

Climate change, species extinctions and ecosystem collapse

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Abstract: Climate change models have predicted many environmental impacts, but there have been relatively few published studies of ecosystem or population changes. Three studies of ecosystems and populations are reported here from the Seychelles islands. A study of a seagrass and lagoon ecosystem on Silhouette island, shows that rising sea-levels are causing changes in current patterns over the reef. Local increases in current speed act as a stress on the seagrass, leading to death of the plants over much of the reef-flat and loss of the stabilising function of the seagrass. Silt is eroded off the reef-flat and into the lagoon, removing habitat for some lagoon-dwelling animals. Significant changes in fish populations have been recorded, including the possible extinction of the goby *Asterropteryx gubbina*; this is the first report of possible species extinction for which sea-level rise appears to be the primary cause. Sea-level rise is also causing increased marine incursion into estuarine habitat, this has led to declines in populations of the dartfish *Parioglossus multiradiatus*, which should be considered to be Critically Endangered. Climate change impacts are also apparent in terrestrial systems: the destabilisation of a hybrid zone, leading to the extinction of the snail *Pachnodus velutinus* in 1994 followed a period of reduced rainfall. This is in accordance with published models that predict increased frequency of extreme weather patterns in the region. These studies indicate that rapid population and ecosystem changes are occurring. If such patterns are representative then climate change may already be one of the primary drivers of extinction.

Key words: climate change, sea level, Seychelles, Mahe, Silhouette, seagrass

Introduction

The impacts of climate change have been widely discussed and are now thought to represent the main threats to the survival of many species and the viability of ecosystems (Thomas *et al.* 2004). Specific impacts of climate change include altered climate patterns and sea-level rise which are thought to have contributed to changes in species distribution patterns and mass extinctions in the past (Twitchett 2006). Future sea-level rise associated with climate change as estimated conservatively to be in the region of 2.2-4.4mm per year with 1.5mm local variations (IPCC 2007), with higher estimates of 1.8-5.9mm (IPCC 2007) or over 14.0mm per year (Rahmstorf 2007). The rises predicted by any of these scenarios have the potential to cause similar effects. Most discussion of the impacts of sea-level rise have focussed on inundation of low-lying coastal areas and coastal erosion, with major impacts predicted for coastal infrastructure and coastal wetlands (including mangroves) and coral reefs (Nicholls *et al.* 2007). For example, studies of coastal wetland loss indicate a loss of at least 33% by 2080 (McFadden *et al.* 2007). Off-shore ecosystems predicted to be affected by sea-level rise include coral reefs and seagrass beds (Short & Neckles 1999; Hobday *et al.* 2006). Both these ecosystems have been in decline in recent years due to a number of different factors, many of which have been attributed to climate change (Hobday *et al.* 2006; Fonseca & Bell 1998) but specific impacts of sea-level rise have not been reported to date.

In the Western Indian Ocean the extinction of the land snail *Rhachistia aldabrae*

has been attributed to climate change (Gerlach 2007). A decline in the species was associated with an increase in the frequency of low rainfall years and it was concluded that this resulted in high mortality of juveniles. The prolonged period of recruitment failure may have resulted in the extinction of this species in about 2000. This is an isolated example as there have been few studies of the direct impacts of climate change on species populations; the following report describes rapid ecosystem and population changes recorded in the Seychelles islands over the past 30 years. Detailed studies of seagrass and lagoon ecosystem changes associated with sea-level rise in a reef-flat and lagoonal system (including seagrass) on Silhouette island, Seychelles have been carried out since 2000. Shorter studies of populations of the localised endemic fish *Parioglossus multiradiatus* since 2007 are reported and the previously reported extinction of the high forest snail *Pachnodus velutinus* (Gerlach 1996) is reassessed in the light of climate data.

Seagrass and lagoon ecosystem

Study site and methods

The present study was carried out in the coastal environments of Silhouette island, Seychelles. A reef system is present on the east coast of the island, comprising 51,090m² of fringing reef, 33,810m² of reef-flat (seagrass and coral rubble) and 4,455m² of lagoon (Fig. 1.) The island is the third largest of the granitic islands of the group but has a relatively low human population density (currently 0.15 per hectare) and consequent low anthropogenic impacts. The marine environment is not significantly affected by pollution, fishing is carried out at a subsistence level off-shore with the only inshore exploitation being low-levels of octopus fishing. Sea-cucumber poaching was recorded in 1998 but was not systematic and has not been recorded recently. The major impacts have been imposed by construction, with a small jetty constructed in the 19th century and replaced by a harbour in 2000 which occupies 2.5% of the historical reef-flat area. Shortly after the dredging of the harbour silt covered 10,800m² of the reef-flat adjacent to the harbour. To the south of the harbour the silt dissipated within 3 months, but remained present on the reef-flat to the north until 2003, in this area slime molds were observed growing on the reef flat by August 2000 and that area is now dominated by the algae *Halimeda macroloba*, *Centrocerca clavulatum*, *Sphacelaria rigidula* and *Padina boergesenii*. No siltation has been detected more than 40m from the harbour, off-shore currents apparently preventing more extensive siltation.

The extent of the seagrass beds and the lagoon environment has been mapped accurately from aerial photographs (1969) or satellite images (2006). Photographs taken from a helicopter have been used to measure changes in habitat distribution between 2000 and 2008. The reef-flat was divided into four categories of cover: dense seagrass (>80% cover), moderate cover (50-79%), low cover (<50%) and reef-flat (coral, coral rubble and sand). For other studies the reef system was divided into three sections defined by geographical features.

The distribution of reef-flat habitats was investigated in March-April each year from 2000 by recording the presence of seagrass, rubble, sand and coral in 1x1m quadrats at the intersections of a grid of 15x15m. Health of seagrasses has been measured

elsewhere by recording leaf size, number per short-shoot, productivity and turnover, and shoot density and biomass (Durako 1995; Phillips & Lewis 1983), these methods were adapted here. In each quadrat the percentage cover of turtle grass was estimated (to the nearest 10%), and notes made of the presence of macro-algae, substrate type and subjective evaluations of the levels of silt made. In addition the number of stems present in a 10x10cm central subquadrat recorded and a sample of the grass nearest to the centre of the quadrat collected. The plant stem was cut at the level of the surface of the sand, leaving root systems undisturbed. Each seagrass species was recorded separately. For each sample the length of the longest leaf blade (in mm) and the number of blades was recorded. Sampling was carried out by walking on the reef-flat at low tide, or by snorkelling in deeper water.

Fish species observed by snorkelling at low-tide in each of the reef sections were recorded, separate lists were kept for species in the lagoon and on the reef flat. Quantitative studies were made for gobies from December 2007; general distributions were determined by snorkelling over all parts of the reef-system. A new species of *Asterropteryx* goby was found in December 2007 (described in 2008 as *Asterropteryx gubbina* Gerlach & Gerlach 2008) and from that date the abundance of this and other goby species were investigated. Gobies were counted by observation of coral rubble patches by a stationary snorkelling observer on the edge of the rubble. This enabled all gobies to be counted in a 5 minute period as the largest rubble patch measured 5x2m. Open sand was investigated by swimming transect lines, spaced 2m apart. Surveys were carried out in 2007 (December), 2008 (March-April, June-July, December) and 2009 (March, June).

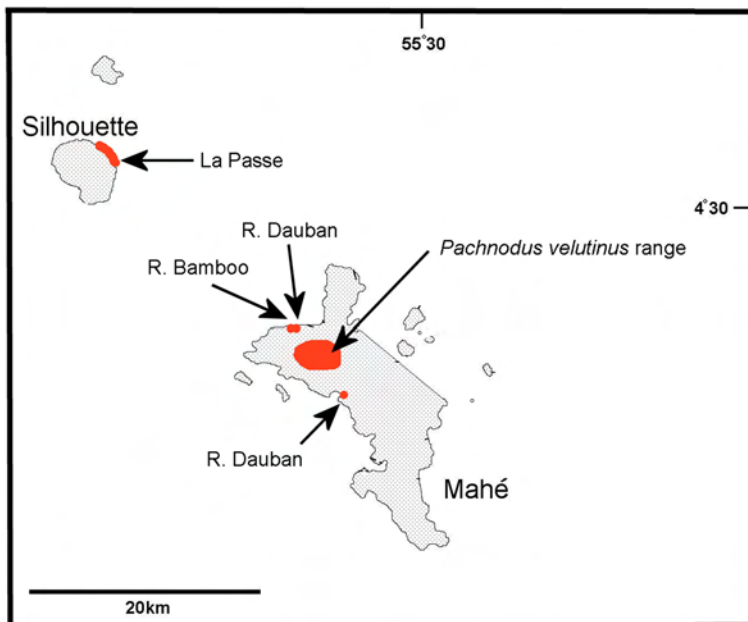


Fig. 1 Location of study sites in the Seychelles islands.

Sea-level data were obtained from the sea-level monitoring station at Pointe Larue, Mahé, 33km from the study site. Current speeds were calculated for each year from the tidal volume and channel dimensions. The effects of current speed on seagrass health were investigated by planting healthy growing stems of *Thalassia hemprichii* in 20cm deep experimental aquaria with sand substrate and sea-water. Water pumps were used to create currents in the aquaria, different individual plants were planted in positions exposing them to different current speeds. After 10 days the length of the leaf blades showing necrosis was measured to the nearest 0.5mm for each leaf.

Results and discussion

Changes in reef-flat area and the distribution of seagrass are shown in Fig. 2. Most of the reef-flat is dominated by one species, *Thalassia hemprichii*. Small patches of *Halodule uninervis* occur in shallow pools in the reef-flat and one area of *Thalassodendron ciliatum* was found in a channel 1.5-2m deep. The approximate area of dense seagrass declined from 12,570m² in 2000 to less than 10m² in 2008. The seagrass health studies show comparable results (Table 2, Fig. 3a). Associated with the decline in seagrass reef-flat holothurians and echinoids have also declined (Fig. 3b) and fish populations have also shown notable changes on the reef-flat and in the lagoon (Fig. 3c, 4). Of the lagoon fish the most significant changes occurred in *Asterropteryx gubbina*, this was abundant in December 2007 but highly localised in March 2008. In April 2008 the population declined and only a single individual could be found, extensive searches subsequently have failed to locate any further animals (Fig. 4).

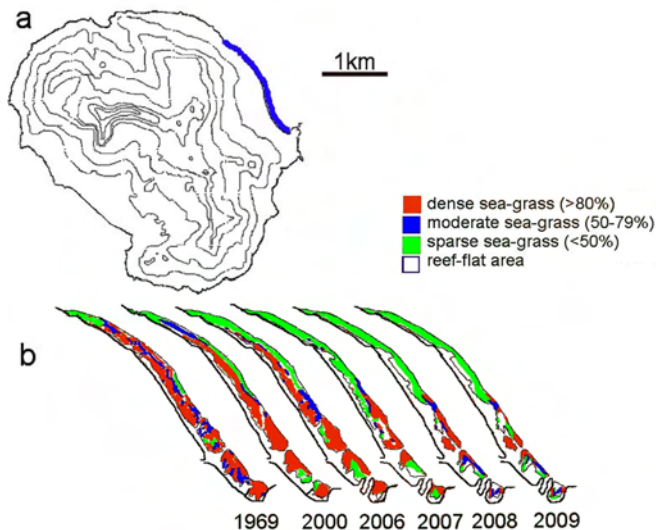


Fig. 2 Changes in reef-flat area from 1969 to 2009 based on aerial photograph. A. Location of La Passe reef flats on Silhouette island (shown in blue). B. Changes in seagrass cover on the reef flat. Note the 1969 categorisation is inferred from the depth of shading of the aerial photograph and is not directly comparable with the 2000-2008 categories where ground-truthing was possible.

Changes in seagrass health have been noted seasonally in some parts of the world, resulting from variation in rainfall and wind speeds (Hsing-Juh & Kwang-Tsao 1998) but these are far less pronounced than the changes recorded above. Extensive seagrass die-off has previously been associated with changes in water quality resulting from natural algal blooms or anthropogenic changes changing chemical balances (Lee & Dunton 2000; Tomasko *et al.* 1996; Eldridge *et al.* 2004). Studies of *Thalassia testudinum* have found this species to be affected by excess freshwater or salinity nutrient availability, light levels, water clarity and competition (Irlandi *et al.* 2002; Dawes *et al.* 1985; Ziemann *et al.* 1999; Short 1983; Lee & Dunton 1997, 2000; Tomasko & Hall 1999; McMillan & Phillips 1979; Phillips & Lewis 1983, Rose & Dawes 1999). The slime mold *Labyrinthula zosterae* is a secondary pathogen of stressed *Thalassia* seagrass (Porter & Muehlstein 1989; Robblee *et al.* 1991; Short *et al.* 1987; Muehlstein *et al.* 1991; Ralph & Short 2002; den Hartog 1987 Vergeer & den Hartog 1994; Ralph & Short 2002; Larkum *et al.* 2007; Durako & Kuss 1994) but may be a major mortality factor when combined with chemical (Carlson *et al.* 1994) or salinity stresses (Muehlstein *et al.* 1991; Burdick *et al.* 1993). Deteriorating health of seagrasses in some areas of Seychelles have been attributed to siltation, salinity changes and pollution associated with effluent discharge (Ingram & Dawson 2001). There have been no reports of *Labyrinthula* infection in Seychelles and no symptoms of infection were observed on Silhouette. Deterioration of the Silhouette seagrass beds cannot be attributed to any of the factors listed above; however, the data presented above indicate that there is a close correspondence between rising sea-levels and seagrass deterioration.

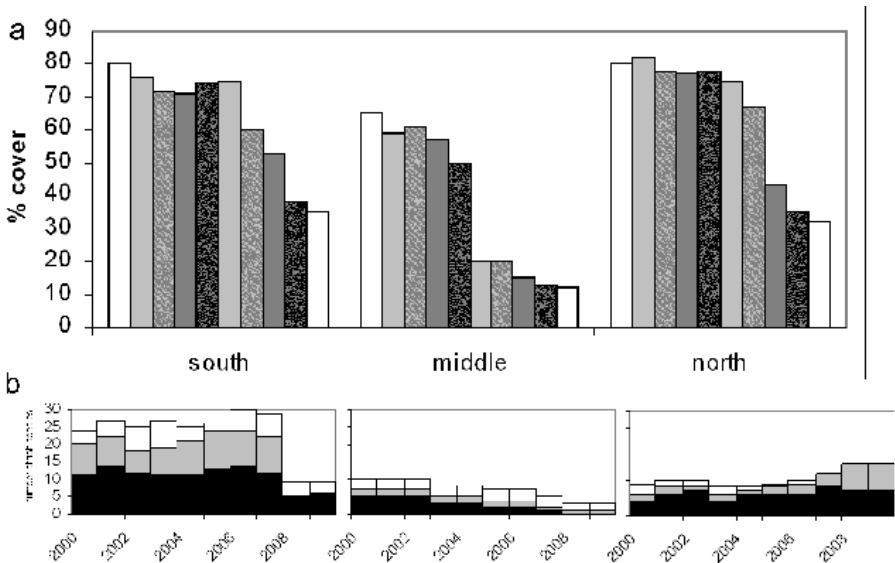


Fig. 3 Changes in seagrass condition and animal populations. a) seagrass condition (bars – years 2000-2009); b) numbers of fish species (black – reef-flat; grey – lagoon; white – lagoon sand)

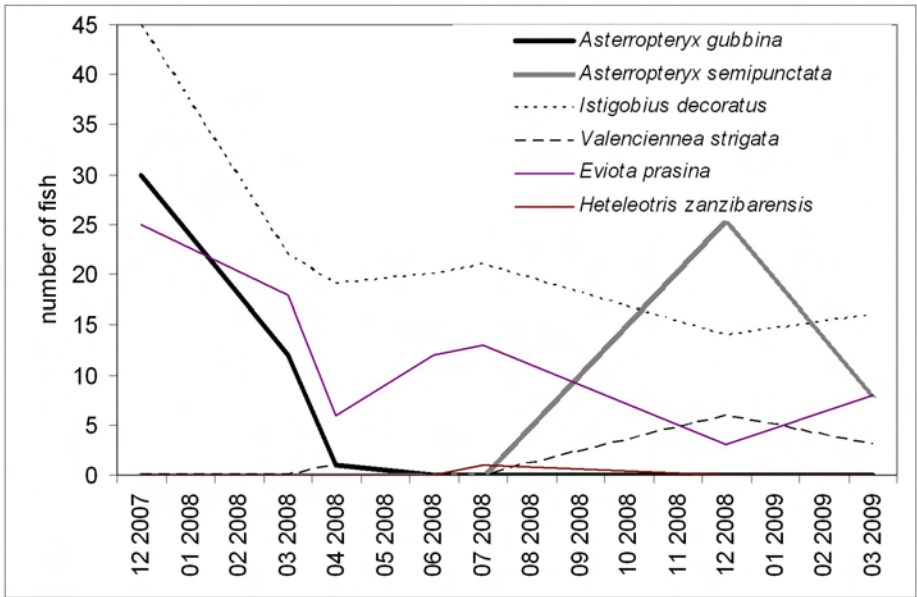


Fig. 4 Gobiidae population changes in the lagoon, t-test significance levels shown for each species

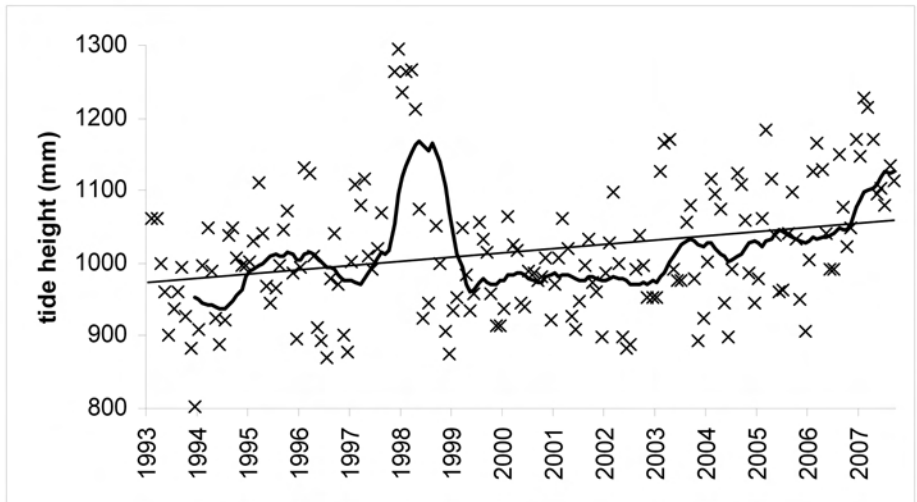


Fig. 5 Sea-level recorded at Pointe Larue, Mahé. Data from www.gloss-sealevel.org. Sea level rise averages 5.78mm per year.

Sea-level rise is significant in the Mahé data (Fig. 5), rising by 5.8mm per year, and there are consequent notable changes in current velocities over the reef. These changes correspond to changes in health of the seagrass which can also be related to increases in seagrass mortality due to excessive current speeds. Experimental data (Fig. 6) show that significant mortality (at least 5% leaf necrosis) occurs when currents exceed at $0.125\text{m}^{-\text{s}}$, 50% necrosis occurs at $0.2\text{m}^{-\text{s}}$ and 95% at $0.22\text{m}^{-\text{s}}$. In 2000 calculated current speeds in the lagoon were high ($0.29\text{-}2.06\text{m}^{-\text{s}}$), but relatively low over the reef flats ($0.02\text{-}0.14\text{m}^{-\text{s}}$). By 2008 currents had increased in the lagoon ($0.82\text{-}2.29$), reaching $8.74\text{m}^{-\text{s}}$ in the channels. Increases on the reef-flat are slight overall ($0.02\text{-}0.15\text{m}^{-\text{s}}$) but reach $0.29\text{m}^{-\text{s}}$ locally. The area of reef-flat with currents below the seagrass growth limit of $0.25\text{m}^{-\text{s}}$ fell by 83% and by 2008 none of the reef-flat remained within the optimum range ($<0.05\text{m}^{-\text{s}}$), compared to 45% in 2000.

Sea-level monitoring shows significant increases over the past few decades in many sites. The data from Seychelles indicate a rise of 16.0cm since 2000. The increased volume of water flow over the reef-flats is associated with a decline in the area of seagrass. Recent measurements of the health of seagrasses on Silhouette's reef system shows that most of the reef-flat has less seagrass cover than healthy systems elsewhere, with stem densities of 10-400 over most of the reef-flat, and locally dense patches of seagrasses with stem densities of $700\text{-}1000\text{m}^{-2}$. Studies elsewhere report average stem densities of $548\text{-}2380\text{ m}^{-2}$ (Efrtemeijer & Herman 1994; Agawin *et al.* 2001; Ingram & Dawson 2001).

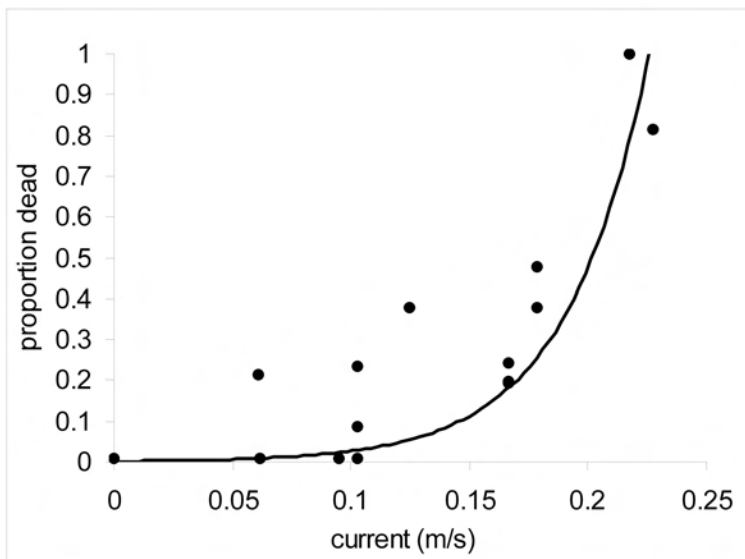


Fig. 6 Changes in *Thalassia hemprichii* health when exposed to different water currents.

Seagrasses are limited by current speeds and for most species extensive cover (at least 50%) is maintained if current speeds remain below $0.25\text{m}\cdot\text{s}^{-1}$ (de Boer 2007). The pattern is not straightforward, as growth is positively affected by rising current speeds due to increased photosynthetic activity to $0.05\text{m}\cdot\text{s}^{-1}$ (Zieman & Wezel 1980), at which point limitations start to be imposed by plant kinetics (Koch 1994; Enriquez & Rodriguez-Roman 2006; Nishihara & Ackerman 2006; Fonseca & Bell 1998). This pattern was confirmed for *Thalassia hemprichii* in the present study with significant mortality where currents reached $0.125\text{m}\cdot\text{s}^{-1}$ and 50% mortality at $0.2\text{m}\cdot\text{s}^{-1}$. Although almost all of the reef-flat area of Silhouette was below the limiting current speed in 2000, by 2008 all areas were stressed, and most no-longer suitable for seagrass growth (Fig. 7). The decline in seagrass on the reef flat has resulted in an increase in the area of bare sand and coral rubble. Associated with this is a visible increase in sedimentation in the lagoon. In 2007 the southern lagoon was clear of sediment but by March 2008 all exposed rubble was covered by a layer of at least 1mm of sand (Fig. 7). At that time depressions on the reef flat remained largely silt free, but continued erosion of the reef-flat led to siltation of all exposed rubble areas by July 2008.

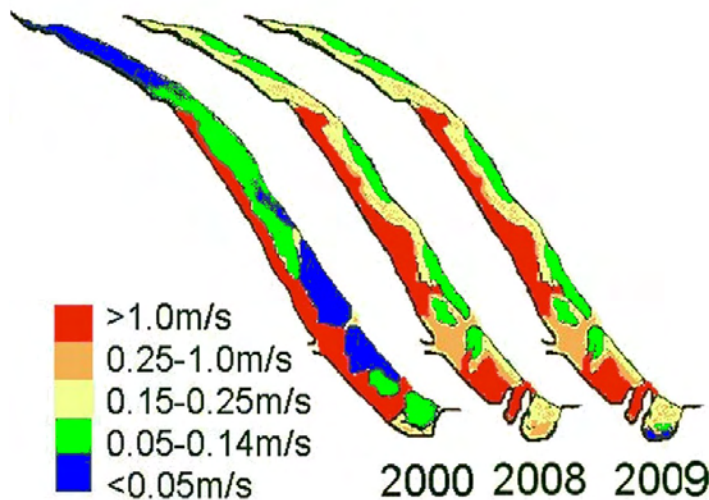


Fig. 7 Calculated current changes between 2000 and 2009, showing the decrease in distribution of slow currents suitable for seagrass growth.

Table 1. Seagrass health changes between 2007 and 2008, mean values and variance

| Site | August 2007 | | | | July 2008 | | | |
|-------|-------------|----------------------------------|----------------------------|-------------|-----------|----------------------------------|----------------------------|-------------|
| | Cover % | Stem density (m^{-2}) | Blades (m^{-2}) | Length (mm) | Cover % | Stem density (m^{-2}) | Blades (m^{-2}) | Length (mm) |
| South | 53 | 185 ± 43 | 545 ± 211 | 83 ± 21 | 38.5 | 83 ± 35 | 413 ± 138 | 56 ± 18 |
| North | 82 | 45 ± 20 | 102 ± 45 | 55 ± 22 | 13 | 35 ± 15 | 73 ± 65 | 30 ± 4 |
| Hotel | 43 | 55 ± 75 | 172 ± 98 | 90 ± 30 | 35 | 50 ± 150 | 145 ± 123 | 83 ± 36 |

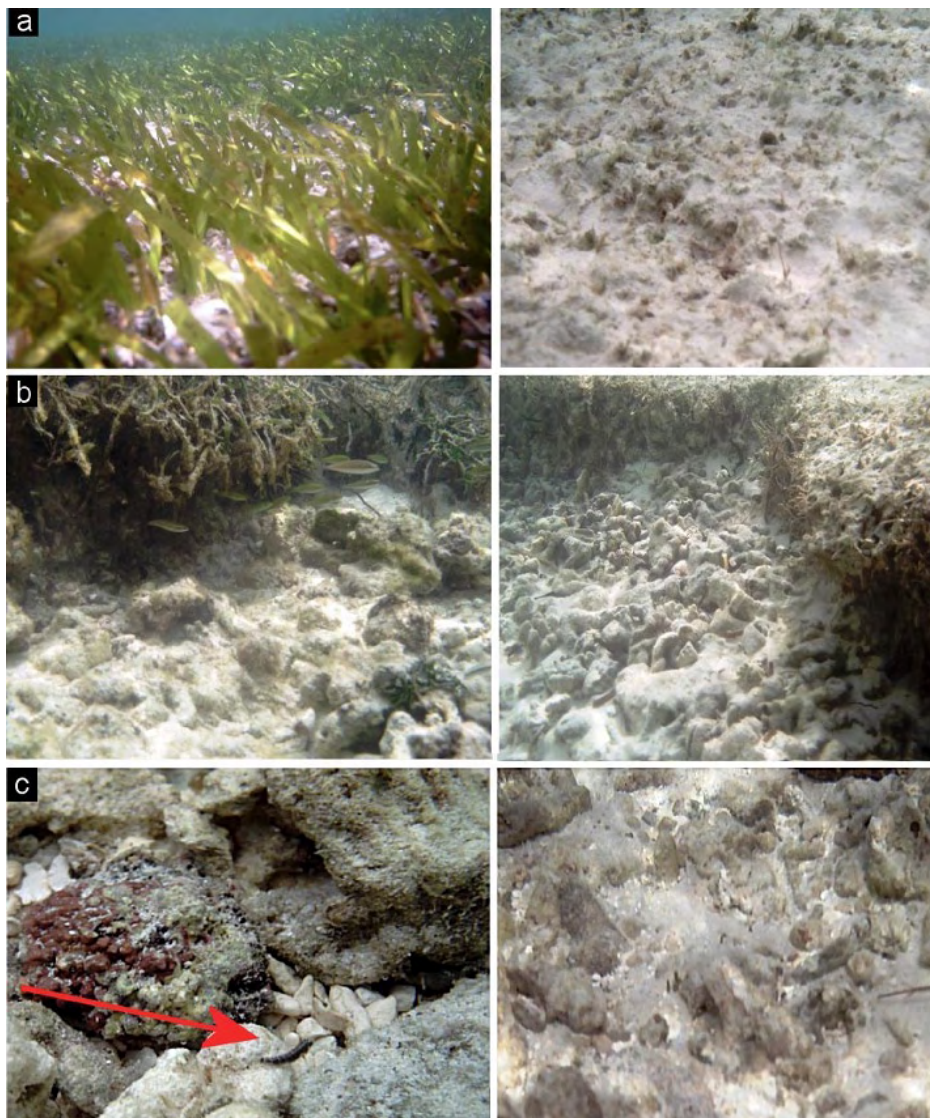


Fig. 8 Changes in the reef-flat and lagoon between 2007 and 2008. a) reef-flat in December 2007 and the same area in July 2008. b) edge of reef flat and lagoon. c) *Asterropteryx gubbina* in lagoonal coral rubble in December 2007 (arrowed) and coral rubble in the same area in July 2008.

The decline of seagrass is associated with changes in the fauna, with significant reductions in the number and diversity of fish species that live in the seagrass beds. Major changes have also occurred in the lagoon environment with decreases in most sand-dwelling goby species when siltation levels are high. In 2000 the northern part of

the lagoon was a narrow and shallow channel, increases in current speed has led to the erosion of the reef-flat and removal of silt from the channel. By 2008 the channel had deepened and widened, providing habitat to a wide variety of fish, including deeper water species such as *Sargocentron* spp. The sand exposed by the death of the seagrass is currently being colonized by the alga *Caulerpa sertularioides*. The middle section was exposed to increased current levels and heavy siltation from the harbour construction, this has resulted in early seagrass death and replacement by coralline algae, producing a relatively stable but low-diversity ecosystem. In contrast, the southern lagoon has received silt from the eroding reef-flat, this has resulted in changes in bottom-dwelling gobies. Increased silt favoured the burrowing sand-goby *Valenciennea strigata* which was observed for the first time in March-April 2008. Contemporaneously, rubble-associated species (*Eviota prasina*, *Istigobius decoratus*, *Gnatholepis cauerensis* and *Asterropteryx gubbina*) declined. *E. prasina* is mainly found under rubble in areas of the reef-flat subject to strong wave action and has remained abundant in those areas. *I. decoratus* and *G. cauerensis* are tolerant of moderate silt levels and remains present at low densities on the reef-flat. In contrast, *Asterropteryx gubbina* only occurs in silt-free rubble. It was highly abundant in December 2007 over a 22m² area but disappeared from most of the lagoon by March 2008 (restricted to 12m²), ongoing decline was observed into April 2008 when only a single individual could be found in a 2m² depression in the reef-flat. By July 2008 no silt-free rubble remained and no individuals could be found in any part of the reef-flat or lagoon. In December 2008 the widespread species *Asterropteryx semipunctata* was present in the area, this being a species of reef flats or turbid lagoon (Myers 1991). *A. gubbina* is now considered to be extinct.

***Parioglossus multiradiatus* populations**

Study sites and methods

The Seychelles dartfish *Parioglossus multiradiatus* was described in 2004 from the inter-tidal zone of two streams on Mahé, Rivière Bamboo (or ‘Bel Ombre river’) in the north of the island and Rivière Dauban in the south (Keith 2007). Surveys of streams subsequently located a third location in another stream called Rivière Dauban 25m from the Rivière Bamboo and joined to it during flood periods, both of these being tributaries of the Rivière Grand St. Louis. The characteristics of these sites are summarised in Table 2. The north Mahé localities have been surveyed on 7 occasions since December 2007 (1st and 10th December 2007, 29th November 2008, 9th and 18th December 2008, 6th April 2009 and 7th July 2009), the south Mahé locality was surveyed once on 6th April 2009. At each site the stream was observed from the banks for 15 minutes, in that time the number of dartfish observed was recorded. As the species forms small shoals near the surface of the water accurate counts could be obtained by observation. Fish were caught in a hand-net and assigned to size categories small (<21mm), medium (21–30mm) and large (>30mm). Water conditions (flow, salinity, turbidity) were recorded.

Results and discussion

Dartfish populations in the northern streams fluctuated widely during the study (Fig. 9), the largest population recorded was 35 fish and no relationship was found

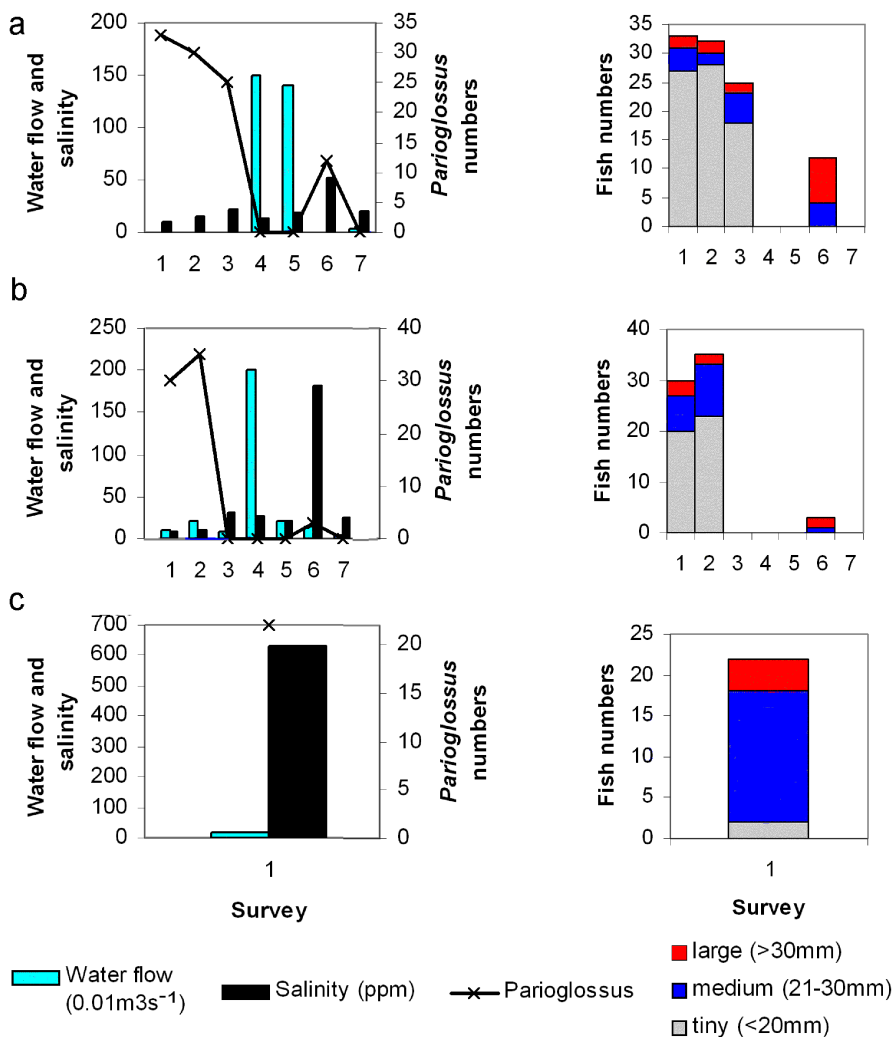


Fig. 9 Changes in *Parioglossus multiradiatus* populations and river conditions between 1st December 2007 and 7th July 2009. a) Dauban river (north Mahé); b) Bamboo river; c) Dauban river (south Mahé).

between stream width and fish numbers. In each stream the fish occupied only 5m of the length of the stream, giving them a total area of 82.5m². Proportions of juveniles varied widely, from 66-87% in the first surveys from the northern rivers to 9% in the southern Rivière Dauban. Fish numbers declined in the northern Rivière Dauban following heavy rain on 9th December 2008 which resulted in a dramatic increase in water flow (an increase of three orders of magnitude). However in the Bamboo river the population had already declined 10 days earlier when flow remained low. At this time high tides

Table 2. Details of the river characteristics at the points occupied by *Parioglossus multiradiatus*

| River | Distance to sea | Width | Water depth | Maximum population density (m ⁻²) | Salinity (ppm) | Flow (cm ³ s ⁻¹) |
|--------------------|-----------------|-------|-------------|---|----------------|---|
| Rivière Bamboo | 15m | 10m | 0.8-1.5m | 1.65* | 8-182 | 5-200 |
| Rivière Dauban (N) | 15m | 4m | 0.8-1.8m | 1.75 | 10-52 | 0.2-150 |
| Rivière Dauban (S) | 150m | 2.5m | 0.3-0.6m | 1.76 | 634 | 20 |

* at this site the river was 8m wide but dartfish only occurred in the 2m along either bank, not the central 6m

had breached the sand bar at the mouth of the river and a 3-fold increase in salinity was apparent. Sea-water was also flowing into the Rivière Dauban and although the salinity increase was less marked there (140%) fish populations had also declined (by 17%). Water flow and salinity remained high until the end of 2008 in the Rivière Dauban but declined in the Bamboo river, no recovery in dartfish populations was recorded. In April 2009 both rivers supported small numbers of dartfish although salinity remained high due to the permanent loss of sandbars. Only adult fish were found and population sizes were only 9-10% (Bamboo) and 36-48% (Dauban) of previously recorded populations. In July 2009 no dartfish could be found.

The single survey of the southern Dauban river shows that the dartfish can survive conditions of very high salinity, but juveniles are very scarce in this population (9% compared to 66-87%). There is a negative correlation between salinity levels and the percentage of juveniles ($r^2=0.829$) in all streams.

The high levels of salinity in these rivers are due to marine incursion. In the Rivière Dauban (south) the tide regularly enters the range of the dartfish behind the mangroves, the fish are able to retreat to a freshwater pool above tide level. In the north Mahé localities sand bars have protected the streams from marine incursion except when high rainfall washes out the sand bar or it is breached by exceptionally high tides. Suitable freshwater refugia do not exist in these more uniform streams. Since 2008 the sand bars have been absent. The increase in marine incursion here can be attributed to rising sea-levels (Fig. 5) as sea-levels since early 2009 have been consistently above a level needed to flood the streams.

***Pachnodus velutinus* extinction**

Study site and methods

The arboreal snail *Pachnodus velutinus* has only been recorded from high forest areas on Mahé island. In 1986 a hybrid between this species and *P. niger* was identified (Gerlach 1994) and between 1986 and 1994 its spread into the main range of *P. velutinus* was observed (Fig. 10). No pure *P. velutinus* were found after 1993. Subsequently detailed monitoring has not been carried out but visits to *P. velutinus* areas in 1996, 2002 and 2006 did not locate any snails resembling this morphotype. In 1996 the species was declared extinct, and all evidence supports this declaration. The 1994 study concluded that a stable hybrid zone had been present between the ranges of *P. velutinus* and *P. niger* since at least 1882 until its destabilisation between 1972 and 1986. Asymmetrical hybrid advantage over *P. velutinus* was demonstrated in the form

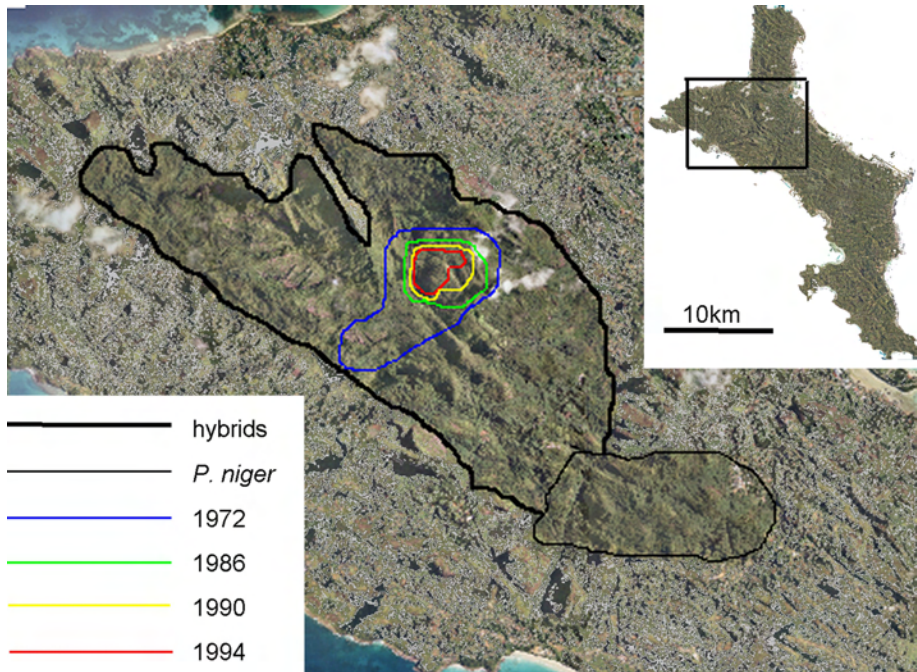


Fig. 10 Changes in *Pachnodus velutinus* distribution

of the latter's inability to enter aestivation in dry conditions due to its reduced shell. *P. niger* retains the impermeable shell and the hybrid therefore lacks any advantage over this parental taxon; there is no evidence of the change in distribution of *P. niger*. The destabilisation was speculated to be due to habitat destruction in the 1970s although such destruction over this time period covered less than 1% of the range of *P. velutinus*.

The data presented in the 1996 study were re-analysed with rainfall data from Mahé to evaluate any possible effects of climate on the extinction process.

Results and discussion

The contraction of the range of *P. velutinus* since 1972 coincides with a decrease in rainfall on Mahé (Fig 11). The rainfall decrease in 1972-87 is not unique, with similar decreases being recorded in the early 1900s. Whether these were associated with changes in hybrid zone stability is not known. The 1972-87 decrease does include an abnormally prolonged period of below average rainfall which may be responsible for the breakdown of hybrid zone stability. Close examination of this period (Fig. 11b) shows that range contraction occurred at a time of uniquely prolonged low rainfall – 6 out of 7 years having less than 95% of the long-term mean rainfall, including four consecutive years with extremely low rainfall. The final extinction of *P. velutinus* occurred in a period of above average rainfall but as noted by Gerlach (1996) the reduced rate of range contraction in 1990-4 is explained by a shift from hybrid advantage (the years 1978-1982 and 1986-1988) to the slower process of genetic swamping by the hybrids.

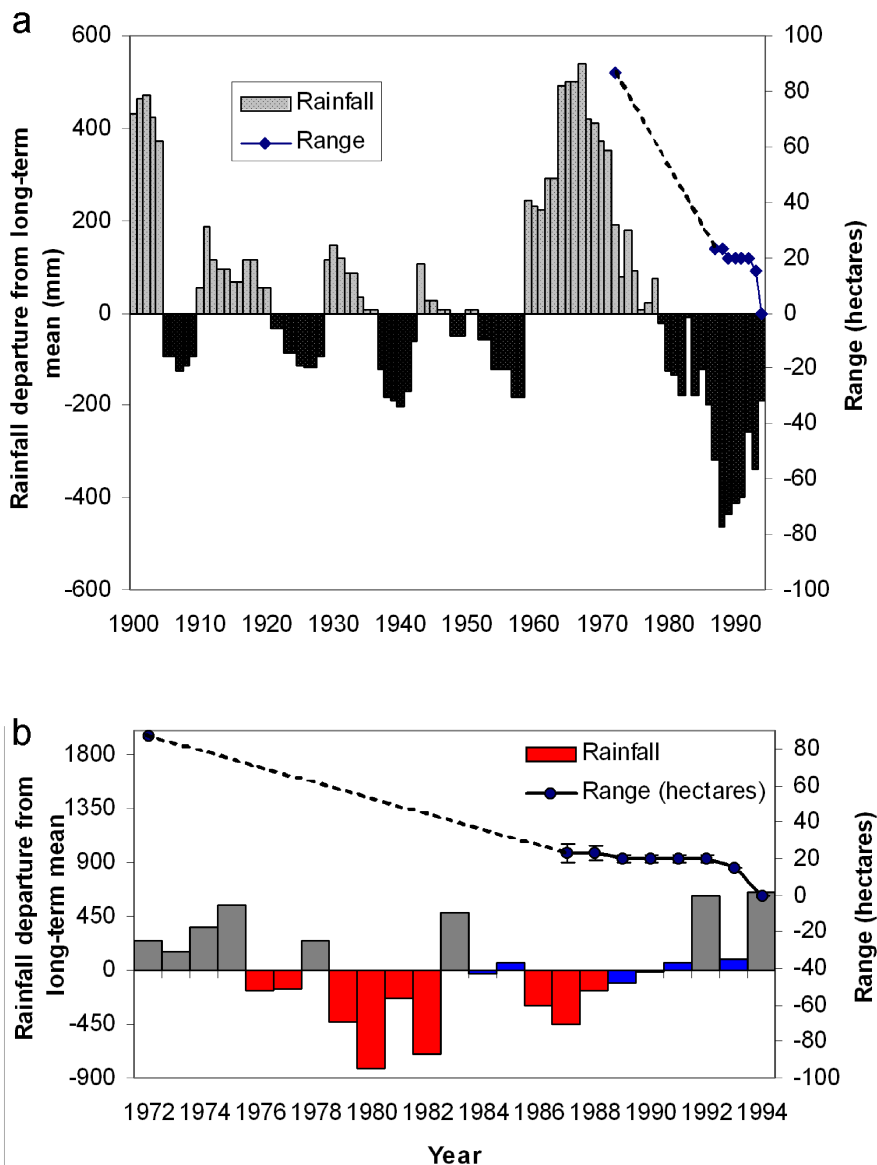


Fig. 11 Rainfall and *Pachnodus velutinus* ranges on Mahé. a) all data; b) changes in 1972-1994. grey = higher than long-term mean; red = lower than long-term mean; blue = within $\pm 95\%$ of long-term mean

Conclusion

The changes in ecosystems and populations described here can all be associated with climate change. In most discussions of the impacts of climate change it is difficult

to separate wider ecological changes of the past few decades from the specific impacts of climate. In the case of the extinction of *Rhachistia aldabrae* (Gerlach 2007) no significant ecosystem changes could be identified at the time of extinction and the only causative factors for the extinction that could be identified were climatic. Similarly, in the present cases no significant confounding ecosystem changes were identified and can be largely excluded from consideration. Changes to the reef system associated with the extinction of *Asterropteryx gubbina* are all attributable to the effects of rising sea levels, effects of other changes such as harbour construction and sea-cucumber poaching appear to be very local, whereas the seagrass ecosystem collapse has occurred over the entire area. Population declines in *Parioglossus multiradiatus* have occurred in the absence of any changes up-stream, with the only detectable change being the opening up of the streams to permanent marine incursion, again attributable to rising sea-levels. The extinction of *Pachnodus velutinus* after 1994 was previously speculated to have followed localised forest clearance (Gerlach 1996) but this explanation is unsatisfactory as the habitat loss only affected less than 1% of the range of the species. Changes in rainfall provide a closer correspondence.

Seagrass beds are of great ecological and commercial significance as they support fisheries, stabilize sediments, maintain water clarity and support diverse plant and animal communities (den Hartog 1977; Zieman 1982; Thayer *et al.* 1984; Kenworthy *et al.* 1988; Duarte 1989). Die-offs of seagrasses, such as the losses of beds of *Thalassia testudinum* in Florida (Robblee *et al.* 1991; Fourqurean & Robblee 1999) have been found to destabilise ecosystem stability (Butler *et al.* 1995). Most previous studies of degraded seagrass beds have concentrated on the ecological changes within the bed, these are highly visible but subsequent changes in adjacent systems may be even more profound. The siltation of the lagoon system described here radically alters conditions for bottom-dwelling organisms. Of the gobiid fish species some may benefit from increased sediment (e.g. *Valenciennae strigata*) whilst others are displaced by siltation of their favoured habitats. Many of these lagoon species are widespread and adaptable, and are thus may be able to recolonise lagoon once silt levels have stabilised. However, at least some species are more restricted. Although some *Asterropteryx* species are widely distributed in the Indo-Pacific (e.g. *A. semipunctatus*), others have nest-guarding reproductive strategies and lack planktonic larvae; these appear to be restricted to relatively small areas (e.g. *A. atripes* and *A. senoui*: Shibukawa & Suzuki 2002, 2007). *A. gubbina* is one such restricted species; there is no evidence of its presence outside of the granitic islands of Seychelles. Searches in similar habitats on other islands in the group have failed to locate the species, all sites being either exposed to silt carrying currents or degraded by reclamation in recent years. The ecological changes within the La Passe reef-flat and lagoon system have caused the extinction of the only known population. Accordingly this may represent the first species extinction attributable to ecological changes arising directly from sea-level rise. *Parioglossus multiradiatus* survives in only one locality, having suffered a range contraction of 76% and a population reduction of approximately 75% in two years. This species should therefore be considered Critically Endangered.

This analysis suggests that climate change in the Seychelles islands may be

responsible for the probable extinction of three species (*Rhachistia aldabrae*, *Pachnodus velutinus* and *Asterropteryx gubbina*) and the decline of another (*Parioglossus multiradiatus*). Due to their tropical location these islands have a relatively stable climate, accordingly it may be expected that climate change caused extinctions may be far more numerous than have been reported to date.

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