



Figure 4.1-1 Macro algal cover on the north of Cayman Brac. Photo Jessica Campbell



Figure 4.1-2 Macro algae washed on to the northern shore of Cayman Brac following storms from the North-West. Photo courtesy of Croy McCoy.

The Cayman Islands have however experienced a lull in such north westerly storms in recent years (Croy McCoy pers. comm.). An absence of disturbance from the North-West has consequently left macro algae to multiply unchecked on the northern shore, steadily increasing in abundance and coming to dominate the reefs. Even though hurricane Paloma decimated the island in 2008, no significant surge accompanied the storm. As a consequence, excluding very shallow reef areas, the macro algal blooms were largely unaffected and so persisted even in the face of a hurricane (Croy McCoy pers. comm.). At the most north-westerly corner of Brac, the Marine Park is exposed to the full effect of such a change in disturbance regime. Although it may be argued that a healthy and numerous fish population (associated with a Marine Park) could exert enough pressure through herbivory to control

macro algal growth, it is not unreasonable to assume that algal biomass may have rapidly surpassed the threshold at which herbivores can control algae (Gardner *et al.*, 2005, Mora, 2007). Evidence that fish density decreases for a period of time immediately following hurricanes and other disturbances, (Rogers & Miller, 2006) would further make this more plausible. Once past this point algae can start to compete with hard corals.

It may also not be the case that algal blooms such as this are caused by poor water quality, as has commonly been assumed (e.g. Bell, 1992, Costa *et al.*, 2000). Reefs at San Salvador in the Bahamas' experienced similar blooms of the algae *Microdictyon* on reefs at the leeward side of the island. These reefs were however, like the Caymans, relatively un-impacted by human activities and had very good water quality (Gall, unpublished 2009 and references within). It is not therefore unreasonable to assume that other factors, such as calmer environmental conditions seen in recent years may be at least in part responsible for the recent growth in macro algae.

Such competition for space between coral and algae may additionally explain the high incidence of stress and disease observed on corals on the northern shore of Brac. Although individual diseases were not identified as standard methodology, it was very notable that a large (but unquantifiable) proportion of recorded stress/disease was due to purple tinged spots/blotches on *Agaricia* (and to a lesser extent on *Siderastrea*). Purple lesions were of varying size and shape, but were rarely accompanied by tissue necrosis. Such symptoms appear to be entirely consistent with Borger's (2005) account of dark spot syndrome (often referred to as dark spot disease) and such a diagnosis is also supported by earlier surveys in the Caymans. Coelho & Manfrino (2007) note that dark spots and white plague were the two most common diseases around Little Cayman in surveys carried out between 1999 and 2004. They further report a significant increase in the prevalence of both dark spot syndrome, and disease in the genus *Agaricia* (mainly dark spots) from 1999 to 2004, and additionally an increase in disease in *Siderastrea* (again due to dark spots) in the last two years of the survey. High prevalence of dark spots in 2010 therefore seems consistent with previous findings.

Although dark spots appear to be a plausible diagnosis, the cause of the syndrome/disease remains controversial. As yet, no pathogen has been found to be unequivocally associated with dark spots. Borger (2005) suggests that this may in fact be because purple markings are as a result of stress rather than pathogenic infection *per.se*. It is suggested that physical irritation, particularly by other organisms such as algae could be enough to cause a stress

response. High prevalence of dark spots may therefore be closely linked to high algal abundance rather than to presence of pathogenic organisms. Similarly in experiments looking at infection by white plague, Croquer & Weil (2009) suggest that close proximity to algae could induce abnormal physiological stress that facilitates later invasion by pathogens. Physical abrasion, shading, predation by associated fauna or allochemical effects due to algal presence have all been suggested to be possible explanations for increased stress (Nugues *et al.*, 2004). The importance of these findings is clear given that greater macro algal cover seen on many Caribbean reefs in recent decades, means increased physical contact between corals and macro algae (Nugues *et al.*, 2004). Indeed even a decade ago Manfrino *et al.* (2003) report observations of macro algal encroachment and overgrowth of corals around the Caymans. Since dark spot syndrome is not always associated with tissue mortality however, the long term significance of the discolouration of corals is unclear. Nevertheless, what is clear is that diseases of coral reef organisms seem to have become a major player in the deterioration dynamics of these communities (Croquer & Weil, 2009). Furthermore, even if dark spot is not sufficient to kill corals, stressed colonies are often more vulnerable to algal overgrowth (West & Salm, 2003). Especially at CB11 and deeper reefs around the north of Brac, coral colonies were observed to be extremely small in size and surrounded by macro algae. It appears likely that coral growth is being severely limited by algal encroachment. Since many corals are unable to reproduce until a certain size is achieved, this could have severe ramifications for larval supply to reefs both locally and at wider scales. The fact that coral larvae are unable to settle on algal covered surfaces further reduces the likelihood of significant coral recruitment in the future. Increased seawater temperatures are known to facilitate the frequency and intensity of epizootic events (Croquer & Weil, 2009). It could therefore be argued that since Cayman Brac was surveyed later in the summer than the two sister islands, the higher incidence of disease observed there could be partly due to the effect of increased water temperature. Since sites on the southern shore of Brac were surveyed after those on the north, yet had much lower disease prevalence however, this possibility can largely be dismissed. If it were the case that time of surveying was having an effect then the South should have shown the highest level of disease.

More generally it is notable that across the Caribbean and beyond, the number, prevalence and virulence of diseases in marine taxa has increased in recent decades (Garzon-Ferreira *et al.*, 2001, Nugues *et al.*, 2004, Borger, 2005). It is not unreasonable to state that the impact of

white plague and white band disease alone have changed the face of Caribbean reefs, removing previously dominant framework builders and hence resulting in a severe loss of coral cover. As such, diseases now play an important role in producing mortality in the Caribbean (Croquer & Weil, 2009). It is important to note however that disease is a secondary rather than primary cause of decline that can only come about after other stresses have already weakened the coral and made it more vulnerable to infection (Croquer & Weil, 2009). As already stated in section one, bleaching is one such factor that can cause corals to be more susceptible to infection by pathogenic organisms. It is therefore not surprising that this study finds a much higher incidence of disease across the three islands than does the study by Gall (unpublished, 2009) (though due to different methodology results cannot be compared statistically). As already noted, incidence of disease is generally observed to be at a peak during summer months (Coelho & Manfrino, 2007), however since both surveys were carried out at same time of year this is unlikely. It does appear therefore that the coral bleaching event of 2009 may have weakened coral colonies and made them more susceptible to disease.

Given the discussion above, it is particularly encouraging that incidence of disease was found to be lower inside Marine Parks on Little Cayman than outside (62.5% of transects showing signs of disease within the parks compared to 87.5% outside). This would be expected if parks were helping to increase reef resilience, allowing corals to recover after stressors such as bleaching and to avoid disease and degradation. Accordingly such a result hints at the success of the Marine Parks. It is also notable however that significantly more disease was observed on the northern shore of Little Cayman, than on the southern shore (87.5% of transects showing disease compared to 62.5% of transects). Since algal blooms were absent, this seems unlikely to be linked to algal abundance and calmer conditions on the northern shore (as on Brac), and may be more influenced by extremely high diver numbers. (The Caymans receive over 350,000 dive tourists each year (Tratalos & Austin, 2001) and Bloody Bay is one of the more popular dive sites). Previous studies have also found that the heavily dived northern reefs of Bloody Bay show a much increased disease prevalence compared to all other sites around the island (Manfrino *et al.*, 2003), though no attempt was made to explain such results. The result is unexpected due to the strict regulation of the northern park, but perhaps suggests that such high diver pressure could have negative effects on coral health. It is already well known that poor buoyancy leading to diver contact with corals is harmful and previous authors have additionally linked diver presence with increased disease

on reefs (Hawkins *et al.*, 1999). Clearly in the present case this is however highly speculative and would require further research to prove or disprove any such statement.

Similarly incidence of bleaching one year after the event of 2009 was found to be lower inside the Marine Parks on Little Cayman than outside (12.5% of transects showing signs of bleaching within the parks compared to 45.8% outside). Although no surveys were done at the time, making it difficult to comment on whether there were initial differences in the number of corals to bleach inside and outside parks, it may well be the case that recovery has been more rapid within the parks. Because bleaching recorded in this study included corals that were pale and still in the process of recovery, this could be one explanation for lower observed prevalence inside parks in 2010. Again this may suggest that the parks are successful at least to some extent in helping to increase reef resiliency. Finding that abundance of bleached colonies was significantly higher at deeper reefs around Little Cayman is not surprising given the nature of the bleaching event in 2009. Bleaching was at the time noted to be much more severe on deep reefs, and additionally it was observed that while many shallow colonies quickly regained colour, bleached colonies on deeper reefs remained bright white for much longer periods (Croy McCoy pers. comm.). The influence of depth could also explain the near significance of aspect, since the northern shore of Little Cayman is on average much shallower than the southern shore. Accordingly higher bleaching prevalence on the southern shore could be a consequence of slower recovery of colonies at depth. Overall lower levels of bleaching on Little Cayman compared to Grand Cayman is also expected given that while Grand Cayman was at the centre of the hot water gyre for three to four weeks, Little Cayman was only on the periphery. Corals on Little Cayman would therefore have not bleached as severely and as such may have recovered more rapidly.

Despite the apparent success of Little Cayman's Marine Parks in terms of bleaching recovery, the same cannot be said for Grand Cayman and Cayman Brac. Recovery from bleaching within the West Bay Marine Park on Grand Cayman appears to be no more substantial than on reefs on the northern and southern shores outside parks. As already noted Cayman Brac was, only mildly affected by the 2009 bleaching event and accordingly showed the lowest bleaching prevalence of the three islands in this study. Finding that bleaching was entirely absent from the north of Brac while present in a small number of colonies on the south, is

especially interesting. Absence of colonies still in the process of recovery on the north could perhaps be linked not with more rapid recovery, but instead with previous mortality of bleached colonies. If the small number of colonies affected in 2009 failed to overcome the stress of bleaching (in combination with already elevated stress levels due to high algal abundance as discussed) then bleaching would fail to be detected in 2010. An alternative explanation is that the colonies suffering mild bleaching in 2009, recovered relatively quickly. The few bleached colonies observed in this study would then represent recent loss of zooxanthellae due to elevated temperatures in June-August 2010, rather than incomplete recovery. The summer of 2010 was in fact expected to be a particularly important year in terms of mass coral bleaching (Global Coral Reef Alliance, Croy McCoy, pers.com.). Already by early June, water temperatures of 30 degrees Celsius were being recorded (per. obs.) suggesting that corals may start to bleach as temperatures continued to rise throughout July and August. At the time of writing however no such mass bleaching event had occurred. Frequent storms and considerable cloud cover experienced throughout the summer appears to have afforded corals some degree of protection and has thus far avoided mass bleaching. Cloud cover and storms can have a protective function, as has previously been noted by a number of authors (see Obura, 2005, Baker *et al.*, 2008). Baker *et al.*, (2008) note that cloud cover can block UV rays and decrease heat flux into the ocean, thus reducing stress on corals. Storms and hurricanes can also mitigate bleaching under some circumstances, since strong vertical mixing of the water column can remove heat (Baker *et al.*, 2008). More rapid current flow has also been thought to be helpful in removing harmful free radicals from close proximity with coral tissue, therefore helping to avoid bleaching (Carpenter & Patterson, 2007). It is difficult to comment on the effect of currents in this study however since no measurements of water flow were taken during survey dives.

It is encouraging to see signs that *Acropora palmata* is making some degree of recovery after being decimated during hurricane Gilbert in 1988 (Manfrino *et al.*, 2003, Croy McCoy pers. comm.). Although not in the ten most abundant taxa on any island, the once dominant coral was seen to be re-sheeting by forming a thin veneer of tissue over the existing dead framework at a (limited) number of sites around all three islands. Though this is clearly a step in the right direction and can result in rapid increases in surface area and coral cover (Bonito & Grober-Dunsmore, 2006) the observations must be viewed with guarded optimism. Bonito & Grober-Dunsmore (2006) accordingly raise concerns about the structural integrity of such

growth. Dead coral framework will, over time have been colonised by a suite of bio-eroding organisms whose effect is to make the reef more fragile and vulnerable to breakage in future storms. Although the newly formed corals may appear healthy, they may therefore be very weak internally meaning they are susceptible to damage. Observations by Bush (pers. comm.) testify to the fragility of the reef as a result of bio-erosion. It was noted when initially attempting to implement permanent transect stakes for monitoring around Cayman Brac, that the whole reef was indeed very fragile making inserting stakes problematic; this should not be the case on a well consolidated and healthy reef.

More generally, species composition appeared to be comparable between the three islands and according to different protection levels. Some significant differences were evident but species richness and diversity were not found to be higher inside Marine Parks than outside on any of the three islands. Similarly, morphological composition was never found to differ significantly between protection levels. It would therefore appear that the second hypothesis of differing species composition within and outside the Marine Parks can be rejected. It is not the case that diversity is higher inside parks, or that reefs outside parks are characterised by more weedy fast growing species. Although results were not statistically significant, it is interesting however that the Marine Park on Grand Cayman had a higher percentage of corals that employ a brooding method of spawning than those that use broadcasting (52.9% brooders). This is in contrast to reefs outside the park, and all reefs on the two sister islands, where broadcasters are dominant. Indeed broadcasters such as *Acropora* and *Montastraea* are historically dominant on Caribbean reefs, with typically small brooding species playing a much lesser role. At a number of sites around the Caribbean including reefs around Belize, brooders have largely replaced these broadcasters (Glynn & Colley, 2008). The most abundant brooding coral taxa recorded in this study were *Agaricia* and *Porites* spp.; these two genera together are largely responsible for the higher proportion of brooders within the Marine Park. Making predictions about and trying to explain species shifts following disturbance is highly complex and problematic; while certain species invest little in stress tolerance and so may be less resistant to environmental change, these same species may be able to rapidly colonise and multiply when space opens up. Therefore which corals are favoured depends very much on the exact nature of the disturbance and may be different in each location studied. Note however that both *Agaricia* and *Porites* are small opportunistic corals, and that some authors have expressed concerns that dominance of these species could reduce reef resilience (Coelho & Manfrino, 2007) and decrease reef health. In the Grand

Cayman Marine Park however it may be that dominance by brooding species such as this is linked more closely to the sheltered westerly location of the park rather than to a shift in species due to disturbance. Opportunity for both gamete and larval supply and dispersal by currents is much more limited on the western shore than on the more exposed northern and southern shores of Grand Cayman and the sister islands. Brooding and shorter range dispersal may therefore represent a more reliable/successful strategy for coral taxa within the Grand Cayman Marine Park. Being more exposed and generally experiencing greater water flow, gamete dispersal would presumably not be an issue for corals at all other sites and is perhaps reflected in the dominance of broadcasters.

Excluding Cayman Brac (and accepting the discussion of algal blooms above) the Marine Parks do appear to have had at least some positive impacts on hard coral and macro algal cover. On both Grand Cayman and Little Cayman, macro algal abundance was significantly lower on reefs inside parks than outside (30.7% compared to 44.9% and 48.4% and 55.9% for Grand Cayman and Little Cayman respectively). Live hard coral cover was also significantly higher inside parks on Little Cayman and was just short of significance on Grand Cayman. Such findings are in agreement with the study by Gall (unpublished, 2009) and also research by McCoy *et al.* (2009) that finds greater fish biomass inside the Marine Park on Grand Cayman than on reefs outside parks. Specifically, McCoy *et al.* (2009) note that biomass of herbivores is two times higher inside the park, with larger individual fish size and additionally a more balanced ecosystem. Following the discussion in section one, herbivores are clearly crucial in maintenance of a healthy reef and may be able to keep algae in check following a disturbance (though see account by Bellwood *et al.*, 2006) enabling coral to recover. Higher rates of herbivory inside the park may therefore be partly responsible for lower algal biomass and higher coral cover. If we may accept that the Marine Parks are successful to at least some degree, a small number of other studies appear to support this conclusion and offer some evidence of the effectiveness of protected areas/ the role of herbivores in aiding recovery after bleaching events (Obura, 2005, Hughes *et al.*, 2007). Hughes *et al.* (2007) carried out experiments simulating chronic overfishing in an area that had recently undergone a bleaching event. While some areas of reef were left open to fish grazing, other areas were placed in cages that excluded large fishes; (inside cages fish abundance was reduced seven to ten fold). This is therefore a similar situation to reefs within and outside a protected area, with fish abundance several times higher inside the protected area. The experiments demonstrated that when fish were present and allowed to graze



unlimited, they were able to suppress macro-algal biomass such that algal cover averaged only 4.1%. In areas where fish populations were reduced however, macro algal coverage reached 91% and by the end of the experiment was 20 times higher than in areas grazed by fishes. Accordingly, coral cover increased much more rapidly after bleaching when fish were able to keep algae under control; by 83% compared to only 28% inside cages. The species composition was additionally found to be markedly different between caged and open areas. Algal shading in caged areas led to a greater contribution of coral taxa usually found only in deeper water or on shaded vertical surfaces (taxa such as *Euphillia* for example). Similarly Obura (2005) reports much increased macro-algal growth outside of protected areas compared to inside such areas (where more abundant fish are able to keep algae in check). Studies therefore appear to support the theory that protected reefs may be more able to recover rapidly after a bleaching event.

Both of these studies, and in particular the study by Hughes *et al.* (2007) appear however to produce much more impressive results than those observed in the present research. In the Caymans, differences in macro algal and live coral cover, although significant, were not as dramatic and clear cut as may have been hoped for after so many years of protection. Gall (unpublished, 2009) suggests that perhaps this reflects the high standard of protection afforded to reefs all around the Caymans. This would be consistent with observations that reefs are free from obvious impacts such as dynamite fishing and fish traps and do appear to have well balanced ecosystems with top carnivores including sharks present at many of the reefs surveyed (pers. obs.). An alternative explanation however is that all reefs are being affected by impacts that cannot be mitigated fully by setting aside Marine Parks and no take fishing areas. Such a theory gains some support from previous authors finding a decline in the health of Caymanian reefs in recent years. Coelho & Manfrino (2007) report for example a 40% decline in coral cover in just five years (from 26.5% cover to 16.3%) on reefs around Little Cayman. They further find that such declines were evident both within and outside the Marine Parks, with no significant difference in the relative rate of reduction. Similarly and in contrast to the present study, they report no significant difference in prevalence of disease inside and outside of parks leading them to conclude that even 20 years after their establishment, Marine Parks have proved inadequate to provide protection. High macro algal and low live coral cover observed in the present study may suggest that despite the efforts of the DoE, Caymanian reefs have undergone the phase shift seen all across the Caribbean (Gardner *et al.*, 2003).

It may appear that research is in conflict and that authors studying the same reefs are reaching dramatically different conclusions. Hence while McCoy *et al.* (2009) state that the Marine Parks have been irrefutably efficient in meeting their goals when referring to fish populations, Coelho & Manfrino (2007) argue the opposite in terms of benthic cover. Apart from different effects on fish and benthic populations, such different conclusions may also be affected by varying experimental designs. Although the findings of this study and those of Gall (unpublished, 2009) and McCoy *et al.* (2009) are encouraging, they are not enough to unequivocally demonstrate success of a protected area. Without before and after controls, and prior knowledge about the state of reefs before being designated Marine Parks, it is not possible to entirely refute the possibility that the higher coral cover inside parks in 2010 reflects a higher baseline at the time of park establishment. Higher coral cover and lower macro algal cover could therefore be a consequence of location rather than protection. Note that it has in fact been suggested that the location of the West Bay Marine Park on Grand Cayman is actually deleterious and leads to a reduced effect of protection due to unfavourable location (see Gall, unpublished 2009); this theory however is speculative and lacking in concrete evidence. Initial differences, if they existed could therefore explain why despite the apparent success reported by some authors, Coelho & Manfrino (2007) find that parks are unable to reverse or even halt the decline in reef health. Although declines are occurring all across the Caribbean (see Gardner *et al.*, 2003) and so this result must be placed into the correct context, this does not remove the fact that declines were equivalent both inside and outside Marine Parks.

The failure of Marine Parks to stop the reported decline in coral cover, and the overall high abundance of macro algae observed in this study may not however be as a consequence of problems with enforcement, as has frequently been seen at many 'paper parks' all around the world (see Mora *et al.*, 2006, Mascia, 2007 and Campbell, unpublished 2008) but may be linked to more regional scale problems. It has already been noted that the Caymans sit within the hurricane belt and experience frequent disturbance from tropical storms and hurricanes. Authors have previously reported severe and lasting declines in coral cover, and increases in macro algal cover following such disturbances (see Rogers *et al.*, 1991, Gardner *et al.*, 2005 and Rogers & Miller, 2006). In a meta-analysis Gardner *et al.* (2005) report for example that coral cover was on average 17% lower in the year following a hurricane and found little evidence of subsequent recovery. One of the reasons for this may be that fish populations are

known to be much reduced in the immediate aftermath of a hurricane. Grazing pressure and the removal of macro algae is therefore decreased, leading to increases in algal cover which make coral larvae settlement less likely in the months/years following. Additionally adult corals themselves may experience impacts that continue long after the physical damage resulting from breakage during storms. One such impact may be that of coral predators since hurricanes have a more limited impact on such predators than on the corals themselves, therefore tending to shift the ratio of coral to predators in a direction that may make recovery difficult if not impossible (Gardner *et al.*, 2005). Importantly, coming back to the issue at the centre of this thesis, bleached corals also display a much reduced reproductive potential even after a full recovery has been made (Mendes & Woodley, 2002). This is significant in light of the fact that to recover from any disturbance, recruitment of coral will be essential; if reproductive potential is limited corals may not be able to recover. It can further be seen that hurricanes and other such disturbances can disrupt any on-going recovery after a bleaching event. Such issues may shed light on findings that even after eight years Gardner *et al.* (2005) find no evidence of return to pre a storm state on many of the reefs studied. In fact Rogers *et al.* (1991) state that to date no-one has quantitatively and comprehensively demonstrated a full recovery of a coral reef following a major natural or human disturbance, and Rogers & Miller (2006) go on to specify that a reversal of a phase shift back to initial coral cover and composition has never been documented on a Caribbean reef. Even in a small number of cases where coral cover has increased, this has been due to small opportunistic taxa rather than the larger framework builders (Rogers & Miller, 2006). It may be the case that despite the apparently healthy fish populations brought about as a result of protection in the Marine Parks, the reefs in the Caymans are unable to recover to a high coral, low algae state. As stated by Gardner *et al.* (2005) removal of one stressor may not be sufficient to return Caribbean reefs to a coral dominated state (see also discussion by Cote & Darling, 2010).

Despite all of the above tenets, it is still the case that at least on Little Cayman, incidence of bleaching and disease was lower inside the Marine Parks, as discussed earlier. Clearly this alone does not prove their effectiveness, but is to some extent less affected by previous baselines since the issue is the effect of a documented event in 2009. Since remaining pale coral colonies are less abundant within the Marine Parks, this would suggest more rapid and complete recovery within parks, potentially suggesting some increase in resilience that cannot fully be accounted for by aspect or depth. It would appear that, much as the study by Gall

(unpublished, 2009) concluded, the parks seem to be beneficial in some respects, but alone, they are not enough to overcome all of the many problems faced by coral reefs in the 21<sup>st</sup> Century. Such a conclusion should not cause us to lose hope and to abandon efforts to conserve reefs however; it is essential that Caymanian reefs continue to be protected in the future. As Grand Cayman in particular continues to grow and develop in response to a rise in tourism, every effort must be made to take account of impacts such as nutrient run off, deforestation and waste disposal. Water quality is currently high around the three islands, and presently may be helping reefs to persist in the face of changes in global weather and climate. If future temperature increases are to be overcome and corals are to avoid or show tolerance to bleaching, it is essential that this remains the case. Indeed evidence suggests that corals regularly experiencing poor water quality are less resistant to thermal stress and display higher bleaching sensitivity (Wooldridge, 2009) potentially bleaching at temperatures 2-2.5°C lower than corals in non polluted waters. In light of this, a proposed Cayman Islands National Conservation Law that makes EIA mandatory for any significantly large project proposals is extremely encouraging. Steps such as this, in combination with existing protection afforded by Marine Parks may not be enough to reverse previous declines in coral cover, but they give reefs every chance possible to overcome future bleaching events and persist for as long as possible in uncertain and unfavourable environmental conditions.

## **4.2 Limitations**

Every effort was made to take account of as many confounding variables and to gather as much data as possible within the time constraints. However, as with any study there are always limitations and room for improvement. Firstly the choice of methodology meant that data collected in 2010 could not be statistically compared with that collected by Gall (unpublished, 2009) immediately before the bleaching event. Notwithstanding issues of low image resolution, the main factor influencing the decision to use underwater visual census in this study was that of time constraints. Use of video analysis and more rapid data collection underwater comes at the cost of very time consuming data analysis that was not possible within the time constraints of the present research. Nevertheless even qualitative comparisons were revealing.

Secondly the fact that Marine Park on Grand Cayman is on western shore acts to confound the effect of protection and aspect. There are no sites afforded the protection of a Marine Park situated on either of the northern or southern shores, and equally there are no sites outside of parks on the western shore. It is not possible therefore to separate any effects that are due to the Marine Parks from those that come about due to the westerly location of reefs. Unfortunately it was not possible to take any steps to overcome this problem since this is the result of decisions taken at the time of Marine Park zonation. It is notable however that a greater number of sites were surveyed outside the Marine Park around Grand Cayman than within the park. This has the beneficial effect of increasing the number of control sites and so allowing more variation between survey sites to be taken into account.

Another limitation in this study came about due to difficulties in the field in determining whether coral that was pale in colour was showing signs of recent loss of symbionts, or was still in the process of recovery from bleaching in 2009. Although the ability to discriminate between the two alternatives improved with increasing experience and familiarity with corals in the area, a decision had to be made at the start of data collection in order to retain consistency. Therefore all pale coral colonies were recorded and no discrimination was made between the two scenarios, clearly however this information would have been useful. Similarly, data on type of disease affecting coral would have provided further insights. Again however different diseases are notoriously difficult to identify in the field, especially under the time constraints imposed by underwater data collection. Therefore disease was simply recorded as present or absent.

### **4.3 Future work**

Interesting and informative extensions to the present research could include addressing the nature of the coral-symbiont relationship by determining what type (clade) of zooxanthellae are housed by corals on different reefs/ within various zones on the reef. This could help to give insights about the potential for survival in the face of future bleaching events (see Rowan & Knowlton, 1995, Rowan *et al.*, 1997, Baker, 2001, 2003, Baker *et al.*, 2004, Rowan, 2004, Baker *et al.*, 2008) and could suggest where conservation efforts might be focused in the future. For example, new protected areas may need to be implemented to provide protection to reefs that show particularly promising scope for future bleaching

resistance. Especially given the limited resources often available for conservation projects, it is clearly wise to protect those reefs that have the most promising chance of survival rather than wasting money on reefs that are unlikely to cope with climate change even if many other anthropogenic impacts are removed.

It was not possible to address current speeds and water flow in the present study, it would be interesting however to look at their effects on future coral bleaching events. It may be the case that certain reefs are afforded some degree of protection by greater water flow that reduces high temperatures thus removing stress ref (Carpenter & Patterson, 2007). Long term monitoring would be required to be able to draw any useful conclusions about such issues and it is suggested that data on current flow should be collected over a period of at least one year to take account of seasonal changes. Further such research on currents around the three islands could also provide information about the sources of recruitment and more generally explain connectivity between reefs in the area. This would be particularly interesting given the sporadic nature of *Acropora* recovery seen in the present study; colonies of the genus were absent from all but a small number of patchily distributed sites, yet multiple colonies were seen at reefs where they were present.

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#### Web resources

<http://coralpedia.bio.warwick.ac.uk/> (Viewed from 05.06.10 to 28.07.10)

Online identification guide to Caribbean Corals and Sponges.

<http://www.doe.ky> (Viewed 22.08.10)

Department of Environment, Cayman Islands Government website.

[www.worldatlas.com](http://www.worldatlas.com) (Viewed 16.08.10)

Online map resources.

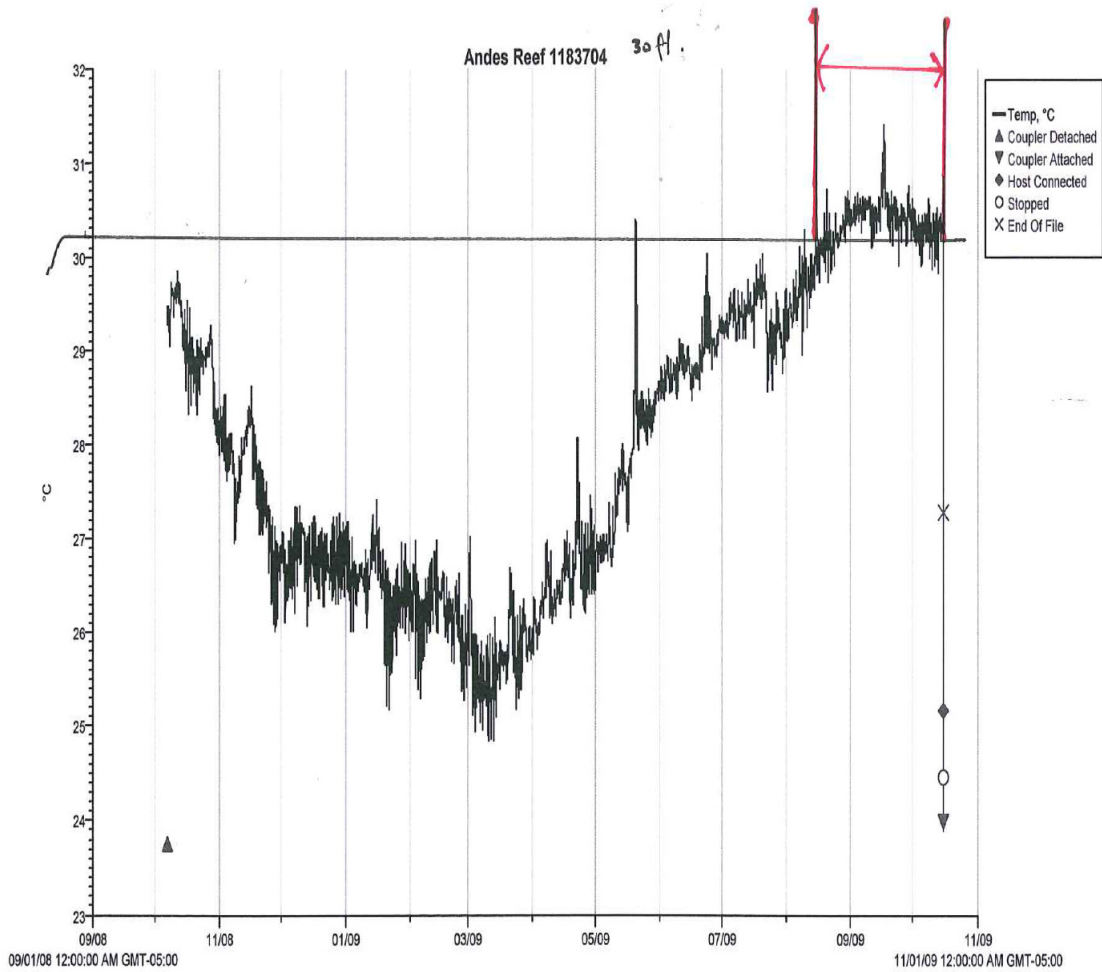
[www.globalcoral.org](http://www.globalcoral.org) (Viewed 01.09.10)

Global Coral Reef Alliance- Coral bleaching predictions

## 6 Appendix

### Appendix one: Temperature logs (2009)

Temperature data for Andes Reef (Grand Cayman) for June to September 2009. Data courtesy of Croy McCoy. Note elevated temperatures of above 30 degree Celsius.



Temperature data for Jacksons Site (Little Cayman) for June to September 2009. Data courtesy of Croy McCoy. Note elevated temperatures of above 30 degree Celsius.

