health database began in the Cook Islands in 1991, it was not until 1993 that cases were separated by island. Therefore, cases reported before 1994 were excluded from the analyses because of reporting inconsistencies in the early stages of the transition (T. Iorangi, pers. comm.). Although information on ciguatera poisoning collected at the Rarotonga hospital only recorded the year of poisoning, and the island where it was experienced, for the purposes of this study this type of information was sufficient to examine the timing of ciguatera-poisoning outbreaks.

Although most people in Rarotonga are familiar with the clinical symptoms of ciguatera poisoning (i.e., gastrointestinal and neurological), in order to avoid recording other types of food poisoning, unusual symptoms were verified with those described in the literature. The symptomatic effects of ciguatera poisoning usually last from 3 to 6 months, and the consumption of fishes, regardless of type, within this period can trigger the recurrence of symptoms (see Baden et al., 1995). To avoid using recurring cases

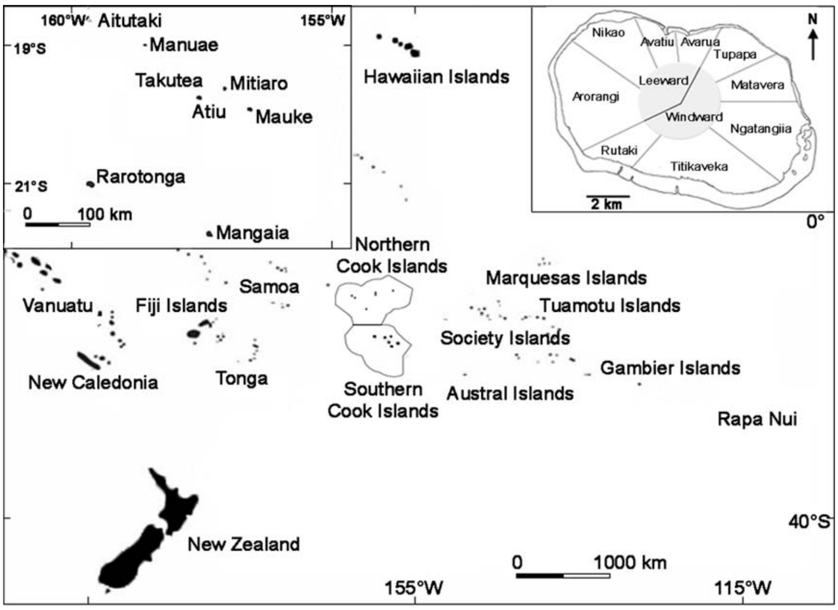

Fig. 1. Map of the Pacific Ocean with the Cook Islands' Exclusive Economic Zone delineated. Top left insert: Southern Cook Islands. Top right insert: Rarotonga (21°14′15″S 159°46′48″W) indicating villages in windward and leeward exposures..

of individuals with multiple poisonings, incidents were only recorded if they were at least 6 months apart.

2.2. Data preparation

Prior to any statistical analysis, sites where fishes were caught were categorized *a priori* as either (1) windward or leeward and (2) in accordance with the width of the lagoon (i.e., narrow, intermediate, and wide) (Fig. 1). The exposure of each site was based on the work of Thompson (1986). Windward villages were Tupapa, Matavera, Ngatangiia, Titikaveka, and Rutaki (which includes Kavera), and leeward villages were Arorangi, Nikao, Avatiu (which includes Panama), and Avarua (see Fig. 1). Using Google Earth, lagoons were categorized as narrow if $\leq$200 m, intermediate if >200 m and $\leq$400 m, and wide if >400 m.

Hospital and survey cases of ciguatera poisoning in Rarotonga were separated into three sampling periods based on the state of the reef prior to analysis: (i) 1989–2000 (although detailed reef surveys in Rarotonga commenced in 1994, prior years were included in this sampling period to encompass cases of ciguatera poisoning identified by the questionnaire survey), (ii) 2001–2005, and (iii) 2006–2009. Between 1989 and 2000, the shift towards a coral-depauperate state was largely the result of an outbreak by the coral-eating *Acanthaster planci* between 1995/96 and 2001 (Lyon, 2000), and to a lesser extent, bleaching events in 1991 and 1994 (Miller et al., 1994; Goreau and Hayes, 1995). The average hard-coral cover on the fore reef was reported to be around 33% in 2000 (Lyon, 2000). Between 2001 and 2005, most coral reefs around Rarotonga supported <5% hard coral cover; *A. planci* density decreased dramatically by 2003 (Lyon, 2003). In 2006, average coral cover was less than 2% on the fore reef (Rongo et al., 2006). In 2009, corals showed signs of recovery, with the fore reefs supporting on average 5% coral cover (Rongo et al., 2009b). Therefore, from 1989 to 2000 was defined as the '*Acanthaster planci*' period, from 2001 to 2005 was defined as the 'transitional' period, and from 2006 to 2009 was defined as the 'recovery' period.

The incidence rate of ciguatera poisoning was calculated annually from hospital data, and the size of the resident population of Rarotonga was taken from census data (conducted every 5 years). Census populations of 1991 (10,886), 1996 (10,337), 2001 (9,424), and 2006 (10,226) were used in calculations of incidence rates for the years 1994–1995, 1996–2000, 2001–2005, and 2006–2010, respectively. Incidence rates were reported per 10,000 population per year (see Tester et al., 2010). For our purposes, years prior to 1989 were considered as the 1970s ciguatera-poisoning event (1970–1988).

2.3. Data analysis

Only reef fish and marine invertebrate species involved in ten or more cases of ciguatera poisoning were included in the analysis. The data were square-root transformed to reduce the influence of sites and fishes with the highest incidents. A similarity matrix was constructed using the Bray-Curtis similarity coefficient. The spatial distribution of cases of ciguatera poisoning was examined using Multi-Dimensional Scaling (MDS). A Principal Component Analysis (PCA) was used to examine whether reef fishes and marine invertebrates, implicated in ciguatera poisoning, differed between sampling periods.

An Analysis of Similarity (ANOSIM) was used to test for differences between exposure (windward and leeward) and between lagoon width (narrow, intermediate, and wide). The results of ANOSIM generated R-values that provided a confidence limit on the degree of similarity: 0 (similar) to 1 (different) at a 0.05 significance level (see Clarke and Warwick, 1994). All multivariate analyses were conducted using Primer 6®.

Table 1
Cases of ciguatera poisoning on Rarotonga, by year, from the questionnaire survey. Annual incidence rates (rounded to nearest whole numbers) were calculated for each period using hospital cases of ciguatera poisoning requested from the Cook Islands Ministry of Health, and the total resident population in census years: 1994–1995 (10,886 in 1991), 1996–2000 (10,337 in 1996), 2001–2005 (9,424 in 2001), and 2006–2010 (10,226 in 2006). Actual incidence rates were calculated based on the 34% reporting determined from the survey. Data in brackets indicates total hospital cases of ciguatera poisoning from 1994 to 2010 (October).

Year	Survey cases	Annual incidence per 10,000 population per year	Actual annual incidence per 10,000 population per year	Period examined	Average incidence per 10,000 population per year	Actual average incidence per 10,000 population per year
1971	1					
1972	3					
1973	0					
1974	1					
1989	3					
1990	1					
1991	5					
1992	9					
1993	9					
1994	19	119	351			
1995	14	200	589			
1996	20	244	717			
1997	15	110	324			
1998	14	139	407			
1999	21	112	330			
2000	31	105	307	1994–2000	147 ± 20 SE	432 ± 60 SE
2001	31	132	387			
2002	24	128	378			
2003	20	179	527			
2004	28	360	1,058			
2005	37	341	1,002	2001–2005	228 ± 51 SE	670 ± 149 SE
2006	54	168	495			
2007	51	180	529			
2008	65	140	411			
2009	33	123	362			
2010		69	204	2006–2010	136 ± 19 SE	400 ± 57 SE
Total	509	[2,851]				

A Chi-square test was conducted to examine the relationship between the observed and expected average incidence of ciguatera poisoning for the three periods examined (1994–2000, 2001–2005, and 2006–2010). A nonparametric Spearman rank correlation was conducted using *Statistica 6*® to examine whether there was any links between hospital cases of ciguatera poisoning and climatic cycles; yearly averages, using monthly climate data, were calculated for the El Niño Southern Oscillation (ENSO) (using the Southern Oscillation Index [SOI]; National Oceanic and Atmospheric Administration Climate Prediction Center [NOAA-CPC]) and the Pacific Decadal Oscillation (PDO) (Joint Institute for the Study of the Atmosphere and Ocean). A negative value for the SOI and a positive value for the PDO would constitute a warm phase for the year examined, whereas a positive value for the SOI and a negative value for the PDO would constitute a cool phase for the year examined. Chateau-Degat et al. (2005) suggested that there was a lag period of 16–20 months between peaks of elevated sea water temperature and ciguatera poisoning appearing in the human population in French Polynesia, while Kaly and Jones (1994) showed a lag period of 1 year between peak densities of *G. toxicus* and peak toxicity in herbivorous reef fishes in Tuvalu. Therefore, we examined correlations between cases of ciguatera poisoning and climate data using 1-, 2-, and 3-year lags. A Chi-square test was also conducted to examine the relationship between observed and expected cases of ciguatera poisoning from hospital records (1994–2010) for the positive and negative phases of (i) SOI and (ii) PDO (with a 1-year lag period between cases of ciguatera poisoning and climate data). The hypothesis was that more cases of ciguatera poisoning occurred during El Niño and positive PDO years.

The frequency of disturbances was compared over time and in relation to ciguatera-poisoning events. Disturbances were included in the analysis only if they were observed to, or potentially have, impacted Rarotonga's reefs. They included: (1) cyclones (Baldi et al., 2009), (2) moderate to strong El Niño events (observed to cause coral bleaching on reefs of Rarotonga in recent years) (NOAA-CPC), and (3) *A. planci* outbreaks (Devaney and Randall, 1973; Dahl, 1980; Lyon, 2003). Subsequently, disturbances were combined within their respective ciguatera-poisoning event before conducting a Chi-square test to examine independence.

3. Results

There were 626 Cook Islanders interviewed, of which 326 (52%) had experienced ciguatera poisoning at least once in their lifetime. Because 42% of these individuals had multiple poisonings, 509 cases were reported and summarized in Table 1. Overall, about 88% of the cases of ciguatera poisoning were from Rarotonga, whereas 12% were from the outer islands (primarily Aitutaki and Atiu; we note that incidents from the outer islands and those with no date were excluded). We estimated that 34% of the cases of ciguatera poisoning were reported to health officials. We also estimated that only 4% of cases were treated at private clinics (listed as unreported in this survey because they were not reported to the public health database). From 1994 to 2010, the annual incidence rate of ciguatera poisoning varied from 69 to 360 per 10,000 population per year, which equates to an actual incidence rate of 204 to 1,058 per 10,000 population per year (calculated from the 34% reporting to hospitals that was determined by the survey; Table 1). We also estimated that the average incidence rate of ciguatera poisoning was significantly higher from 2001 to 2005 than in other periods ($\chi^2_2 = 29.7$, $p < 0.001$). The actual incidence rate for the three periods was estimated at 432, 670, and 400 per 10,000 population per year, respectively.

Table 2
Reef fishes and marine invertebrates involved in ciguatera poisoning that were reported in the questionnaire survey conducted between December 2008 and January 2010. Species are ranked from highest to lowest in incidents of poisoning (*N*). Although 678 incidents were recorded in the survey, only 526 incidents of fish and invertebrate poisonings were reported because of identification difficulties involved with some fish species.

Scientific name	Common name	Local name	*N*
REEF FISHES			
Scarus psittacus	Palenose parrotfish	Pakati	72
Caranx melampygus	Bluefin trevally	Titi'ara	62
Ctenochaetus striatus	Striped bristletooth	Maito	56
Crenimugil crenilabis	Fringelip mullet	Kanae	43
Chlorurus frontalis	Tan-faced parrotfish	U'u	32
Promethichthys prometheus	Snake mackerel	Mangā	31
Mulloides flavolineatus	Yellowstripe goatfish	Vete	27
Epinephalus tauvina	Greasy grouper	Pātuki taraava	26
Gymothorax javanicus	Giant moray	A'a pata	17
Lutjanus fulvus	Flametail snapper	Tangau	14
Cephalophalus argus	Peacock grouper	Pātuki roi	13
Epinephalus fasciatus	Blacktip grouper	'Ātea	10
Naso unicornis	Bluespine unicornfish	Ume	10
Sarcocentron spp.	Squirrelfish	Kūtā	10
Caranx ignobilis	Giant trevally	Urua	8
Lutjanus monostigma	Onespot snapper	Kiriva	8
Parapeneus spp.	Goatfish	Ka'uru	5
Caranx lugubris	Black trevally	Ru'i	4
Variola louti	Lyretail grouper	'Oka	4
Chaetodon auriga	Threadfin butterflyfish	Taputapu	3
Cheilinus undulates	Humphead wrasse; Napoleanfish	Maratea	3
Fistularia commersonii	Smooth cornetfish	Tātāvere	3
Kyphosus spp.	Rudderfish	Pipi	3
Lethrinus xanthochilus	Yellowlip emperor	'Īroa	3
Monotaxis grandoculus	Bigeye emperor	Mu	3
Rhinecanthus aculeatus	Picasso triggerfish	Kōkiri	3
Sargocentron spiniferum	Sabre squirrelfish	Taraki'i	3
Siganus argenteus	Forktail rabbitfish	Mōrava	3
Scarus altipinnis	Filament-finned parrotfish	'Aumauri	3
Acanthurus triostegus	Convict surgeonfish	Manini	2
Caranx sexfasciatus	Bigeye trevally	Kōmuri	2
Epinephalus merra	Dwarf spotted grouper	Pātuki marau	2
Selar crumenophthalmus	Bigeye scad	Ature	2
Sphyraena spp.	Barracuda	Ono	2
Synanceia verrucosa	Stonefish	No'u	2
Triaenodon obesus	Whitetip reef shark	Mangō maru	2
Tylosurus crocodilis crocodilis	Crocodile needlefish	Pāpā	2
Acanthurus xanthopterus	Yellowfin surgeonfish	Parangi	1
Chaenomugil leuciscus	Acute-jawed mullet	Ka'a	1
Cheilio inermis	Cigar wrasse	Kavakava	1
Diodon hystrix	Spot-fin porcupinefish	Tōtara	1
Epinephalus hexagonatus	Hexagon grouper	Pātuki paru	1
Gerres spp.	Mojarra	A'ore	1
Gymnosarda unicolor	Dogtooth tuna	Varu	1
Gymothorax undulates	Undulated moray	Matakiva	1
Hipposcarus longiceps	Pacific longnose parrotfish	Māmāringa	1
Liza vaigiensis	Yellowtail mullet	'Avake	1
Lutjanus bohar	Twinspot snapper; red snapper	Angamea	1
INVERTEBRATES			
Tridacna maxima	Giant clam	Pā'ua	12
Octopus scyanea	Big blue octopus	'Eke	2
Percnon spp.	Nimble spray crab	Pāpaka akau	2
Dendropoma maxima	Large worm shell	Ungakoa	1

3.1. The spatial distribution of fishes and invertebrates involved in ciguatera poisoning

From our survey, a total of 48 fish species, representing 24 families, were implicated in ciguatera poisoning (Table 2). Forty-eight percent of the cases were reported from individuals eating

Table 3
Fishes currently considered edible (from local knowledge) in Rarotonga. Information obtained from survey data, with most targeted fishes indicated (√).

Scientific name	Common name	Local name	Targeted
Acanthurus achilles	Achilles tang	'Iku toto	
Acanthurus guttatus	Whitespotted surgeonfish	Api	
Acanthurus leucopareius	Whitebar surgeonfish	Maito	
Acanthurus triostegus	Convict surgeonfish	Manini	
Carangoides orthogrammus	Thicklip trevally	Pava	
Caranx ignobilis	Giant trevally	Kōkōkina (juvenile)	
Caranx melampygus	Bluefin trevally	Kōkōkina (juvenile)	
Chaenomugil leuciscus	Acute-jawed mullet	Aua (juvenile)	
Chlororus frontalis	Tanface parrotfish	Pakati/U'u	
Cirrhitus pinnulatus	Stocky hawkfish	Pātuki toka	√
Epinephalus hexagonatus	Hexagon grouper	Pātuki paru	√
Kyphosus spp.	Rudderfish	Pipi	√
Mulloides flavolineatus	Yellowstripe goatfish	Kōma (juvenile)	
Mulloides vanicolensis	Yellowfin goatfish	Takua (juvenile)	
Myripristis spp.	Soldierfish	Kū	√
Naso lituratus	Orangespine unicornfish	Maipo	√
Naso unicornis	Bluespine unicornfish	Ume	√
Priacanthus hamrur	Crescent-tail bigeye	Kūpā	√
Siganus argenteus	Forktail rabbitfish	Mōrava	√
Siganus spinus	Scribbled rabbitfish	Maemae	√

the fishes *Scarus psittacus*, *Caranx melampygus*, *Ctenochaetus striatus*, *Crenimugil crenilabis*, *Chlorurus frontalis*, *Promethichthys prometheus*, *Mulloides flavolineatus*, and *Epinephalus tauvina.* The giant clam (*Tridacna maxima*) was the most common invertebrate implicated in ciguatera poisoning, all of which were collected from the Titikaveka lagoon area. The survey also identified targeted reef fishes considered safe for consumption by locals; they included *Cirrhitus pinnulatus*, *Epinephalus hexagonatus*, *Kyphosus* spp., *Myripristis* spp., *Naso lituratus*, *Naso unicornis*, *Priacanthus hamrur*, *Siganus argenteus*, and *Siganus spinus* (Table 3).

There was no significant difference in the number of cases of ciguatera poisoning between windward and leeward lagoons. By contrast, narrow lagoons (i.e., Avatiu, Avarua, Tupapa, and Matavera) showed significantly fewer cases (Fig. 2; Table 4), whereas the widest lagoons (i.e., Ngatangiia, Titikaveka, and Nikao) showed the most cases of ciguatera poisoning (ANOSIM, $R = 0.796$, $p = 0.029$). However, narrow and intermediate lagoons ($R = 0.429$, $p = 0.267$), and intermediate and wide lagoons ($R = 0.333$, $p = 0.200$) showed no significant differences in the number of ciguatera poisoning cases.

3.2. Vector shifts of ciguatera poisoning

Ordination analyses, using PCA, clearly separated the three periods examined (Fig. 3). Vector plots suggested that herbivore poisonings, primarily from *Ctenochaetus* spp. and to a lesser extent *Naso unicornis*, were most common between 1989 and 2000. Between 2001 and 2005, poisonings by *Epinephalus fasciatus* were common, but most interesting were the increased poisonings by soft-bottom benthic invertivores (i.e., *Mulloides flavolineatus* and *Crenimugil crenilabis*) and filter feeders (i.e., *Tridacna maxima*) during this period (Table 5). Between 2006 and 2009, carnivorous fishes (i.e., *Promethichthys prometheus*, *Gymnothorax* spp., *Caranx melampygus*, and *Epinephalus tauvina*) and herbivorous fishes (i.e., *Scarus psittacus* and *Chlorurus frontalis*) caused most poisonings.

Table 4
Analysis of Similarity pair-wise test between exposures (leeward vs. windward) and among the three categories of lagoon widths (narrow, intermediate, and wide).

Exposure & Lagoon width	*R*-Statistic value	Significance level ($p < 0.05$)
Exposure		
Leeward and Windward	0.175	0.889
Lagoon width		
Narrow and Wide	0.796	0.029*
Narrow and Intermediate	0.429	0.267
Wide and Intermediate	0.333	0.200

* Indicates significant difference at $p < 0.05$.

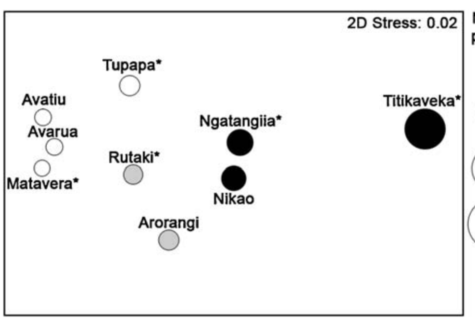

Fig. 2. An ordination of sites using total cases of ciguatera poisoning that was derived from the questionnaire survey conducted between December 2008 and January 2010. Sites with narrow lagoons (open circles), intermediate lagoons (grey circles), and wide lagoons (black circles) were examined. Asterisk (*) indicates windward sites. The size of the bubbles reflect relative cases of ciguatera poisoning.

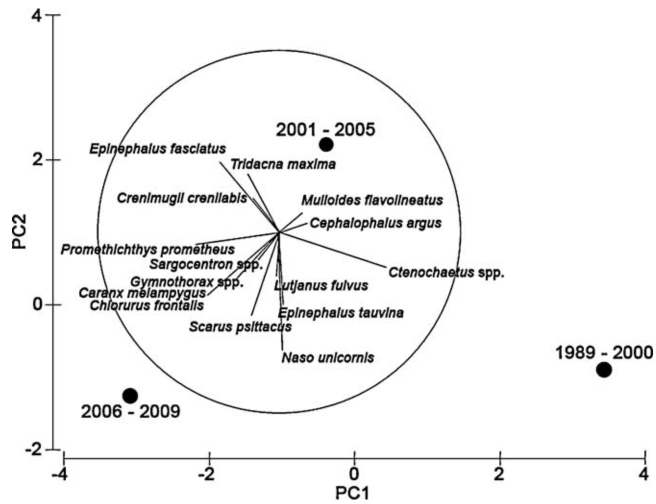

Fig. 3. Principal Component Analysis for reef fish and marine invertebrate species involved in ten or more cases of ciguatera poisoning in Rarotonga in relation to the three periods (1989–2000, 2001–2005, and 2006–2009).

Table 5
Eigenvalues and eigenvectors of Principle Component Analysis of ciguatoxic reef fishes and marine invertebrates in Rarotonga for the three periods examined (1989–2000, 2001–2005, and 2006–2009).

PC	Eigenvalues	% Variation	Cum. %variation
Eigenvalues			
1	10.8	68.1	68.1
2	5.06	31.9	100

SPECIES	PC1	PC2
Eigenvectors		
Sargocentron spp.	−0.117	−0.124
Naso unicornis	0.002	−0.546
Epinephalus fasciatus	−0.329	0.324
Tridacna maxima	−0.172	0.270
Cephalophalus argus	0.155	0.041
Lutjanus fulvus	0.009	−0.129
Gymnothorax spp.	−0.015	−0.213
Epinephalus tauvina	0.024	−0.336
Mulloides flavolineatus	0.126	0.091
Promethichthys prometheus	−0.464	−0.055
Chlorurus frontalis	−0.391	−0.290
Crenimugil crenilabis	−0.144	0.155
Ctenochaetus spp.	0.587	−0.158
Caranx melampygus	−0.231	−0.209
Scarus psittacus	−0.148	−0.380

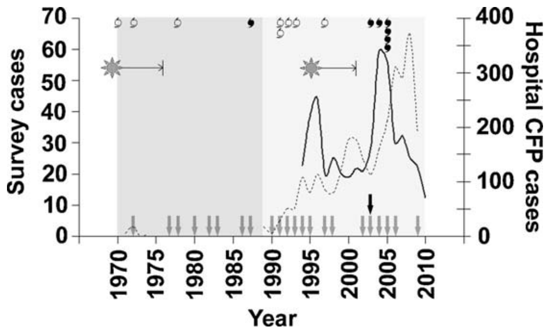

Fig. 4. Cases of ciguatera poisoning from Rarotonga's hospital data (solid line) and the questionnaire survey (dashed line). Shaded regions indicate the two ciguatera-poisoning events identified by the survey (1970–1988; 1989–2009). Sun-shapes and horizontal arrows indicate periods of two major *Acanthaster planci* outbreaks: 1969–1976 (Devaney and Randall, 1973; Dahl, 1980), and 1995/96–2001 (Lyon, 2003; Rongo et al., 2006). Also shown are the years of Category 3 or less cyclones (open) (1970, 1972, 1978, 1991, 1992, 1993, 1997) and Categories 4 and 5 cyclones (black) (1987, 2003, 2004, 2005) impacted Rarotonga (Baldi et al., 2009). Grey arrows indicate El Niño years taken from the National Oceanic and Atmospheric Administration Climate Prediction Center (1972, 1977, 1978, 1980, 1982–1983, 1986–1987, 1990–1995, 1997–1998, 2002–2006, 2009). Black arrow marks the Titikaveka Irritant Syndrome in 2003.

3.3. Climate cycles, disturbances, and ciguatera poisoning

Cases of ciguatera poisoning reported by the Rarotonga hospital showed two peaks, around 1995–1996 and 2004–2005 (Fig. 4). Using a 1-year lag period between hospital cases of ciguatera poisoning and climate data from 1994 to 2010, there were clear correlations between the number of cases of ciguatera poisoning and the Southern Oscillation Index (SOI) and the Pacific Decadal Oscillation (PDO) cycle (Fig. 5; Table 6). Although the SOI was also significant using a 2-year lag period, the PDO was only significant from 1998 to 2008; no significant correlations were evident when we forced a 3-year lag period (see Table 6). In addition, the positive and negative phases of the SOI and the PDO showed different numbers of ciguatera-poisoning cases ($\chi_1^2 = 537$, $p < 0.001$; $\chi_1^2 = 371$, $p < 0.001$, respectively), with greater numbers of ciguatera poisoning during El Niño and positive PDO years (Table 7). The survey also identified two periods of ciguatera poisoning: (1) in the early 1970s and (2) from 1989 onwards. There were significantly more disturbances during 1989–2009 ($\chi_1^2 = 4$, $p = 0.037$), which corresponded to a higher incidence of ciguatera poisoning (Table 8; see Fig. 4).

4. Discussion

Ciguatera poisoning has been problematic in Rarotonga since the late 1980s, and according to our survey a minor ciguatera event also occurred in the early 1970s, which eluded previous studies

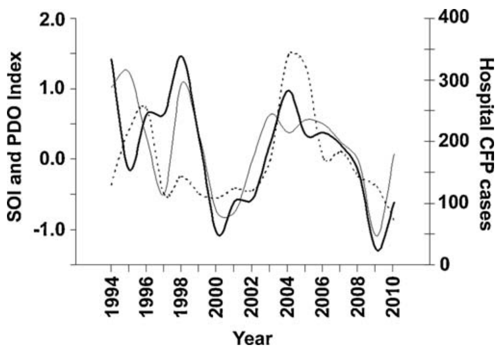

Fig. 5. Hospital cases of ciguatera poisoning in Rarotonga (dashed line) from 1994 to 2010 and the Pacific Decadal Oscillation (PDO; thick line) and the Southern Oscillation Index (SOI; thin line) from 1993 to 2009. Data for PDO and SOI were taken from 1993 to 2009 to implement a 1-year lag period for cases of ciguatera poisoning. The SOI index data were multiplied by −1 for comparative purposes.

(e.g., Lewis, 1979, 1986). Consequently, we estimated that 52% (∼5,318) of residents have experienced ciguatera poisoning at least once in their lifetime. Such estimates are considerably higher than the 1.8–2.5% reported from two Pacific island communities in Australia (Gillespie et al., 1985). Such differences may reflect island populations relying more on marine resources than their Australian counterparts. We also note that 34% of cases of ciguatera poisoning were reported to health officials, which was higher than the 20% generally assumed in the literature (e.g., Lewis, 1986;

Table 6
Nonparametric analysis using Spearman rank order correlations between cases of ciguatera poisoning (from Rarotonga hospital records) and the Pacific Decadal Oscillation (PDO) and the Southern Oscillation Index (SOI).

Variables	*N* (years)	Spearman *R*	Significance level ($p < 0.05$)
One-year lag			
SOI (1993–2009) & Ciguatera cases (1994–2010)	17	−0.565	0.018*
Two-year lag			
SOI (1992–2008) & Ciguatera cases (1994–2010)	17	−0.487	0.048*
PDO (1992–2008) & Ciguatera cases (1994–2010)	17	0.246	0.340
PDO (1998–2008) & Ciguatera cases (1994–2010)	11	0.691	0.019*
Three-year lag			
SOI (1991–2007) & Ciguatera cases (1994–2010)	17	−0.185	0.477
PDO (1991–2007) & Ciguatera cases (1994–2010)	17	0.105	0.687

* Significance at $p < 0.05$ were noted.

Table 7
Chi-square tests comparing Rarotonga's hospital cases of ciguatera poisoning from 1994 to 2010 during the positive and negative phases of the Southern Oscillation Index (SOI; La Niña and El Niño, respectively) and the Pacific Decadal Oscillation (PDO) (using a 1-year lag period between cases of ciguatera poisoning and climate data). Results showed significant differences between SOI ($\chi_1^2 = 537$, $p < 0.001$) and PDO ($\chi_1^2 = 371$, $p < 0.001$) phases.

	El Nino	La Nina
Observed ciguatera cases	2,044	807
	Positive PDO	Negative PDO
Observed ciguatera cases	1,940	911

Table 8
Chi-square test comparing natural disturbances between the two ciguatera-poisoning events identified by the survey conducted between December 2008 and January 2010. Results showed a significant difference between the periods ($\chi_1^2 = 4$, $p < 0.037$).

Disturbances	1970–1988	1989–2009
Cyclone frequency	4	11
El Niño frequency	8	14
Acanthaster planci outbreak	1	1
Total	13	26

Dalzell, 1991; Chinain et al., 2009). Yet, over 80% of individuals surveyed in 2010 were consuming reef fish in Rarotonga, contrary to Hajkowicz's (2006) findings who reported 71% of the human population avoided reef fishes during a peak period of ciguatera poisoning. However, 96% of these individuals were selective in their consumption habits, and were particularly cautious of fishing locality (i.e., the Titikaveka area is generally avoided). In fact, 22% of these individuals only consumed reef fishes imported from islands in the northern Cook Islands (i.e., Manihiki and Penryhn), Palmerston in the southern group, and to a lesser extent from other islands in the southern group, (i.e., Mangaia, Mauke, and Mitiaro) that were thought to be largely free of ciguatera.

Our survey indicated that in 2010, 75% of the resident population of Rarotonga was consuming more pelagic species than previously, even though the pelagic fishes were more expensive than reef fishes. This shift was also reported by Solomona et al. (2009) in 2001. We also estimated that 14% of individuals surveyed limited themselves to pelagic species, and only 1% completely excluded reef fish from their diet. Residents, especially those whom have had multiple poisonings over their lifetime, have accepted ciguatera poisoning as part of a reef-fish diet, and continue to consume reef fishes despite the risk. This risk-taking attitude is common among island populations, as was noted some 40 years ago in the Gilbert Islands (Cooper, 1964), and recently on Raivavae of the Austral Islands in French Polynesia (Chinain et al., 2009).

Between 1973 and 1983, the Cook Islands was considered a ciguatera anomaly, with a very low reported incidence of ciguatera (Lewis, 1986). However, over the past 16 years, Rarotonga has been a hotspot for ciguatera poisoning, alongside the neighboring islands of French Polynesia. For example, the average incidence rate from 1996 to 2006 in 14 countries of the Caribbean ranged from 58.6 to 0.003 per 10,000 population per year (see Tester et al., 2010), compared with 183 per 10,000 population per year in Rarotonga for the same period. On Raivavae, the average incidence from 2007 to 2008 was 140 per 10,000 population per year (Chinain et al., 2009), while our estimate for Rarotonga, for the same period, was 160 per 10,000 population per year. However, we noted that the actual annual incidence from 1994 to 2010 in Rarotonga ranged from 204 to 1,058 per 10,000 population per year, which suggests that ciguatera poisoning can affect up to 11% of the population in a year.

4.1. Spatial distribution of ciguatera poisoning

Most cases of ciguatera poisoning were recorded from the Titikaveka section of the Rarotonga lagoon, yet our results showed that the degree of reef exposure did not contribute to the incidents of poisoning. Other studies examining *G. toxicus* densities with regard to wind and wave exposure have shown mixed results. Some studies found higher densities on windward reefs (e.g., Kaly et al., 1991), whereas other studies found higher densities on leeward reefs (e.g., Carlson, 1984; Taylor, 1985). By contrast, the width of the lagoon played a significant role in our study. More incidents of ciguatera poisoning were reported from individuals that consumed fishes caught in wide lagoons. It could be argued that narrow reefs support fewer fishes in general, and therefore the probability of not having ciguatera poisoning from fishes caught in narrow lagoons is simply a statistical artifact. We suggest that these results are not artifacts, especially since most fishing activities on Rarotonga are carried out on the narrow reef locations, which are considered safe by residents. We propose that wide lagoons tend to have reduced water circulation that is ideal for the establishment of harmful algae populations. Constant water movement on fore reefs, and frequent flushing across narrow lagoons, might reduce the densities of toxic dinoflagellates. Turbulent conditions affect dinoflagellate growth (Smayda, 1997), and certainly reduce their densities on macroalgal hosts (see Nakahara et al., 1996). The present study may explain why reef fishes resident in surf zones (i.e., *Epinephalus hexagonatus* and *Cirrhitus pinnulatus*) were considered 'safe' in this survey (see Table 3), regardless of where they were caught. Toxicity studies of resident fish species in different reef zones may help clarify the relationship between toxic fishes and the distribution of ciguatera-causing dinoflagellates.

4.2. Vector shifts of ciguatera poisoning

Between 1989 and 2009, there were clear shifts in the types of reef fishes involved in ciguatera poisoning in Rarotonga. Firstly, between 1989 and 2000, herbivores (i.e., acanthurids) were primarily implicated in ciguatera poisoning. By contrast, ciguatera poisoning from benthic invertivores (e.g., goatfish and mullet) and filter feeders (i.e., *Tridacna* spp.) became more evident from 2001 to 2005. Herbivores (i.e., scarids) and notably carnivores were most frequently implicated in ciguatera poisoning from 2006 to 2009 (see Fig. 3). Similar temporal shifts in ciguatera-poisoning vectors were noted in the Gambier Islands over a 20-year period (Bagnis et al., 1988). Herbivores were mainly responsible for ciguatera poisoning from 1967 to 1975, whereas herbivores and carnivores were equally involved from 1976 to 1982, and carnivores were mostly involved from 1983 to 1987—with incidents of ciguatera poisoning declining after 1987 (Bagnis et al., 1988). Our findings in Rarotonga agree with the Gambier Islands study that ironically showed an increased involvement of carnivorous fishes, in cases of ciguatera poisoning, before incidences in these same fishes declined.

Ciguatera poisoning was historically thought of as a 'carnivore problem' (Lewis, 2006). Alternatively we suggest that if carnivores are the *only* fishes involved in ciguatera poisoning, then the general incidence of ciguatera should decline in the near future. Our argument stems from the fact that at low dinoflagellate densities, toxins only bioaccumulate in large carnivores, and remain low in other fishes. The shift in ciguatera-poisoning vectors from herbivorous fishes to benthic invertivores (i.e., *Mulloides*), between 2001 and 2005, is also of interest. Although the transfer of toxins from specific dinoflagellates into the food web is mainly recognized through herbivorous fishes (Randall, 1958), there is also some evidence that crustaceans transfer toxins (Lewis et al., 1994), which may explain the involvement of soft-bottom invertivores in ciguatera poisoning in Rarotonga. In addition, our survey indicated that parrotfishes and surgeonfishes are currently edible (in 2010) despite local knowledge of their involvement in many past cases of ciguatera poisoning.

Nevertheless, human adjustment to ciguatera poisoning is certainly evident in Rarotonga. Residents know that juveniles of known toxic species (e.g., *Mulloides flavolineatus* and *Caranx* spp.) are safe to eat. However, 'low-risk' fishes are now being targeted (Table 3). For example, *Kyphosus* spp., arguably the safest reef fish recorded in the survey, may be experiencing heavy fishing pressure. Moreover, *Naso unicornis* is in 2010 considered to be among the safest reef fish in Rarotonga, while in Raivavae (of the Austral Islands) this species is considered high-risk (Chinain et al., 2009).

4.3. Algal community and ciguatera-causing dinoflagellates

Macroalgal cover has been uncommon on the fore reefs of Rarotonga in recent years. However, macroalgae such as *Sargassum*, *Dictyota*, *Jania*, and *Galaxaura*, known to host high densities of *G. toxicus* (e.g., Shimizu et al., 1982; Anderson and Lobel, 1987; Cruz-Rivera and Villareal, 2006), were common in the lagoon for most of the 1990s (T. Rongo, pers. obs.). Although we lack detailed information on algal species composition through time, the shift in (ciguatera-poisoning) vectors may reflect the dominant algae present on the reefs during these periods. For example, macroalgae grazers (e.g., *Ctenochaetus* spp.) and browsers (e.g., *Naso unicornis* and *Acanthurus* spp.) were prolific in the 1990s, and were potential ciguatera-poisoning vectors. Yet, in the 2000s, the ciguatera-poisoning vector shifted to benthic invertivores (e.g., goatfish and mullet) that feed among sediments and turf algae. Notably, turf algae were the dominant substrata during the 2000s (Rongo et al., 2006, 2009b), and have been reported to host among the highest densities of *G. toxicus* (see Cruz-Rivera and Villareal, 2006). Because macroalgae and turf grazers such as *Ctenochaetus* spp. (the most implicated herbivore in ciguatera poisoning in the early sampling) were avoided for much of the 2000s, it is difficult to distinguish whether this early shift in vectors is the result of algal community shifts or human adjustment to ciguatera poisoning.

Shifts in algal communities may also result from changes in herbivorous fish densities. We note that the average density of herbivorous fishes in Rarotonga increased from a mean of 0.26 ($\pm$0.07 standard error) fishes m^{-2} in 1999 (Ponia et al., 1999) to a mean of 1.40 ($\pm$0.63 standard error) fishes m^{-2} in 2006 (Rongo et al., 2006). Such increases in herbivorous fish densities may have been a consequence of coral loss (Wilson et al., 2006) (because of *Acanthaster planci* outbreaks) and declines in fishing pressure, because of ciguatera poisoning. Changes in fishing pressure may have also affected trophic cascades. For example, high fishing pressure in the early 1990s, particularly on herbivorous fishes (see Ponia et al., 1999), could have led to macroalgae dominance. But, when ciguatera poisoning became chronic in the early 2000s, fishing pressure declined and herbivorous fish densities would have increased; such increases would have reduced macroalgae cover, as was apparent for much of the 2000s.

Increased herbivorous fish densities may have also facilitated the transfer of ciguatoxins through the food web (an idea mentioned by Bagnis et al., 1988 in their study in French Polynesia), causing the 'flare-up' of ciguatera poisoning after 2003. Increased herbivore densities may also explain the increased number of fish species involved in ciguatera poisoning, especially during the transitional period, where some 40 species were involved compared with 33 species involved during both the *A. planci* period and the recovery period. Notably, the average density of herbivorous fishes decreased from a mean of 1.40 ($\pm$0.63 standard error) fishes m^{-2} in 2006 to 0.51 ($\pm$0.20 standard error) fishes m^{-2} in 2009 (Rongo et al., 2009a,b), which also corresponded with a decline in hospital cases of ciguatera poisoning in Rarotonga.

The shift to turf algae in the 2000s may have altered the assemblages of toxic dinoflagellates in Rarotonga. For example, the Cook Islands Ministry of Marine Resources (who has occasionally monitored *G. toxicus* density since the early 1990s in Rarotonga) indicated that *G. toxicus* density decreased from 1000 s per 100 g (wet algae) in the 1990s to 10 s per 100 g in recent years; interestingly, the decline in *G. toxicus* density coincided with an increase in the density of *Ostreopsis* spp. (T. Turua, pers. comm.). Temporal shifts in dinoflagellate populations have been noted in the literature to relate with allelopathic interactions. For example, Taylor and Gustavson (1985) found an inverse relationship between abundance of *G. toxicus* and *Ostreopsis* spp. Similarly, Carlson (1984) found inverse correlations between *G. toxicus* and both *Prorocentrum rhathymum* and *Amphidinium carterae* abundance.

The involvement of other dinoflagellates in ciguatera poisoning are poorly understood (e.g., Lewis et al., 1998; Chinain et al., 1999), although Ballantine et al. (1985) and Carlson and Tindall (1985) suggested that *Ostreopsis* spp. and *Prorocentrum lima* were important contributors to ciguatera poisoning in the Caribbean. High densities of a *Prorocentrum* species, reported in Titikaveka lagoon from sediments in 2003 (Skinner et al., unpublished data), coincided with the sharp increase in cases of ciguatera poisoning in 2004 and 2005 (see Fig. 4), involving benthic invertivores (e.g., goatfish and mullet) and filter feeders (i.e., *Tridacna* spp.) (Fig. 3). In addition, Rhodes et al. (2009) in their 2007 study in Rarotonga, found that dinoflagellates from the genus *Amphidinium*, *Ostreopsis*, and *Coolia* were toxic, and could co-occur with *Gambierdiscus*. However, further research is needed to examine if, in fact, other dinoflagellates are responsible for ciguatera poisoning in Rarotonga.

Laurent et al. (2008) identified cyanobacteria from the genus *Hydrocoleum* as the major cause of ciguatera-like poisonings in New Caledonia. Several authors have also identified the involvement of *Lyngbya* in fish poisonings in the Pacific Ocean (e.g., Habekost et al., 1955; Dawson et al., 1955; Hashimoto et al., 1976; see also Osbourne et al., 2001). Although *Lyngbya* was implicated as the major source of eye and throat irritation and dermatitis in 2003 (called the Titikaveka Irritant Syndrome) (Eason and Hope, 2005), their involvement in fish and *Tridacna* poisonings in Rarotonga cannot be ruled out. Van Dolah (2000) also suggested that factors such as nutrient loading, anomalous drought and storm events, and El Niño events can cause toxic cyanobacteria outbreaks. Therefore, it may be difficult to differentiate the effects of *Lyngbya* (the most common substrate on sandy lagoons around Rarotonga; T. Rongo, pers. obs.) uptake, from co-occurring ciguatoxic dinoflagellates in both reef fishes and marine invertebrates.

4.4. Climate cycles, disturbances, and ciguatera poisoning

Irrespective of the specific dinoflagellates responsible for ciguatera poisoning in Rarotonga, our study showed that both the Pacific Decadal Oscillation (PDO) and El Niño Southern Oscillation (ENSO) were significantly related to the incidence of ciguatera poisoning (Table 6). From 1994 to 2010, cases of ciguatera poisoning were correlated with both the positive phase of the PDO and with El Niño years (Table 7). Hales et al. (1999) showed a significant correlation between SST associated with El Niño events and incidents of ciguatera poisoning in Rarotonga from 1973 to 1994. Still, we note that ciguatera poisoning did not become a problem in Rarotonga until the late 1980s to the early

1990s. We also note that such major differences in previous reporting of ciguatera poisoning involved the erroneous pooling of data that was obtained from the South Pacific Epidemiological and Health Information Service (SPEHIS) database. Cases of ciguatera poisoning reported to the SPEHIS database combined the southern and the northern Cook Islands, where climate conditions are different (see Baldi et al., 2009; Rongo et al., 2009a).

Randall's 'new surface hypothesis' suggests that new surfaces become available usually after the loss of corals (Cooper, 1964; Bagnis et al., 1985, 1992; Bagnis, 1994; Kohler and Kohler, 1992; Chinain et al., 1999). Our data from Rarotonga 'cautiously' agrees with this hypothesis, but we add that the state of the reef is less important than the generation of denuded carbonate substrate by disturbances. Disturbances provide fresh, primary successional surfaces for algae to host opportunistic populations of ciguatoxic dinoflagellates. Indeed, turf algae has been the most dominant substrate on Rarotonga's reefs for over 10 years, and coral cover has been well below 10% at most localities (Rongo et al., 2006, 2009b). Yet, even in this algal-dominated system, outbreaks of ciguatera poisoning only occurred after several major disturbances to the reef.

The scale, intensity, and frequency of disturbances may play an important role in ciguatera-poisoning outbreaks (Kaly and Jones, 1994). While small-scale, local disturbances might have little impact on ciguatera outbreaks (Kaly and Jones, 1994), large-scale disturbances, such as cyclones and coral bleaching events, seem to be a prerequisite of ciguatera-poisoning outbreaks (de Sylva, 1994; Chinain et al., 1999). We note that the frequency of reef disturbances and the incidence of ciguatera poisoning were low in Rarotonga between the 1970s and 1980s. By contrast, the cyclone frequency was high between 2001 and 2005, coinciding with highest incidences of ciguatera poisoning on record (Fig. 4). The frequency of cyclones in the southern Cook Islands is influenced by the mean location of the South Pacific Convergence Zone that shifts in relation to the ENSO and PDO cycles (Folland et al., 2002); cyclone frequency in the southern Cook Islands is greater during El Niño and positive PDO years (de Scally, 2008; Baldi et al., 2009).

Although the disturbance frequency was significantly different between the two ciguatera-poisoning events (Table 8; Fig. 4), we could not verify which disturbance, if any, was responsible for the ciguatera-poisoning outbreak. Rarotonga tends to be *cool and dry* during El Niño events (Baldi et al., 2009), with lower than average sea level (Pacific Country Report: Cook Islands, 2003). Under such conditions, the reef edge is frequently exposed and lagoonal exchange with the adjacent ocean is reduced; such conditions might favor toxic algal blooms. Alternatively, La Niña years that are *warm and wet* might be unfavorable for ciguatoxic dinoflagellates, which prefer higher salinity waters (Yasumoto et al., 1980; Taylor, 1985; Bomber et al., 1988). While elevated temperatures have been identified as the cause of ciguatera poisoning (e.g., Hales et al., 1999), outbreaks of ciguatera poisoning in Rarotonga have historically occurred when temperatures were cooler. Therefore, warmer temperatures may not be driving ciguatera poisoning in Rarotonga.

Although we are far from identifying one primary cause of ciguatera poisoning, this study suggests, however, that the combination of climate conditions during El Niño events and a positive phase of the PDO, cause a high frequency of storm activity that facilitates ciguatera poisoning in Rarotonga. Perhaps further examining this geographic region will aid our understanding of these factors, and assist in the development of predictive models of ciguatera-poisoning outbreaks.

In conclusion, our results agree with Rongo et al. (2009a), who proposed that ciguatera-poisoning events in the southern Cook Islands were linked to the positive phase of the Pacific Decadal Oscillation, a phase that is also accompanied by a high frequency of El Niño events and cyclones (see de Scally, 2008). The recent decline in the incidence of ciguatera poisoning in Rarotonga, and the continued decline through 2010, coincided with the recent shift of the Pacific Decadal Oscillation into a negative phase. We therefore predict that ciguatera poisoning will continue to decline in Rarotonga over the next decade, during this negative phase.

Over the longer term, however, ocean temperatures are predicted to increase by up to 3 °C by the end of 2100 (Hegerl et al., 2007). If ciguatera poisoning is related to high, anomalous water temperatures, then the incidence of ciguatera poisoning is also expected to rise (de Sylva, 1994; see Papua New Guinea and Pacific Islands Country Unit, 2000). However, Llewellyn (2009) proposed that a warmer climate would suppress ciguatera poisoning. The predicted warmer climate in the future will, however, increase the intensity of cyclones but potentially reduce their frequency (Hegerl et al., 2007). Therefore, and according to our findings, we should see fewer outbreaks of ciguatera poisoning during periods of low cyclone activity. Yet, with the pole-ward migration of species predicted under climate change (e.g., Parmesan and Yohe, 2003), we may simply see the impact of toxic dinoflagellates shifting into higher latitudes, and potentially impacting fisheries in those regions (Rhodes et al., 2009).

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