

Climate and Vegetation Changes at Coringa-Herald National Nature Reserve, Coral Sea Islands, Australia¹

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Abstract: Climatic changes at Coringa-Herald National Nature Reserve (CHNNR) in the last 82 yr include a 0.7°C rise in mean minimum winter temperatures and increases in drought duration and frequency. Between 1991 and 2002, a plague of the scale insects *Pulvinaria urbicola* (Cockerell), together with attendant ants destroyed *Pisonia grandis* R.Br. rain forest at South-West Coringa Islet. Scale insect damage of *P. grandis* has also been recorded at North-East Herald Cay. This study explored the reasons for vegetation dieback during current climate. Woody species such as *Argusia argentea* (L.) Heine, *Cordia subcordata* Lam., and the grasses *Lepturus repens* (G. Forst.) R.Br. and *Stenotaphrum micranthum* (Desv.) C. E. Hubb. have also declined at CHNNR. *Ximения americana* L. and *Digitaria ctenantha* (F. Muell.) Hughes were found to be locally extinct. Dieback of forests results in reduction of canopy-breeding seabirds and burrowing shearwaters (*Puffinus pacificus* [Gmelin]). Dieback species were replaced by the shrub *Abutilon albescens* Miq. and/or fleshy herbaceous plants such as *Achyranthes aspera* L., *Boerhavia albiflora* Fosberg, *Ipomoea micrantha* Roem. & Schult, *Portulaca oleracea* L., and *Tribulus cistoides* L. Increasing duration of droughts and increased temperatures, together with damage caused by exotic insect pests, appear to be the key drivers of the current vegetation changes.

CORINGA-HERALD National Nature Reserve (CHNNR), located about 400 km east of Cairns, forms part of the northern Australian Coral Sea Islands Territory. It includes three groups of coral sand cays and islets: the Herald Cays, the Coringa Islets, and the Magdeline Cays. The total vegetated area is around

125 ha (Batianoff et al. 2009). The remoteness of CHNNR has safeguarded this area from high levels of human disturbance. It was declared a National Nature Reserve in 1982 because it provides critical breeding habitat for colonies of large numbers of nesting seabirds and turtles (Environment Australia 2001). According to Batianoff et al. (2009) these northern Coral Sea islands provide important wildlife linkages between the Great Barrier Reef and the Melanesian islands by the influence of the prevailing currents and winds (Figure 1).

CHNNR terrestrial ecosystems have been studied by various biologists including Heatwole (1979), Hicks and Hinchey (1984), Hicks (1985), Hinchey and Stokes (1987), Donaldson (1994), the Royal Geographical Society of Queensland (2001), Batianoff (2001a,b; unpubl. data), Smith and Papacek (2001a,b,c), Smith et al. (2004), Freebairn (2006a,c,d, 2007), Batianoff and Naylor (2007), Greenslade and Farrow (2007), Batianoff et al. (2009), Greenslade (2008); J. Hicks, unpubl. data; and P. O'Neill, J.A.O.,

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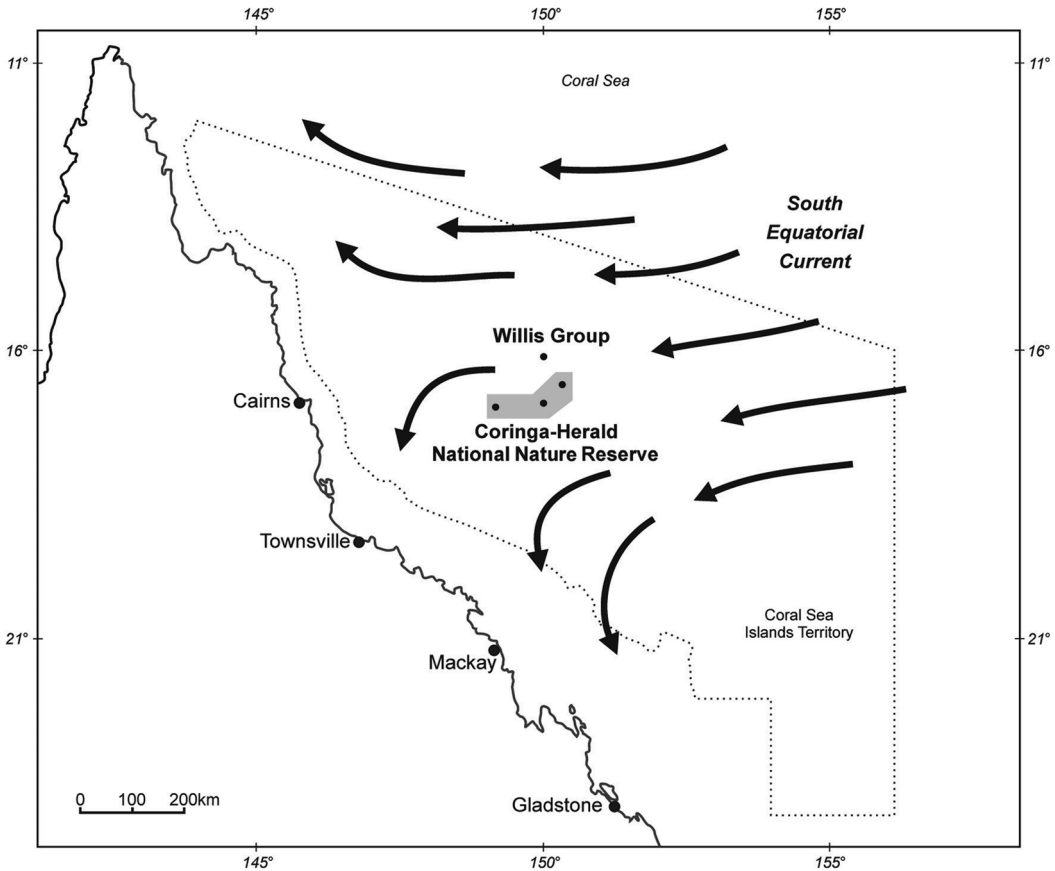


FIGURE 1. Location of Coringa-Herald National Nature Reserve and Willis Island within the Coral Sea Islands Territory showing major currents (arrows) operating within the region (derived from Smith [1994]).

and R. Elder, unpubl. data. These surveys revealed that dieback of forests dominated by *Pisonia grandis*, *Argusia argentea*, and *Cordia subcordata* has occurred over the last few decades along with several other environmental changes. The dieback of *P. grandis* due to the effects of plague *Pulvinaria urbicola* scale infestations during the last two decades has been widely reported at CHNNR (Smith and Papacek 2001*a,b,c*, 2004, Smith et al. 2004, Batianoff and Naylor 2007, Greenslade 2008; P. O'Neill, J.A.O., and R. Elder, unpubl. data) and elsewhere across the Pacific and Indian oceans (Handler et al. 2007; A. Kay, J.A.O., R. Elder, and K. Bell, unpubl. data; J.A.O., K. L. Bell, and R. J. Elder, unpubl. data; J.A.O., R. J. Elder, R. M. Charles, J. R. Platten, and K. L. Bell, unpubl. data).

Severe defoliation of *P. grandis* from the larvae of the Australian native hawkmoth, *Hippotion velox*, has also been reported at CHNNR (Donaldson 1994, Freebairn 2006*a,c,d*, 2007). The dieback of *A. argentea* and *C. subcordata* is more localized to the CHNNR.

At CHNNR, insect damage to *P. grandis* and to a lesser extent to *C. subcordata* is well documented (Smith and Papacek 2001*a*, Smith et al. 2004, Freebairn 2006*a,c,d*, 2007). The use of the Australian native *Cryptolaemus montrouzieri* (Molsant) as a biological control agent to control the population of plague *P. urbicola* scale infestations at North-East Herald Cay was outlined by Smith and Papacek (2001*a,b,c*) and Freebairn (2006*a,c,d*, 2007). Freebairn (2006*a,c,d*, 2007) also discussed the

use of two species of egg parasitoid (*Trichogramma pretiosum* and *Trichogramma carverae*) to control *H. velox* outbreaks. Continuous drought conditions have been shown to induce the dieback of trees in northern Australian savannas (Fensham et al. 2009). Droughts have also been linked to explosions in pest insect plagues and tree dieback by White (1969). Greenslade (2008) suggested that *Pisonia* dieback at CHNNR may have been exacerbated by drought stress and nutrient stress due to reduced populations of nesting seabirds. Both of these stressors may be indirectly caused by climatic trends. The aim of this study is to examine the role of climatic trends over the last two decades in relation to vegetation changes and with reference to dependent wildlife at CHNNR in accordance with international climate knowledge (Intergovernmental Panel on Climate Change 2007).

On coral cays, vegetation structure and individual plants are important in providing “functional” diversity (Walker and Salt 2006). Some of these functions include landform development, habitat diversity for wildlife (including invertebrates [Greenslade and Farrow 2007, Greenslade 2008]), and nesting/roosting and/or shelter for a large number of seabirds (Batianoff 2001a,b, Batianoff and Cornelius 2005). In particular, *A. argentea* and *P. grandis* plant communities at CHNNR form critical nesting habitat for the arboreal and ground-nesting seabirds. More than half of the breeding seabirds at CHNNR utilize these woody species. For example, the Lesser and Great Frigatebirds (*Fregata ariel*, *F. minor*), the Common Noddy (*Anous stolidus*), the Black Noddy (*Anous minutus*), the Red-footed Booby (*Sula sula*), as well as the burrowing Wedge-tailed Shearwater (*Puffinus pacificus*) are the most common breeding seabirds utilizing *Pisonia* and *Argusia* stands (Batianoff 2001a).

MATERIALS AND METHODS

Climate Trends

Climate data for the region were extrapolated and are presented in this paper from 82 yr of records from the Bureau of Meteorology sta-

tion no. 200283 located at Willis Island (16° 29' S, 149° 07' E) (Batianoff and Naylor 2007, Batianoff et al. 2009, Bureau of Meteorology 2008a). Rainfall trends were determined by comparing annual rainfall with the long-term mean. “Dry” years were those when rainfall was below the long-term mean, as well as those within 50 mm above the mean (Batianoff and Naylor 2007). The duration of droughts was extrapolated by the number of consecutive “dry” years (Batianoff et al. 2009). However, Willis Island rainfall data from September 2004 to October 2006 are not available due to the upgrading of monitoring equipment (Bureau of Meteorology 2008a). We were not able to fill this gap from other sources because there is no offshore rainfall data available for nearby areas. As a result, some of the interpretation of droughts during that period is based on observations of vegetation condition (Batianoff et al. 2009).

A drought index ($D_{m,y}$) was calculated for each month m of each year y based on the actual annual rainfall for 3 yr before every month less the expected (long-term average) rainfall for that period, divided by the mean annual rainfall:

$$D_{m,y} = \sum_{i=y-x+1}^y \frac{a_{m,y} - A}{A},$$

where $a_{m,y}$ is the summed monthly rainfall over the previous 12 months of year y (that is, from month m to $m-11$), and A is the long-term annual average rainfall (for Willis Island it was averaged over 82 yr). Three years was chosen as deficit period, x . The Foley Index of drought (Foley 1957) was chosen over other drought indices for reasons outlined in Fensham et al. (2009). That is, it can be calculated for any period and only requires rainfall data.

Trends in ambient temperatures were determined by comparing the mean maximum and minimum temperatures for summer and winter months for each decade since 1925–1934 (Batianoff et al. 2009). Unless otherwise stated, all references to seasons are to the standard Australian allocation of months (i.e., summer is from December to February,

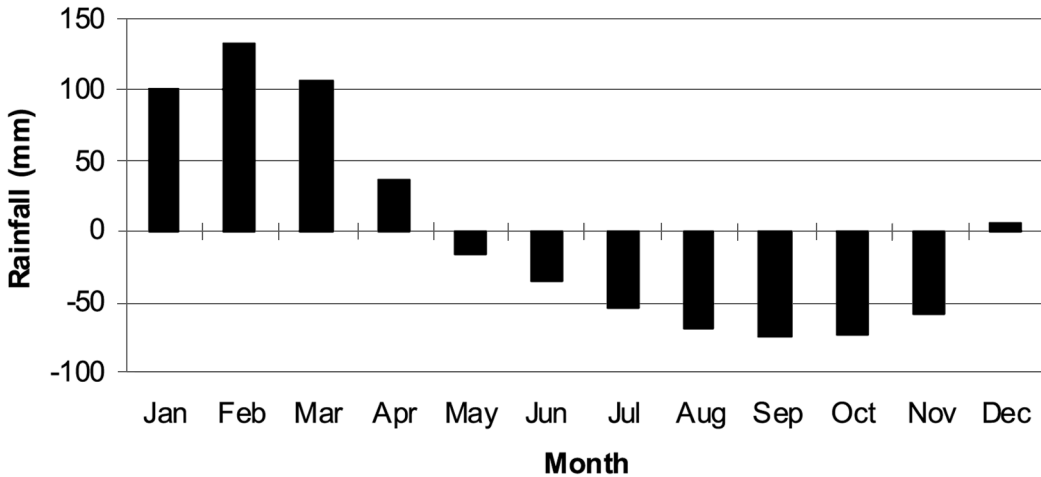


FIGURE 2. Mean monthly rainfall anomalies from the long-term monthly mean of 93 mm at Willis Island, 1922–2007 (Batianoff and Naylor 2007, Bureau of Meteorology 2008a).

autumn from March to May, winter from June to August, and spring from September to November. Data were accurate up to 31 July 2008.

Vegetation Changes

The effects of climate on vegetation and associated wildlife at CHNNR are based on three terrestrial surveys from 1997 (Batianoff 2001b) and 2006–2007 (Batianoff and Naylor 2007, Batianoff et al. 2009). Dieback and other species turnover were identified using all available data including unpublished reports (J. Hicks; P. O’Neill, J.A.O., and R. Elder) and personal communications with CHNNR monitoring team leader, M. Hallam (1997, 2006, and 2007). These additional data include the extensive reporting on the scale insect infestations of *Pisonia grandis* by Smith and Papacek and Smith et al. (2001–2004), Freebairn (2006–2007), and Green-slade (2008). Other predicted climate change impacts are based on an extensive literature review of tropical regions. The localized effects of climate change to seabirds on coral cays follow data presented by Batianoff and Cornelius (2005), Turner and Batianoff (2007), and Baker et al. (2006, 2008).

RESULTS AND DISCUSSION

Current Climatic Trends

The northern Coral Sea Islands Territory study area experiences a dry, tropical maritime climate. According to Mueller-Dombois and Fosberg (1998), Willis Island, with a mean annual rainfall of 1,115 mm is a “dry island.” According to Batianoff and Naylor (2007) the current climate patterns include 3–4 months of “wet” and 8–9 months of “dry” seasons as well as longer-term patterns of “dry” and “wet” years (Figures 2 and 3). During the summer period, tropical depressions in the zone of convergence between “monsoon” and “trade” winds bring storms and cyclonic weather to tropical areas (Farrow 1984, Myers and Kent 2005). Stoddart and Walsh (1992) reported that the variability in rainfall at Willis Island has longer cycles of 12–30 yr between wet and dry “epochs.” It is generally assumed that these fluctuations correspond with the El Niño–Southern Oscillation (ENSO) and the Inter-decadal Pacific Oscillation (Myers and Kent 2005). Prolonged drought stress in the northern Coral Sea region occurs as part of the ENSO phenomenon. The ENSO phenomenon brings periods of higher rainfall to Australia during the “La Niña” phase and drier conditions

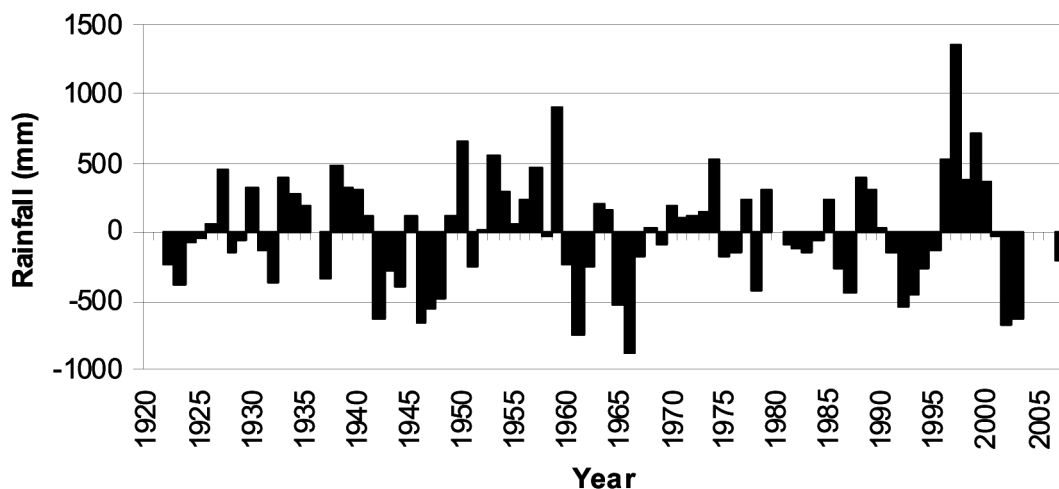


FIGURE 3. Annual rainfall at Willis Island, 1922–2007 expressed as deviation from the long-term mean of 1,115 mm (Batianoff and Naylor 2007, Bureau of Meteorology 2008a).

due to lower rainfall during the “El Niño” phase (Suppiah et al. 2001, Cai et al. 2003).

Summaries of global current climate trends (Intergovernmental Panel on Climate Change 2007) and regional trends (Suppiah et al. 2001, Cai et al. 2003, Hughes 2003, Johnson and Marshall 2007) provide generalized overviews of climate trends based on the modeling of large climate data sets from many locations. In this study, climate trends are discussed at a local level using one set of climate data depicting a relatively small tropical offshore region (Batianoff and Naylor 2007). These data are used to provide probable explanation for some of the current *Pisonia grandis* forest and other vascular plants dieback at CHNRR (Batianoff et al. 2009).

Ambient Temperatures and Humidity

Ambient temperatures at Willis Island show little variability throughout the year (Table 1). Temperatures are between 25.7°C and 30.7°C during the warmest month of January, with the highest temperature recorded at 35.2°C. The temperature for the coolest month of July ranges between 21.9°C and 26.0°C, with the lowest temperature recorded at 18.9°C. The stable and relatively high annual temperatures above 20°C result in high

evaporation. Evaporation data are not available for Willis Island; however, Farrow (1984) reported that January and April were the only months at Willis Island where rainfall exceeded evaporation. Relative humidity at Willis Island is between 68% and 81% (Table 1).

According to Batianoff et al. (2009), between 1925–1934 and 1995–2004 the mean minimum temperature at Willis Island has increased +0.7°C, though the mean maximum temperature has remained relatively unchanged (Table 2). Temperature increase has been most pronounced for the mean minimum winter temperature (+0.7°C), with the summer minimum temperature increasing +0.6°C. The mean winter maximum and summer maximum temperatures have remained relatively unchanged (Table 2). The temperature rises recorded in the study area are lower than for the Australian mainland. According to Suppiah et al. (2001), the mean minimum temperature increase on the mainland has been +0.96°C and maximum temperature +0.56°C in a similar period.

Rainfall

According to Batianoff and Naylor (2007) most of the annual rainfall at Willis Island

TABLE 1
Willis Island Monthly Temperature, Rainfall, Humidity, and Cyclone Data: 1921–2007
(Bureau of Meteorology 2008a,b)

Month	Mean Min. Temp. (°C)	Mean Max. Temp. (°C)	Min. Terrestrial Temp. (°C)	Mean Monthly Rainfall (mm)	Total Rain Days	% of Total Rain Days	Relative Humidity (%)	% of Cyclones per Month
Jan.	25.7	30.7	24.3	193	10	13	74–78	26
Feb.	25.6	30.4	24.3	226	12	15	78–81	30
Mar.	25.4	29.8	24.1	198	12	15	77–79	19
Apr.	24.7	28.8	23.4	129	9	12	75–77	11
May	23.8	27.7	22.4	76	7	9	74–75	1
June	22.6	26.5	21.1	58	6	8	72–73	0
July	21.9	26.0	20.4	38	5	6	69–71	0
Aug.	21.9	26.3	20.2	24	3	4	68–71	0
Sep.	22.5	27.3	20.9	17	2	3	68–70	0
Oct.	23.6	28.7	22.1	19	2	3	69–71	1
Nov.	24.7	29.9	23.2	36	3	4	69–72	0
Dec.	25.4	30.6	23.9	99	6	8	71–74	12
Average	24.0	28.6	22.5	93	7	—	72–74	—

TABLE 2
Ambient Temperature Increases for Willis Island: 1921–2007 (Bureau of Meteorology 2008a)

Period	Annual		Summer		Winter	
	Mean Maximum Temp. (°C)	Mean Minimum Temp. (°C)	Mean Maximum Temp. (°C)	Mean Minimum Temp. (°C)	Mean Maximum Temp. (°C)	Mean Minimum Temp. (°C)
1925–1934	28.4	23.7	30.2	25.4	26.1	21.8
1935–1944	28.7	23.8	30.8	25.6	26.3	21.8
1945–1954	28.6	24.0	30.5	25.5	26.5	22.1
1955–1964	28.8	23.9	30.7	25.4	26.6	22.1
1965–1974	28.6	23.9	30.6	25.4	26.3	22.2
1975–1984	28.6	24.0	30.7	25.6	26.1	22.1
1985–1994	28.4	24.2	30.5	25.8	26.1	22.4
1995–2004	28.7	24.4	30.8	26.0	26.2	22.5
Trend	No trend	+0.7°C Strong trend	No trend	+0.6°C Moderate trend	No trend	+0.7°C Strong trend

occurs during the “wet” summer/autumn months of January–April (Figure 2). The corresponding “dry” winter/spring months are from May to November, with December recording close to the mean monthly rainfall. The rainfall pattern at Willis Island over the past 87 yr indicates fluctuating “wet” and “dry” periods that may last for 1–7 yr (Figure 3). The annual rainfall has varied from 241

mm in 1966 (22% of the mean) to 2,482 mm in 1997 (223%) (Bureau of Meteorology 2008a). Extreme high and low rainfall fluctuations are generally assumed to correspond with ENSO and/or the Inter-decadal Pacific Oscillation (Cai et al. 2003, Lough 2007). However, lower (negative) Southern Oscillation Index values (El Niño) do not always relate to lower rainfall events in tropical areas.

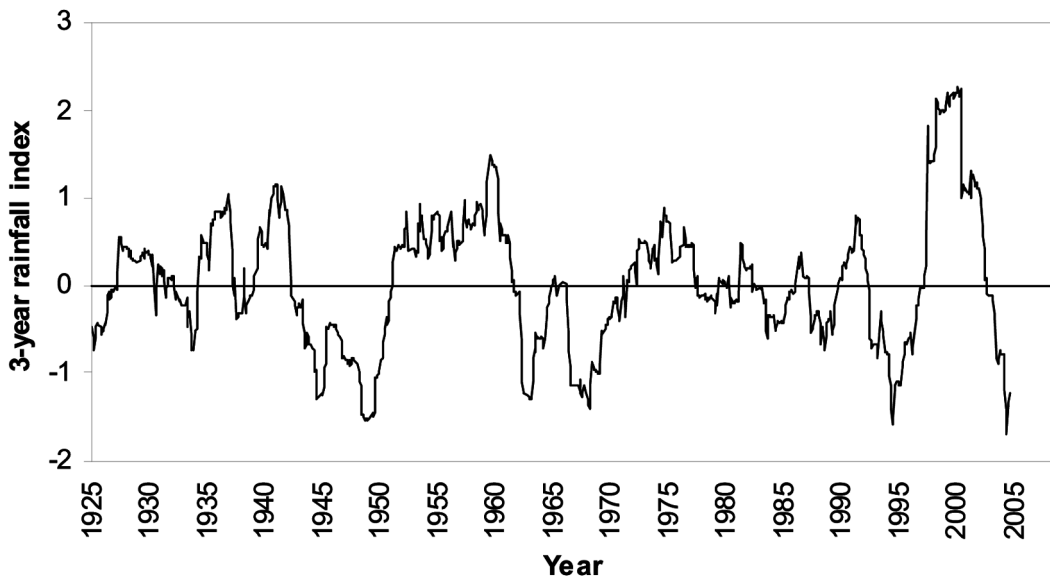


FIGURE 4. Drought index for Willis Island, 1922–2004. Pest insect population outbreaks at South-West Coringa Islet in 1991 and North-East Herald Cay in 2001. Note that data after 2004 are not available from Willis Island (Bureau of Meteorology 2008a, Foley 1957).

The strong El Niño event of 1997 was also the year that Willis Island recorded its highest rainfall (Bureau of Meteorology 2008a) due to the influence of Tropical Cyclone Justin (Bureau of Meteorology 2008b).

According to Batianoff et al. (2009) there is no discernible long-term trend of changing rainfall amounts; nevertheless, it is noted that since 1990, 8 yr have recorded rainfall > 100 mm below the long-term mean (Figure 3). Five years have recorded above-average (median) amounts, and 1 yr has been neutral. Rainfall amounts during the 2004–2006 period were not available for this analysis. However, regular visits by field personnel and examination of oblique air-monitoring photographs indicated continuing “dry” conditions during those periods (Batianoff et al. 2009). Therefore, we suggest that a continuous rainfall deficit occurred from 2001 to 2006. In the 18 yr since 1990, 11 (61%) have been “dry,” 8 (33%) have been “wet,” and 1 yr (6%) has been “neutral.”

The conditions of variable patterns of rainfall at CHNNR are illustrated in Figures 3 and 4. These data show that monthly rain-

fall deficits (Figure 3) correspond with yearly rainfall deficits (Figure 4) because they highlight the same periods of droughts and intense rainfall. The Drought Index (Foley 1957) of monthly rainfall shows that the most severe drought on record occurred around 1994, followed by the second most intense rainfall period from 1997 to 2001 (Figure 4). The worst period of rainfall deficit on record occurred after 2001.

A moderate trend toward an increase in drought frequency and duration is apparent at Willis Island (Figure 5). The duration of droughts at CHNNR in the last 43 yr has increased from a mean of 2.3 yr (1922–1964) to 3.3 yr (1965–2007) (Batianoff et al. 2009). Myers and Kent (2005) predicted drier conditions for northern tropical Australia due to projected climate change. However, rainfall projections for the northeastern Queensland region are uncertain due to the limited ability to model the Australian summer monsoon (Lough 2007). However, longer durations of drought conditions are expected due to the higher temperatures increasing evaporative losses (Lough 2007).

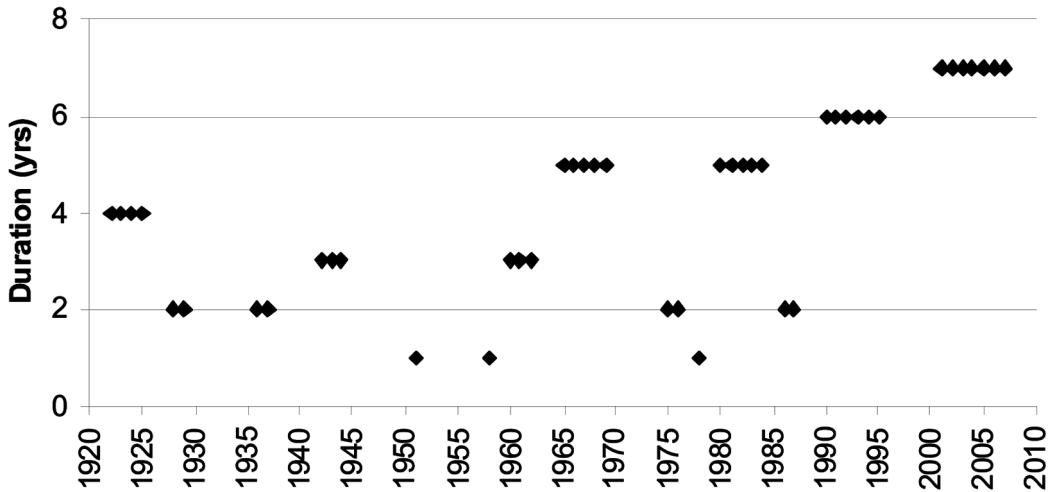


FIGURE 5. Duration of contiguous “drought” years at Willis Island, 1922–2007 (Bureau of Meteorology 2008a). Note that drought years are considered as those years where rainfall is below 1,165 mm (i.e., long-term mean +50 mm). Data from Willis Island for 2004–2006 are not available; however visiting personnel and oblique aerial photographs taken during that period indicate continued drought conditions (Batianoff et al. 2009).

Wind and Cyclones

At CHNRR, windy conditions occur 95% of the year (Bureau of Meteorology 2008a). The southeasterly “trade” winds make up 80% and the remaining 20% are known as surges of northerly and northwesterly “monsoon” winds during January to March (Farrow 1984, Bureau of Meteorology 2008a). A total of 184 cyclones has passed within 400 km of Willis Island since 1906, with about 18 per decade and almost two cyclones per year (Bureau of Meteorology 2008b). The usual cyclone season at Willis Island is from December to April (Table 2), with February and January the most cyclonically active months. However, singular cyclone events have been recorded within 400 km of Willis Island in October and May (Bureau of Meteorology 2008b).

Long-term cyclone data for Willis Island indicate a gradual increase in the frequency of cyclones (Batianoff et al. 2009). However, the number of cyclones recorded at CHNRR for the last decade has been below average (Bureau of Meteorology 2008b). Hughes (2003) reported a slight decline in the number of tropical cyclones in the Australian re-

gion but a slight increase in the intensity of cyclones. Under these conditions, a greater frequency of high winds and storm surges (Hughes 2003) will have direct and indirect effects on coral cay ecosystems.

Vegetation Changes

During the last two decades, widespread dieback of trees and annual/short-lived grass species across all islands in CHNRR portrays images of environmental stress. As one dominant species dies out, the open areas are rapidly colonized by other species, thus maintaining vegetation cover and adaptive behavior (Walker and Salt 2006). The replacement of *Pisonia grandis* littoral rain forest with herbland/shrubland communities on South-West Coringa is part of fluctuating seral stages of coral cay vegetation (Kepler and Kepler 1994, Batianoff 2001b). The replacement of the rain forest’s structure by low-growing vegetation produces a “normal” vegetation pattern, be it at a reduced functional diversity.

This species turnover at CHNRR and/or fluctuating change of dominance between different species is an ongoing natural phe-

nomenon that provides coral cay resilience to changing environmental conditions (Heatwole 1979, Kepler and Kepler 1994, Specht and Specht 1999, Batianoff 2001*b*). However, in the last 20 yr it has been more pronounced. Under current climatic trends, the indigenous and exotic insect species (unintentional human introductions) benefit from warmer winters during wet years by the onset of earlier breeding and consequently higher populations that damage vegetation (Smith et al. 2004). The increased duration and frequency of drier seasons favor more drought-tolerant and/or disturbance-adapted plant species. The increasing plant populations are the shrub *Abutilon albescens*, the perennial grass *Sporobolus virginicus*, and fleshy plants such as *Achyranthes aspera*, *Boerhavia albiflora*, *Ipomoea micrantha*, *Plumbago zeylanica*, and *Portulaca oleracea* (Table 3).

Dieback of *Pisonia grandis*

High populations of the soft scale insect *Pulvinaria urbicola* and damage to *P. grandis* forest were first recorded at South-West Coringa Islet (SW Coringa) in 1991 (Donaldson 1994, Smith and Papacek 2001*a,b,c*). Severe defoliation by the larvae of the hawkmoth *Hippotion velox* was also observed on the *P. grandis* immediately before the soft scale outbreak (M. Hallam, pers. comm., 1997). According to rainfall data presented in Figures 3 and 4, this high population of insects followed two to four “wet” periods of above-average rainfall. By 1994, up to 90% of the trees were defoliated due to scale damage (Donaldson 1994). According to P. O’Neill, J.A.O., and R. Elder (unpubl. data) and M. Hallam (pers. comm., 2007), during the 1997 surveys *P. grandis* dieback was estimated at between 80% and 90%. Examination of climate data indicates that loss of *Pisonia* forests from 1993 to 1998 occurred during a transition from a “wet” period to a “drought.” This infestation caused the loss of the entire 10 ha of *Pisonia* forest at SW Coringa by 2002 (Figure 6) (Smith et al. 2004).

Scale insects have also been detected on North-East Herald Cay (NE Herald) since 1994 (Donaldson 1994; P. O’Neill, J.A.O.,

and R. Elder, unpubl. data). During June 1997, when rainfall was above average (Figures 3 and 4), G.N.B. recorded healthy woody vegetation and a relatively low population of scale insect on *Pisonia* foliage (Batianoff 2001*b*). The same relatively low levels were also reported in December of the same year by P. O’Neill, J.A.O., and R. Elder (unpubl. data). However, the scale insect population at NE Herald had increased by the time of the 2000–2001 surveys there (Smith and Papacek 2001*a*). Between the March and December 2001 surveys of NE Herald, a 150-fold increase from 1997 values of the scale population was recorded (Smith and Papacek 2001*a,b*). The 2001–2002 period also coincides with a transition from a “high” rainfall phase to a “drought” phase (Figures 3 and 4). Repeated *H. velox* hawkmoth outbreaks at NE Herald have caused stress to *Pisonia* trees and the defoliation of foliage (Smith and Papacek 2004). It was recorded in 2006 and in May 2007 that the larvae of *H. velox* had caused approximately 72% defoliation (Freebairn 2006*c,d*, 2007). In November 2006 G.N.B. (unpubl. data) also recorded Australian native giant grasshopper (*Valanga irregularis*) grazing *Pisonia* foliage. It is important to note that there was no dieback of *Pisonia* trees directly linked to the Australian native insects (i.e., giant grasshopper and/or hawkmoth defoliation [G.N.B., unpubl. data; J.A.O., pers. obs., May 2007]).

In July 2001, biological control of scale insects was implemented at NE Herald using the Australian native predatory ladybird (*Cryptolaemus montrouzieri*) and three species of soft scale parasitoid wasps (Smith and Papacek 2001*b,c*). In subsequent years, *C. montrouzieri* and hawkmoth larvae egg parasitoids were also released at NE Herald (Freebairn 2006*c,d*, 2007). According to Freebairn (2007), major losses of *Pisonia* dieback were avoided due to the effective biological control program of predatory ladybird releases and the baiting of exotic ants (*Tetramorium bicarinatum* and *Monomorium* sp.). The exotic ants facilitate the spread of *Pulvinaria urbicola* through active attending of the scale insects. Nevertheless, during the prolonged drought period that has occurred since 2001 (Figures

TABLE 3

Terrestrial Plant Species Recorded at Coringa-Herald National Nature Reserve: 1960–2007 (Batianoff et al. 2009)

FAMILY Species Name	Life Form ^a	Islands ^b				
		NEH	SWH	CI	SWC	SEM
AMARANTHACEAE						
<i>Achyranthes aspera</i> L.	Hp	X	X	X	X	X
BORAGINACEAE						
<i>Argusia argentea</i> (L.f.) Heine	ST	X	X	X	X	X
<i>Cordia subcordata</i> Lam.	ST	X	—	X		X
BRASSICACEAE						
<i>Lepidium englerianum</i> (Muschl.) Al-Shehbaz	Ha	X	X	X	X	X
CONVOLVULACEAE						
<i>Ipomoea macrantha</i> Roem. & Schult.	Vp	X	X	X		X
FABACEAE						
<i>Canavalia rosea</i> (Sw.) DC.	Vp	—	—	—	—	X
GOODENIACEAE						
<i>Scaevola taccada</i> (Gaertn.) Roxb.	ST	E	—	—	—	—
MALVACEAE						
<i>Abutilon albescens</i> Miq.	S	X	X	X	X	X
NYCTAGINACEAE						
<i>Boerhavia albiflora</i> Fosberg var. <i>albiflora</i>	Hp	X	X	X	X	X
<i>Boerhavia mutabilis</i> R. Br.	Hp	X	—	—	X	X
<i>Pisonia grandis</i> R. Br.	T	X	—	—	E	X
OLACACEAE						
<i>Ximения americana</i> L.	ST	E	—	—	—	—
PLUMBAGINACEAE						
<i>Plumbago zeylanica</i> L.	Hp	—	X	X	X	X
POACEAE						
<i>Digitaria ctenantha</i> (F. Muell.) Hughes	Ha	E	—	—	—	E
<i>Lepturus repens</i> (G. Forst.) R. Br.	Ha	X	X	X	X	X
<i>Sporobolus virginicus</i> (L.) Kunth	Hp	X	X	X	—	X
<i>Stenotaphrum micranthum</i> (Desv.) C. E. Hubb.	Ha	X	X	X	—	X
PORTULACACEAE						
<i>Portulaca oleracea</i> L.	Ha/Hp	X	X	X	X	X
RHAMNACEAE						
<i>Colubrina asiatica</i> Brongn.	S	—	—	—	—	X
ZYGOPHYLLACEAE						
<i>Tribulus cistoides</i> L.	Ha/Hp	X	X	X	X	X
12 families, 16 genera, 17 spp. (excluding 3 spp. locally extinct)		14 (+3E)	12	12	12 (+1E)	17 (+1E)

Note: Status: X, recorded; E, presumed locally extinct. Nomenclature follows Bostock and Holland (2007).

^a Ha, annual/early maturing short-lived herb; Hp, perennial herb; Vp, perennial vine; S, shrub; ST, tall shrub/small tree (2–5 m); T, tree (>2 m).

^b NEH, NE Herald Cay; SWH, SW Herald Cay; CI, Chilcott Island; SWC, SW Coringa Islet; SEM, SE Magdelaine Cay.

3–5), some *Pisonia* dieback (1–2 ha) was recorded (Batianoff et al. 2009). The 2001–2007 drought period has been the longest (Figure 5) and most intense continued rainfall deficit ever recorded (Figure 4). The combined loss of *P. grandis* at SW Coringa and NE Herald is estimated at 12 ha, 30% of its

original CHNNR of 40 ha (Batianoff et al. 2009).

Other islands where *P. urbicola* infestations of *P. grandis* have contributed to dieback include Palmyra Atoll 1,680 km south of Hawaii, Rose Atoll in American Samoa, Bird Island in the Seychelles, and Tryon Island

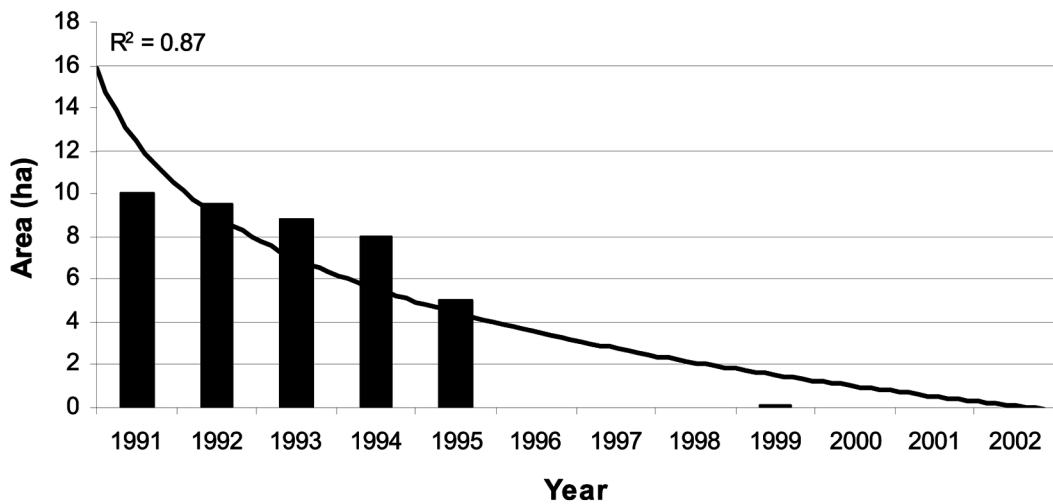


FIGURE 6. Decline in *Pisonia grandis* cover at South-West Coringa Islet, 1991–2002, derived from M. Hallam (pers. comm., 2005).

within the Capricornia Cays of the southern Great Barrier Reef (Handler et al. 2007; A. Kay, J.A.O., R. Elder, and K. Bell, unpubl. data; J.A.O., K. L. Bell, and R. J. Elder, unpubl. data; J.A.O., R. J. Elder, R. M. Charles, J. R. Platten, and K. L. Bell, unpubl. data). Around 90% of the *Pisonia* forest at Tryon Island was lost between 1994 and 2005. Infestations have also been recorded on Wilson and Heron islands within the Capricornia Cays from 2006 to 2007 (Freebairn 2006b; J.A.O., unpubl. data; J.A.O., K. L. Bell, and R. J. Elder, unpubl. data). Damage to Wilson Island forest was prevented by use of ladybird predators (*Cryptolaemus montrouzieri*). The small Heron Island infestation of *Pisonia* rain forest was contained by pruning and removing scale insect-affected material. Most of the Capricornia Cays scale infestations are reported without any reference to climatic conditions (Freebairn 2006b). Rainfall records for Heron Island (30 km southeast of Tryon Island) show increased rainfall before the *P. ur-bicola* outbreak at Heron Island and reduced rainfall during the outbreak on Tryon Island (J.A.O., K. L. Bell, and R. J. Elder, unpubl. data).

The scale insect increases at CHNNR have always followed periods of moisture

buildup during “La Niña” phases of ENSO, with dieback occurring during and after prolonged droughts. Extremes in climate variability have been suggested as an underlying cause affecting the resilience of *P. grandis* individuals, thus making them more susceptible to insect infestation (Greenslade 2008). However, this claim of *P. grandis* vulnerability to climate variability is somewhat negated by natural resilience of *P. grandis* to its environment. This species is well adapted to seasonal drought conditions by its ability to be dormant (shed leaves) and store moisture in its trunk. The rapid deaths of *Pisonia* trees at SW Coringa (Figure 7) may be due to a combination of factors such as the increased insect populations, as well as the increased moisture stress and brackishness of the freshwater lens during the prolonged drought conditions and current temperature increases (Table 2).

The phenomenon of rapid insect population increases during wet periods and dieback during droughts has been reported from elsewhere in northern Australia (White 1969). Insect populations have also been shown to increase in abundance with temperature rises (Porter et al. 1991, Cannon 1998, Ayres and Lombardero 2000, Bale et al. 2002). During drought conditions, freshwater lenses often

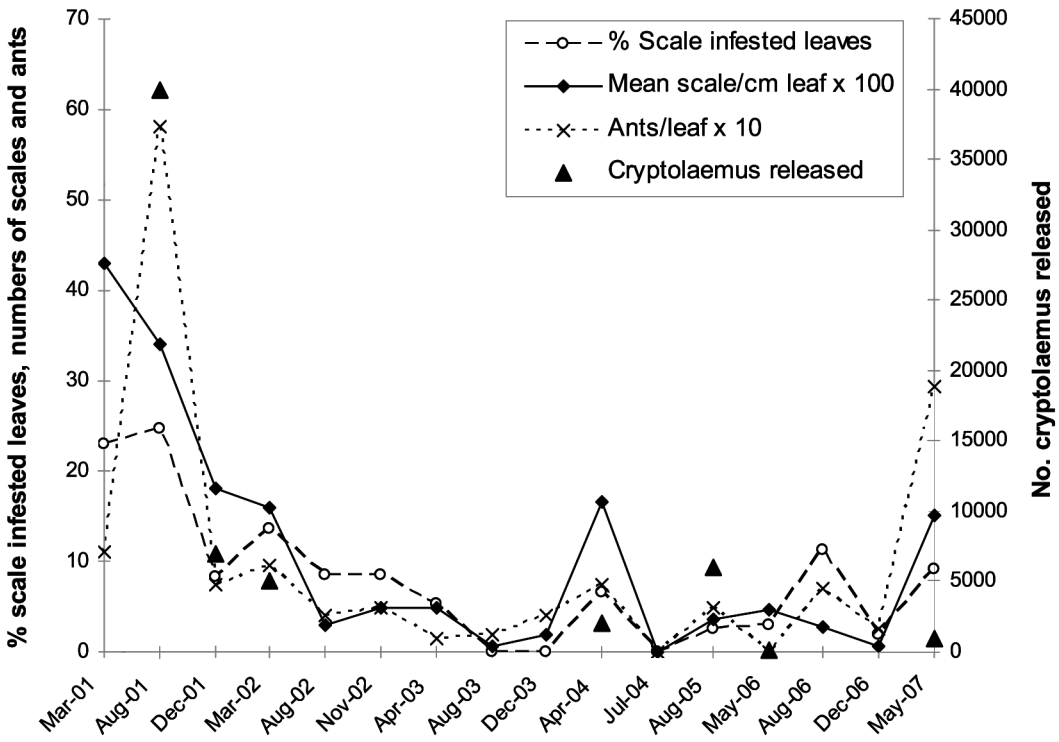


FIGURE 7. Analysis of scale insect (*Pulvinaria urbicola*) populations on *Pisonia grandis* leaves following release of predatory ladybirds (*Cryptolaemus montrouzieri*) at North-East Herald Cay, 2001–2007. (Source: Smith and Papacek 2001a,b and Freebairn 2007.)

become more brackish and/or undergo retreat (White et al. 2007) thereby stressing plants that rely on freshwater lenses (Gessel and Walker 1992). The insect damage and moisture stress could also have affected the soil nutrient uptake of *Pisonia* by reducing the abundance of vesicular-arbuscular mycorrhizal (VAM) fungi that inhabit *P. grandis* roots (Ashford and Allaway 1982, Chambers et al. 2005).

Pulvinaria urbicola is a soft scale that is native to the West Indies but is exotic to Australia (Smith and Papacek 2001a, Handler et al. 2007; A. Kay, J.A.O., R. Elder, and K. Bell, unpubl. data). It is assumed that this pest has invaded CHNRR only recently because the species had not been reported damaging *Pisonia* trees before the early 1990s. Most infestations at CHNRR were recorded in relation to higher numbers of the exotic at-

tendant ant *Tetramorium bicarinatum* (Nylander) (Smith and Papacek 2001a). This facultative mutualism between scale and other exotic ant species has also been highlighted for the *P. grandis* dieback at other locations (i.e., *Pheidole megacephala* [Fabricius], *Anoplolepis gracilipes* [Smith], and another exotic brown ant, *Monomorium* sp. [Handler et al. 2007; A. Kay, J.A.O., R. Elder, and K. Bell, unpubl. data; J.A.O., unpubl. data; J.A.O., K. L. Bell, and R. J. Elder, unpubl. data; J.A.O., R. J. Elder, R. M. Charles, J. R. Platten, and K. L. Bell, unpubl. data]). Scale insects together with attendant ants have been able to rapidly infest most of the *Pisonia* forest including other species that were more resilient to the scale’s damage (Smith and Papacek 2001a,b,c).

The question remains as to why scale insect outbreaks have occurred at so many loca-

tions in the Pacific and Indian oceans at more or less the same time. The reasons for current *Pisonia* dieback are complex. The increased winter temperatures may have resulted in the earlier breeding onset of scale insect generations during favorable wet seasons. However, the main reason for current *Pisonia* dieback seems to be the combination of climatic factors that favor native and exotic insect herbivores, together with the natural vulnerability of *Pisonia* to scale insect damage.

Dieback of *Argusia argentea*

Argusia argentea is a wind-shear tolerant, halophytic species that readily colonizes the seaward margins of coral cays. It is able to tolerate salt spray and brackish water table conditions (Batianoff et al. 2009). The *A. argentea* community occurs on all of the vegetated cays at CHNNR (Table 3). During the 2006–2007 surveys, up to 50% of *A. argentea* stands were dead or showing symptoms of dieback (Batianoff et al. 2009). Dieback and tree deaths were most pronounced along landward margins within the older stands. The estimated proportion of trees affected by dieback were South-East Magdelaine Cay (SE Magdalaine) (30–50%), NE Herald (25–30%), SW Coringa (10–20%), South-West Herald Cay (SW Herald) (10–15%), and Chilcott Island (5–10%). Some basal resprouting was observed within the dieback populations. However, on average, only 3–5% of the populations was showing any recovery (Batianoff et al. 2009).

Dieback of *A. argentea* was first reported by Donaldson (1994) at NE Herald. He suggested that a combination of an unknown fungal disease and lower rainfall had caused the dieback. These observations were made toward the end of a 5- to 6-yr drought (Figures 2 and 3), and moisture stress was undoubtedly a key element that caused dieback in mature stands of *A. argentea*. During a survey in July 1997, Batianoff (2001b) observed *Argusia* communities to be in a relatively healthy state. Oblique aerial photography (2001) of NE Herald indicates that some *A. argentea* stands along landward margins had recovered from the 1994 dieback

by establishing new populations (Batianoff et al. 2009). The May 2007 and October/November 2007 surveys showed large areas of dead *Argusia*, at the same time that some new colonizations were taking place along the seaward margins. During the last 20 yr, the *Argusia* plant community has undergone cyclic dieback of older stands followed by some recovery as well as establishment of new seedlings along the seaward margins.

Argusia argentea is listed as a threatened species on the IUCN Red List (IUCN 2008). Substantial losses of *Argusia* stands at CHNNR should be regarded as a threatening process to coral cay ecosystems. The overstory canopy provides key primary nesting habitat for many arboreal-nesting seabirds such as the Red-footed Booby, Great and Lesser Frigatebirds, and Common and Black Noddies (Batianoff 2001a). It is also the preferred shelter for ground-nesting birds such as the Red-tailed Tropicbird and the Brown Booby. On smaller cays, it is the only woody plant providing “arboreal” nesting habitat for seabirds. As in the case of *Pisonia grandis* loss, we have observed that ground cover species take the place of *Argusia* vegetation soon after the dieback. Along the seaward margins the most common species replacing *Argusia* is *Sporobolus virginicus*. The landward or the interior open areas with *Argusia* dieback are frequently invaded by ground cover species such as *Abutilon albescens*, *Achyranthes aspera*, *Ipomoea macrantha*, and *Portulaca oleracea*.

The extent of *Argusia* dieback was variable due to the configuration and size of each island. The larger islands (i.e., NE Herald, SE Magdelaine, and SW Coringa) have greater areas of mature *A. argentea* stands extended inshore. As a result, the dieback areas of *Argusia* were also greater. No symptoms of pests or pathogens were observed during the 2006–2007 surveys, though susceptibility to heliotrope moth larvae has been reported on other Pacific islands (Manner and Elevitch 2006). We postulate that *A. argentea* is adapted for warmer climatic conditions and periodic 2- to 4-yr droughts. However, mature *Argusia* plants along landward margins are vulnerable to the current prolonged drought conditions.

Dieback of *Cordia subcordata*

Well-developed stands of *Cordia subcordata* are recorded on the larger cays within CHNRR (NE Herald and SE Magdelaine). Smaller populations of *C. subcordata* occur at SW Coringa and Chilcott Island. The well-developed, dense *Cordia* thickets are difficult for larger seabirds and humans to traverse (Batianoff et al. 2009). Leafless conditions of *Cordia* were first reported by J. Hicks (unpubl. data) in December 1983 and October 1984 at NE Herald. During the 1994 survey of NE Herald, Donaldson (1994) recorded *Cordia* defoliation by grasshoppers and/or hawkmoth larvae. In June 1997, a wet period (Figures 2, 3), Batianoff (2001*b*) reported healthy stands of *C. subcordata* at NE Herald. His descriptions indicated full recovery of *Cordia* stands from the stressful periods reported by J. Hicks (unpubl. data) and Donaldson (1994).

Insect herbivory of *C. subcordata* was also recorded at NE Herald by Smith and Papecek (2001*a*) and Freebairn (2006*a,c,d*, 2007). They reported damage caused by the larvae of the noctuid moth *Armaetia columbina* (Walker). Complete defoliation by the same species was observed by Freebairn (2006*c*) during the May 2006 survey of NE Herald. During our November–December 2006 survey, more than half of the total *C. subcordata* areas showed symptoms of dieback, with extensive leaf damage by the noctuid moths and/or the giant grasshopper *Valanga irregularis* (Walker). About 15% of plants were recovering vegetatively via basal shoots. By May 2007, *C. subcordata* plants showed preferential defoliation by noctuid moth larvae and grasshoppers, with >90% of the *Cordia* population leafless. The symptoms in dieback stands were more evident (>70% of total area), with less than 10% recovery via vegetative basal shoots.

It should be noted that the soft scale insect *Pulvinaria urbicola* appears to avoid *Cordia* plants. For example, at SW Coringa two small *Cordia* stands remain standing after the loss of the entire *Pisonia* forest from 1991 to 2001 (Batianoff et al. 2009). *Cordia* is listed as threatened on the IUCN Red List due

to extensive clearing for timber and development throughout its global distribution (IUCN 2008). Our current interpretation of forest defoliation and/or dieback during dry periods requires caution. *Cordia subcordata* is capable of strong recovery during wet periods through vegetative regrowth. As a result, some dry leafless stems that may have been recorded as dead due to dieback may actually recover by root sprouting following suitable rainfall events.

Dieback of Other Vegetation

Ximения americana was first recorded at NE Herald as a small tree in December 1983 (Du Puy and Telford 1993). Ten individuals of different ages were recorded at the same location in June 1997 (Batianoff 2001*a*). During the 2006–2007 surveys, it was observed that the entire *X. americana* population had died out (Table 3). The loss of *X. americana* from NE Herald is not attributed to scale insect damage. On Tryon Island, *Ximения* plants were not observed to be damaged by scale insects (G.N.B., unpubl. data). The most likely reason for *Ximения* dieback is the prolonged droughts (Batianoff et al. 2009). However, *X. americana* forms parasitic relationships with the roots of nearby host plants (Malécot and Nickrent 2008). As a result, the loss of the host plants such as *Pisonia* during the drought most likely contributed to local extinctions of *X. americana*.

Three annual or short-lived grass species in the last decade have shown reduced coverage and/or local extinction: *Digitaria ctenantha*, *Lepturus repens*, and *Stenotaphrum micranthum* (Table 3). Specimens of *Digitaria ctenantha* were collected at SE Magdelaine in May 1984 and October 1987 (Telford 1993). It was also collected at NE Herald during June 1997 (Batianoff 2001*a*). However, during the 2006–2007 surveys, we were not able to find *D. ctenantha* at either island. It is now considered locally extinct from CHNRR (Batianoff et al. 2009).

Decline in grassy ground cover in the interior areas at NE Herald was estimated at 20–30% in June 1997 but reduced to 5–10% in 2006–2007. This moderate reduction of

ground cover of *Lepturus repens* and *Stenotaphrum micranthum* occurred under the *Abutilon albescens* overstory. At SW Herald, the dominant grassy cover in the interior was estimated at 50–70% in October 1984 by J. Hicks (unpubl. data) and T. Ayling (pers. comm., 2007) but was reduced to 10–15% by 2006–2007 (Batianoff et al. 2009). The loss of grasses was associated with an expansion of *A. albescens* open-heath. These major changes mainly affected short-lived grasses, because coverage of the perennial grass *Sporobolus virginicus* remained unaffected at NE and SW Herald.

The reasons for reduction and/or loss of short-lived grass species in the interior areas during droughts is difficult to explain. Loss of soil seed bank due to the inability to reproduce during moisture stress periods is offered as one of the major reasons. However, at NE Herald during the 2006–2007 surveys, many *Lepturus repens* seeds were observed being harvested by ants (*Tetramorium bicarinatum*). Hence, the decline in annual grass species at CHNRR is potentially being caused by exotic ants removing seeds and thus reducing the soil seed bank.

Management Implications

Coral cay ecosystems are resilient to climatic disturbances (Batianoff 2001b, Turner and Batianoff 2007). Continuing sediment production and additional accumulation of sand most likely negate some of the predicted effects of sea immersion of low cays. During the current climate change, we require local knowledge for adaptive management (Turner and Batianoff 2007). The conservation authorities managing low islands should prepare for two scenarios: (1) major reduction in size, and losses, of low islands; or (2) increased sizes of the older surviving cays and formation of new coral cays.

Loss of large areas of vegetation due to dieback, particularly the *P. grandis* forests in CHNRR, indicates that major environmental changes have already occurred. We postulate that severity of dieback is due to current climatic trends of prolonged droughts and temperature increases. Unintentional human

activities have also facilitated the introduction of insect pests, causing or exacerbating major wildlife changes in this region. The exotic ant *Tetramorium bicarinatum* is native to Southeast Asia (Shattuck and Barnett 2001). According to Greenslade (2008), this ant and *Pulvinaria urbicola* may have been present in the study area in low numbers for many years.

We have no documented knowledge as to when, and how, exotic pest insects arrived in CHNRR. *Pulvinaria urbicola* utilizes a broad spectrum of plant species, and it is common in Queensland and other regions (Freebairn 2007; J.A.O., K. L. Bell, and R. J. Elder, unpubl. data). According to Smith and Papacek (2001a), this scale insect most likely arrived from the mainland on fruits and/or vegetables consumed by visitors to CHNRR and/or Willis Island. Once established locally, the scale insects are probably successfully traversing the shorter distances required for island hopping via transportation by seabirds and/or wind. The recent arrival of the exotic ant *Pheidole megacephala* at Willis Island coincided with the expansion of the Meteorological Station facilities. We speculate that arrival of *T. bicarinatum* in CHNRR was also facilitated by human activities, such as the use of old contaminated camping equipment on NE Herald Cay (Greenslade 2008). Insect pest introduction by rafting on floating materials is also a possibility.

Human activities, combined with current climate-change conditions, are already having a major impact on the day-to-day conservation management of CHNRR. To maintain biodiversity, wildlife habitats, and ecosystem function in CHNRR, Batianoff et al. (2009) and Greenslade (2008) have listed several biosecurity procedures designed to minimize the risk of further pest introduction. Batianoff et al. (2009) recommended corrective interference and restoration of some habitats to facilitate desirable ecosystems after establishment of pest populations at conservation reserves. For example, biological control programs to control scale populations using the predatory ladybird *Cryptolaemus montrouzieri* and parasitoid wasps have been relatively successful (Smith et al. 2004, Freebairn 2007; J.A.O., unpubl. data; J.A.O., K. L.

Bell, and R. J. Elder, unpubl. data). These effective control measures were used on NE Herald and Wilson Island. The release of ladybird predators is usually done in conjunction with ant baiting.

If rain forest diversity that is functional for wildlife such as seabirds is to be maintained and/or restored at CHNNR, then management interventions may be necessary to restore former *P. grandis* rain forest areas at SW Coringa. Hand-in-hand with this vegetation management restoration is the need for continued control of populations of scale insects and exotic ants. The recovery of *P. grandis* by planting stem cuttings following scale insect damage has been successfully implemented at Tryon Island (Cruise et al. 2006; J.A.O., unpubl. data; J.A.O. and Brushe, unpubl. data, 2007 and 2008). Once a few patches of *P. grandis* have been reestablished, the natural ability of vegetative growth by stem layering is considered sufficient for its survival and spread. We also advise that any future planting of *P. grandis* should coincide with the “wet” seasons to improve the chances of reestablishment. Finally, the analyses of scale insect population increases at CHNNR indicate that it is most important to check for plague scale damage immediately after a “La Niña” phase of ENSO.

CONCLUSIONS

The reasons for the current dieback of a wide range of plants at CHNNR are complex. Most of the wildlife changes reported here are linked to current climate trends including moderate rises in temperature and drought frequency and intensity. Unintentional spreading of exotic insect species through human activities, combined with climate change, has contributed to unforeseen changes in vegetation. The loss of the *Pisonia grandis* rain forest is due to damage by scale insects and their attendant exotic ants. The reasons for the reduction of short-lived grass species requires further investigation. *Argusia argentea* dieback has resulted in the major loss of the salt spray/windbreak, as well as nesting and shelter habitats for seabirds and other animals. The protective fringe formed by *A.*

argentea also facilitates colonization and stabilization of coral cay beaches.

Overall, CHNNR vegetation shows resilience to variable and harsh seasonal climatic conditions and is capable of withstanding a wide range of temperatures, droughts, cyclones, and saltwater intrusions. These relatively simple communities are ideal ecosystems in which changes can be rapidly assessed. Despite current climate trends, the rates of vegetation change and species turnover are also dependent on individual coral cay characteristics and human activities. The decline of some important plant species that provide a high level of functional diversity is a challenge for conservation management of CHNNR. Cooperative conservation work at Willis Island is an important step for wildlife management at CHNNR as well as other cays in the Coral Sea Islands Territory.

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