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ABSTRACT

The loss of coral reef habitats has been witnessed at a global scale including in the Florida Keys and the Caribbean. In addition to field surveys that can be spatially limited, remote sensing can provide a synoptic view of the changes occurring on coral reef habitats. Here, we utilize an 18-year time series of Landsat 5/TM and 7/ETM+ images to assess changes in eight coral reef sites in the Florida Keys National Marine Sanctuary, namely Carysfort Reef, Grecian Rocks, Molasses Reef, Conch Reef, Sombrero Reef, Looe Key Reef, Western Sambo and Sand Key Reef. Twenty-eight Landsat images (1984–2002) were used, with imagery gathered every 2 years during spring, and every 6 years during fall. The image dataset was georectified, calibrated to remote sensing reflectance and corrected for atmospheric and water-column effects. A Mahalanobis distance classification was trained for four habitat classes (‘coral’, ‘sand’, ‘bare hardbottom’ and ‘covered hardbottom’) using in situ ground-truthing data collected in 2003–2004 and using the spectral statistics from a 2002 image. The red band was considered useful only for benthic habitats in depths less than 6 m. Overall mean coral habitat loss for all sites classified by Landsat was 61% (3.4%/year), from a percentage habitat cover of 19% (1984) down to 7.6% (2002). The classification results for the eight different sites were critically reviewed. A detailed pixel by pixel examination of the spatial patterns across time suggests that the results range from ecologically plausible to unreliable due to spatial inconsistencies and/or improbable ecological successions. In situ monitoring data acquired by the Coral Reef Evaluation and Monitoring Project (CREMP) for the eight reef sites between 1996 and 2002 showed a loss in coral cover of 52% (8.7%/year), whereas the Landsat-derived coral habitat areas decreased by 37% (6.2%/year). A direct trend comparison between the entire CREMP percent coral cover data set (1996–2004) and the entire Landsat-derived coral habitat areas showed no significant difference between the two time series (ANOVA; F-test, p=0.303, n=32), despite the different scales of measurements.

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1. Introduction

Coral reefs worldwide are under multiple stresses and their health and extent are declining (Pandolfi et al., 2003). Among the different habitats found in coral reefs, coral-dominated habitats are degrading in extent and quality (Wilkinson, 2004). Coral cover often decreases in these coral habitats and may not return to the previous levels if stressors are chronic. Coral-dominated habitats are thus phase-shifting into algal-dominated and rubble habitats. Strategy shifts have also been reported in coral habitats, from one type of slow growing coral community towards other opportunistic, fast growing, type of coral community (Done, 1999). Phase and strategy shifts induce coral habitat loss. The loss can be massive (after a hurricane for instance), or patchy (after a bleaching or disease event), but it leads to fragmentation of coral reef habitats at both regional or reef scales. This fragmented state can be permanent or temporary. However, it implies that for a certain amount of time, coral diversity, habitat diversity and ecosystem processes can be degraded due to decreasing connectivity between regions, reefs and habitats. This can lead to a shift towards less complex, less diverse systems. This dynamic may be a confounding factor when measuring biodiversity patterns, since the chosen reference may be an already degraded ecosystem. This was referred to as the “shifting baseline” syndrome (Pauly, 1995). Putting coral reef...
habitat monitoring results into a large temporal and spatial context should be a priority.

In the Atlantic Ocean and Caribbean Sea reefs, live coral cover has declined markedly over the past 30 years (Gardner et al., 2003). In the Florida Keys, stressors associated with coral habitat decline include poor water quality (Boyer and Jones, 2002), overfishing and changes in water temperature (Dustan, 1999; Dustan, 2003). These stresses have increased the frequency in coral colony diseases (Patterson et al., 2002), bleaching (Hoegh-Guldberg, 1999) and algal overgrowth (Koop et al., 2001). As a result, systematic monitoring programs have been implemented over the large spatial extent of the Florida Keys National Marine Sanctuary (Dustan, 1999; Klein and Orlando, 1994; Murdoch and Aronson, 1999; Ogden et al., 1994).

The FKNMS was established in 1990. Subsequently, the Environmental Protection Agency (EPA), the National Oceanic and Atmospheric Administration (NOAA) and the state of Florida established the Water Quality Protection Plan (WQPP) in 1995 to monitor water quality and benthic habitats (i.e., coral and seagrass habitats) in the FKNMS. As part of the WQPP, coral cover has been monitored at 40 sites in the FKNMS under the Coral Reef Evaluation and Monitoring Project (CREMP), formerly the Coral Reef Monitoring Project (CRMP) (Beaver et al., 2006; Porter et al., 2002). This data set is unique and provides the basis for a comprehensive study of change in coral cover over time. The annual surveys provide live coral percent cover by species, as well as the percent cover of broader benthic categories (e.g., substrate, sponges, macroalgae). Beyond the CREMP data, several change detection studies have been performed in the FKNMS using in situ and remote sensing data (Cockey et al., 1996; Dustan & Halas, 1987; Dustan et al., 2001; Hallock et al., 2003; Miller et al., 2002; Palandro et al., 2003a; Palandro et al., 2003b; Porter and Meier, 1992; Porter et al., 2002). Of these, only two studies have looked at more than one site (Porter & Meier, 1992; Porter et al., 2002). The CREMP effort is significant and collects precise information, but synoptically monitoring the entire FKNMS is simply not possible due to the size of the sanctuary.

Remote sensing technology has been used to map shallow coral reef habitats in a variety of sites worldwide (e.g., Ahmad & Neil, 1994; Andréfouët et al., 2005; Mumby et al., 1997). For local mapping studies, at reef-scale, the current trend is to use high spatial resolution data (e.g., Quickbird and IKONOS), available since 1999 (Andréfouët et al., 2003). A recent line of work for coral reef change detection studies employed spatial statistic operators (textural measurements) to detect changes in heterogeneity, assuming that high heterogeneity reveals good 'reef health' (LeDrew et al., 2004). A recent study separating branching and boulder coral assemblages used spatial autocorrelation with the same philosophy (Purkis et al., 2006). The Landsat suite of satellites carrying the Thematic Mapper (TM, Landsat 4 and 5) and the Enhanced Thematic Mapper Plus (ETM+, Landsat 7) sensors provides the longest time series of medium resolution images since 1984. This unique resource often provides the only way to go back in time for many reefs worldwide. Landsat provides 16-day repetitive coverage for sites at a 30 m spatial resolution. TM and ETM+ data generally allow the study of habitat distributions, and in some cases habitat dynamics as well (Andréfouët et al., 2001; Andréfouët et al., 2003; Dustan et al., 2001; Palandro et al., 2003b).

Here, our objective is to measure changes in coral reef habitat extent using an 18-year (1984–2002) time series of Landsat TM and ETM+ images for eight sites in the FKNMS, thus spatially extending the results previously acquired on only one reef (Carysfort Reef, Palandro et al., 2003b). The results are compared with the percent coral cover measurements from CREMP (1996–2002). This application is challenged by availability of remote sensing data, by their processing, and by the different ecological scales that need to be considered when comparing in situ and remotely sensed information captured at two different spatial scales.

2. Materials and methods

2.1. Study sites

Four sites were selected in the Upper Keys (Carysfort Reef, Grecian Rocks, Molasses Reef and Conch Reef), one site in the Middle Keys (Sombrero Reef) and three sites in the Lower Keys (Looe Key Reef, Western Sambo and Sand Key Reef) (Figs. 1 and 2). Each site is a Sanctuary Preservation Area (SPA). They represent the three Florida Keys regions (Upper, Middle, Lower) (Shinn et al., 1989). They are monitored by the CREMP as ‘Offshore Shallow’ sites, with reef crest depths less than 6 m. The 6 m threshold was suitable to use all three Landsat visible bands (blue, green, red) since only 10% of light in the red band (630 nm–690 nm) can reach 5.6 m in depth, even in pure water (Kirk, 1994; Pope & Fry, 1997).

The selected sites are representative of other FKNMS reefs and display typical reef habitat zonations found throughout the Atlantic–Caribbean region (Jaap & Hallock, 1990). Although Acropora palmata

![Fig. 1. RGB image with locations of the eight reef sites used in this study. They are, from north to south; Upper Keys (white) — Carysfort Reef (25.20°, –80.25°), Grecian Rocks (25.10°, –80.30°), Molasses Reef (25.00°, –80.42°), Conch Reef (24.94°, –80.49°); Middle Keys (green) — Sombrero Reef (24.61°, –81.09°); Lower Keys (yellow) — Looe Key Reef (24.55°, –81.40°), Western Sambo (24.47°, –81.75°), Sand Key Reef (24.43°, –81.92°). Inset map shows location and extent of path/rows 15/43 (north) and 16/43.](image-url)
(Elkhorn coral) was once the major reef-building coral of the Florida Keys (Porter et al., 2002), this is no longer true. The remains of the *A. palmata* are now largely low-relief rubble covered with turf algae (*Ceramium* spp.). The current dominant live corals are *Montastraea cavernosa* and *Montastraea annularis* (Porter et al., 2002). Depending on depth and exposure, other major hardbottom constituents include *Millepora* spp., gorgonians (*Gorgonia* spp.) and zoanthids (*Palythoa* spp.), which can be abundant on the reef crests (Haywick & Mueller, 1997; Jaap & Hallock, 1990). Zoanthid cover is significant for image classification interpretation due to its abundance in shallow-water hardbottom zones and to its similar spectral signatures with scleractinian corals (Eric Hochberg, personal communication).

### 2.2. Coral Reef Evaluation and Monitoring Project (CREMP) data

CREMP data collection and analysis protocol is detailed in Porter et al. (2002) and is only briefly outlined here. Each CREMP site characterizes a single reef zone. One to four stations located on the shallow (~6 m) or deep forereef (~15 m) were surveyed. Stations, demarcated by a pair of permanently placed stakes, consisted of three parallel video transects spaced 0.6 m apart. Each transect was approximately 0.4 m wide and approximately 22 m long. Video transects were analyzed with PointCount® software to estimate benthic cover. Here, the station-level CREMP percent coral cover data were pooled to provide a site-level mean percent coral cover for each of the eight reef sites.
2.3. In situ ground-truth data for image classification

CREMP data provide along transect benthic cover for forereef zones only, and are not immediately suitable to define a reef-wide typology of habitats that can be mapped at the resolution of the Landsat images (30 m). CREMP coral cover is only one of the variables used to define the different habitats found along a coral reef. Within that coral reef, a habitat with high coral cover will be a coral (-dominated) habitat. The same coral reef may also have habitats without any coral, for instance a sand (-dominated) habitat. Thus, specific ground-truthing was required to train the habitat classification, in addition to the CREMP data.

Ground-truth data were collected during two summer seasons (2003–2004). Semi-quantitative cover of a number of key benthic categories (Table 1) was visually assessed using SCUBA for each 10 (along transect)×20 (across transect) m areas along a 90 m underwater transect line. Three 10 m increments were averaged to produce a single point of ground-truthed data. All transect lines were geo-located using a hand-held GPS (Garmin GPS 12XL or Garmin GPS 76, depending on field season) and ship-borne GPS (Leica MX412). A total of 192 points were acquired haphazardly in areas on or near specific CREMP sampling sites, covering a total of 115,200 m$^2$.

2.4. Landsat dataset and image processing

Twenty-two (22) Landsat 5 TM and six Landsat 7 ETM+ images were used for this study. Both sensors are equipped with three discrete visible spectral bands useful for this study, specifically the blue (band 1, 450 nm–520 nm), green (band 2, 520 nm–660 nm) and red (band 3, 630 nm–690 nm) bands. All images were resampled at 30 m spatial resolution. The study spanned the period from 1984 to 2002, with individual images every 2 years for the local spring season (March–May) and every 6 years for the local fall season (September–November). Two scenes were required to cover the Florida Keys Reef Tract (path/row 15/43 and 16/43). The images were georectified and transformed to at-sensor radiance ($L$, mW cm$^{-2}$ μm$^{-1}$ sr$^{-1}$) using the calibration coefficients provided with each file (Chander & Markham, 2003).

An atmospheric correction was performed. Atmospheric path radiances and diffuse transmittances over deep ($z >20$ m) clear water were estimated by computing the Rayleigh component using a single-scattering approximation. Aerosol properties were estimated over deep, clear water (“dark pixel” approach) using the red band and further extrapolated to other visible bands assuming a white-aerosol signal (Hu et al., 2000; Hu et al., 2001) and to adjacent shallow-water pixels. The water-leaving radiance ($L_w$) was divided by the downwelling irradiance ($E_d$) estimated at the ocean’s surface to provide the remote sensing reflectance ($R_{rs}$) at each Landsat band (Kirk, 1994).

Using a high resolution bathymetry derived from a compilation of various data sets (Palandro et al., 2004), we estimated the diffuse attenuation coefficient ($K_d$, m$^{-1}$) for the blue and green bands at the time of each Landsat image. $K_d$ is an apparent optical property (Kirk, 1994) that characterizes water clarity and how light is attenuated through the water column (Smith & Baker, 1978). $K_d$ was derived here as in Palandro et al. (2004). Briefly, patches of the same high reflectance bottom type (sand) at different depths were selected within each reef site. For bright sand bottoms, we can assume a negligible contribution of the shallow-water column to the total signal, and we have:

$$R_{rs} = C*e^{-2K_dz}$$  \hspace{1cm} (1)

where $R_{rs}(z)$ is the Landsat-derived above-water remote sensing reflectance for a pixel with bottom depth $z$, and $C$ is a pixel-independent constant. Using the selected pixels, $K_d$ was then derived as the slope of a linear regression between $z$ and $\ln(R_{rs}(z))$ after the inversion of Eq. (1). The factor “2” was to account for the attenuation in both the downward and upward directions, assuming $K_d$ can be used for both (Maritorena et al., 1994). $K_d$ values for Landsat band 3 could not be retrieved accurately from the images themselves due to the rapid attenuation of red light in water. Therefore, the absorption coefficient ($a$, m$^{-1}$) for pure water, i.e., 0.41 m$^{-1}$ (Pope & Fry, 1997), was used for $K_d$ for band 3. Although $K_d$ is also inversely proportional to the cosine of the solar zenith angle (below water), for small above-water zenith angles (<30°), the error in such estimated $K_d$ is smaller than 8%.

$K_d$ estimated over the selected sand pixels were generalized across the entire reef site and we removed the water-column attenuation effect from $R_{rs}$ and derived the benthic surface remote sensing reflectance ($R_{rs}(b)$) using Eqs. (2) and (3) below.

$$R_{rs}(z) = R_{rs}(b)/0.54$$  \hspace{1cm} (2)

$$R_{rs}(b) = R_{rs}(0')*e^{2K_dz}$$  \hspace{1cm} (3)

where $z$ is the known depth, $R_{rs}(0')$ is the below-water reflectance and 0.54 corrects for the air–water refraction (Gordon et al., 1988).

Finally, to smooth out possible remaining artifacts during the calibration process, an empirical line calibration (ELC; Jensen, 2004) was performed to account for inter- and intra-sensor variability (Teillet et al., 2001; Teillet et al., 2006), using the spring 2002 image as a reference. ELC typically uses in situ spectral data from optically bright (sand) and optically dark (seagrass) targets to calibrate an image, but has also been used to inter-calibrate two different images using pseudo-invariant features (Andréfouët et al., 2001). The shape of patches must be at least consistent throughout the time series, partially justifying the assumption that they are also spectrally stable over time.

2.5. Image classification and change detection

From the ground-truthing data (Table 1), four habitat classes were defined and labeled as ‘sand’, ‘coral’, ‘bare hardbottom’ and ‘covered hardbottom’. These classes were inherently heterogeneous given the resolution of the Landsat pixels. The label highlights the most dominant or remarkable benthic property of the class.

In situ ground-truth data indicated the location of training pixels for each habitat class. The spring 2002 images were used to define the spectral signatures statistical parameters (minimum, maximum, median, mean, standard deviation) for each class.
A Mahalanobis Distance classifier was then applied using ENVI® image processing software to every calibrated image. The habitats were mapped above 6 m of water depth, within each reef sites’ SPA demarcation.

To verify the plausibility of the computed \( R_{rs}(b) \) data, \( R_{rs}(b) \) values were converted to reflectance \( (R) \) (Gordon et al., 1988) by assuming a lambertian surface:

\[
R = R_{rs} \pi \quad (4)
\]

and compared to \textit{in situ} \( R \) measurements by Hochberg et al. (2003) after they have been linearly-mixed according to the average proportions of the benthic components in each habitat class (Table 2).

For each site, we analyzed the evolution of the coral habitat extent over time. This type of change detection analysis is similar to Palandro et al. (2003b) and Vanderstraete et al. (2006). The trend in coral habitat change was then compared to the evolution of coral cover from CREMP observations. It is necessary here to remind the reader that we compare two patterns measured at different spatial and ecological scales: a coral habitat change (from Landsat) vs. a change in live coral cover (from CREMP).

High spatial resolution IKONOS multispectral satellite imagery acquired in the spring of 2006 with 4 m pixel size was also used for each reef site to better discriminate coral reef zones and visually assess the accuracy of the Landsat classification. IKONOS is a privately owned satellite (GeoEye) launched in 1999 and possesses four visible multispectral bands close to the first four bands of Landsat ETM+ (Palandro et al., 2003a). Due to this spectral similarity, IKONOS data have been used to validate interpretation of Landsat patterns (Andréfouët et al., 2001). Here, a good agreement between Landsat class boundaries and visually interpreted limits on IKONOS data qualitatively validated the Landsat classification. Conversely, discrepancies pointed to likely errors in Landsat habitat classification. Other studies have quantitatively compared IKONOS and Landsat classifications for Carysfort Reef (Palandro et al., 2003b), but such cross-comparisons was not generalized here.

### Table 2

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<tr>
<th>Final habitat class label</th>
<th>Visually estimated benthic percent cover</th>
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<tbody>
<tr>
<td></td>
<td>Sand</td>
</tr>
<tr>
<td>Sand ((n=16))</td>
<td>75 (20.0)</td>
</tr>
<tr>
<td>Bare hardbottom ((n=25))</td>
<td>8 (10.0)</td>
</tr>
<tr>
<td>Covered hardbottom ((n=105))</td>
<td>7 (10.4)</td>
</tr>
<tr>
<td>Coral ((n=42))</td>
<td>8 (20.6)</td>
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The data in bold are where the classes on the horizontal match that of the vertical (e.g. sand to sand). Seagrass were never abundant on reefs, and did not justify the inclusion of a ‘seagrass’ habitat class. Classes were labeled according to their dominant cover, except for the habitat labeled ‘coral’. However, the high abundance of corals in that class justified its label.

3. Results

3.1. Spectral signatures

Landsat-derived \( R \) values were compared to the linearly-mixed \textit{in situ} \( R \) values for the four benthic classes (Fig. 3). The agreement was the most satisfactory for covered hardbottom, bare hardbottom and coral habitat. Sand spectra were less in agreement. When the \textit{in situ} \( R \) data were averaged around 485 nm (450–520 nm), 560 nm (520–600 nm) and 660 nm (630–690 nm) for each of the four classes and compared to the Landsat-derived \( R \) at these wavelengths, the correlation was high, with \( r^2=0.891 \) for all bands and classes.

![Fig. 3. Comparison of Landsat-derived reflectance (\( R \)) with ±1 standard deviation (black solid) to \textit{in situ} \( R \) values (grey solid) based on values published by Hochberg et al. (2003). Actual spectral data were provided by E. Hochberg and linearly-mixed to compare sand (a), bare hardbottom (b), covered hardbottom (c) and coral habitat (d), dashed lines indicate ±1 standard deviation.](image-url)
3.2. Classification analyses

The 2002 Landsat-derived coral habitat pixel locations were overlaid on top of the 2006 IKONOS image data (Fig. 4). From the comparison of Landsat-derived classifications and visually interpreted reef zonations from IKONOS, it appeared obvious that several reef site pixels at certain dates were misclassified, prompting a closer examination of the spatial patterns for each classification, and their coherence across time. Specifically, the IKONOS imagery allowed for a more accurate delineation of the backreef rubble zone from the reef crest and forereef zones. We found Landsat-derived coral habitat pixels in the backreef rubble zone for the eight reef across the full time series of Landsat classified image data. These backreef zones have historically low coverage of live stony coral (Wheaton & Jaap, 1988). However, Palythoa spp. can exist in abundance in this zone and explain why some backreef areas are assigned to the coral habitat type (see Discussion).

3.3. Seasonal variability

Image pairs from the spring and fall for 1984, 1990, 1996 and 2002 were analyzed for seasonal variability in percent coral habitat. Where paired reef site values were present (n=20), there was no significant difference in Landsat-derived percent coral habitat due to season (paired t-test, p=0.535). Accordingly, the spring and fall percent coral habitat values have been averaged to derive a single yearly value.

3.4. Change detection

A total of 89 reef site images over time provided interpretable data, out of a possible 112 opportunities (8 reef sites×14 images per reef site). Cloud cover explains the 23 missed opportunities. For illustration, we show a sequence of classified images for two reefs, Looe Key Reef and Molasses Reef (Fig. 5. Given available space here, not all reef sites can be shown. For the full suite of change detection imagery, please contact D. Palandro). The ecological succession and spatial consistency for each of the four benthic classes were examined for each site, in order to detect inconsistent erroneous classifications. Inconsistencies were first revealed when the spatial distribution of the different classes varied too abruptly to be realistic. For instance, a patch growing in size in several images that abruptly decreased in the next image, before returning to a larger size in the subsequent image, pointed to a likely problem in the second to last image. Second, inconsistencies occurred with abrupt changes in the nature of the habitat, for instance a large sand patch turning into a coral habitat in 2 years was suspicious and pointed to an unlikely sequence of habitat evolution given the considered time frame (Scopéllitis et al., 2007). Each of the eight reef sites is discussed individually below, followed by a synthesis discussion.

3.4.1. Carysfort Reef (Upper Keys)

The time series of Carysfort Reef showed areas of little to no change in most of the images. These areas included a sand patch in the northwest, the coral habitat clustered in the southern center, bare hardbottom area south of the sand patch, a second bare hardbottom area west of the coral habitat cluster and covered hardbottom along the eastern edge of the reef site. However, the 1986, 1990 and 1998 images show a lack of continuity in the coral habitat class progression (or regression). The class was more fragmented in 1986 and 1990, extended too far south in 1990, and appear larger in 1998. Overall, also looking at the other classes, the 1986, 1990 and 1998 images lacked the consistency displayed by the other images within the time series.

3.4.2. Grecian Rocks (Upper Keys)

The time series of Grecian Rocks also had a central coral habitat area, with the surrounding areas classified as a mix of bare and covered hardbottom. The coral habitat progressed from nearly the full width of the central area to just the central western portion of the reef site. The coral habitat became more fragmented in the 1988 image with areas of bare hardbottom mixed into the central region and the 1998 image showed two distinct coral habitat areas. The trend showed a lack of consistency considering all classes mainly in the 1988, 1996 and 1998 images.

3.4.3. Molasses Reef (Upper Keys)

Several areas located on Molasses Reef are consistent throughout the time series. There is a central southwest to northeast coral habitat area, covered hardbottom areas in the northwest, south and southeast, and a sand area in the west. The coral habitat area showed a wide variation for the 1984, 1992, 1998 and 2002 images. The western sand area was inconsistent for the 1986, 1990 and 1996 images. The northwest covered hardbottom area was inconsistent only in the 1984 image, whereas the south area was uniform throughout the time series. The southeast area, which ran parallel to the coral habitat area, was dominated by covered hardbottom, except in the 1992 and 1998 images. The later images had a mix of bare hardbottom and sand in this area. Overall, the 1992 and 1998 images were outliers.
3.4.4. Conch Reef (Upper Keys)

The time series on Conch Reef showed the highest consistency between images and showed a plausible progression of the loss of coral habitat. This occurred in two areas; one centrally in the north and one centrally in the south. The coral habitat areas remained tightly clustered with the exception of the 1984 and 1998 images, where the latter displayed the southern coral habitat area as two separate areas.

3.4.5. Sombrero Reef (Middle Keys)

There were only seven images for Sombrero Reef. The coral habitat was clustered in the southern area with a mix of varying levels of sand, bare and covered hardbottom surrounding it. The 1984 and 1990 images displayed a number of coral habitat pixels outside of the southern cluster. These images were also the noisiest. Overall, the inconsistent images in this time series were the 1984, 1986 and 1990 images.

Fig. 5. Classified dataset for spring images for Looe Key Reef (A) and Molasses Reef (B). Classification color codes are: red = coral habitat, brown = covered hardbottom, yellow = bare hardbottom and green = sand.
3.4.6. Looe Key Reef (Lower Keys)

The coral habitat extended along the southern edge, with small clusters that ran along a central southeast to northwest area. Coral habitat pixels also possessed a highly noisy pattern throughout the time series, specifically in the 1992, 1994, 1996 and 1998 images. The central southeast to northwest area was dominated by bare hard-bottom and extended farther to the south upon the decline of coral habitat. This was not the case for the 1996 and 1998 image, which displayed a mix of covered hardbottom and coral habitat instead. The 1996 and 1998 images showed the least consistency for this time series.

3.4.7. Western Sambo (Lower Keys)

Two dense areas of coral habitat were initially detected. As both areas of coral habitat regressed through time, bare hardbottom increased in those areas. In the western coral habitat area, the bare hardbottom increased in size and split in two the coral habitat area. The 1990 and 2000 images had coral habitat pixels in the large area to the north, inconsistent with the other images in the time series. Also, the 2002 image showed sand along the southernmost edge of the reef site. Overall, the 1990 and 2000 were the least consistent images.

3.4.8. Sand Key Reef (Lower Keys)

The time series of Sand Key Reef included a single cluster of coral habitat located in the southern central area, a covered hardbottom dominated area to the east and sand and bare hardbottom area to the north. The 1984 image was inconsistent in the northern area, displaying coral habitat pixels. It also displayed a noisy central coral habitat area.

3.4.9. Synthesis for all sites

Including the suspect images, the total mean Landsat-derived percent coral habitat data for all reef sites was 7.6% in 2002 down from 19% for 1984 (adjusted) (Table 3). The adjusted mean value compensates for the lack of a 1984 image for Western Sambo. An adjustment was deemed necessary as this reef has the second highest percent coral habitat in 1988 (the first year that data were available). The adjustment was made by assuming a linear trend to extrapolate back to 1984. The overall 18-year coral habitat decline for all reef sites was 61%, with an average of 3.4% loss per year. Every time period (2 years), except 1986–1988, showed a decline in percent Landsat-derived coral habitat. The percent decline per time period ranged from 2.2% (1994–1996) to 32% (1998–2000).

The Lower Keys had the highest presence of coral habitat throughout the study, followed by the Upper and Middle Keys, respectively. Grecian Rocks in the Upper Keys started with the highest percent coral habitat value in 1984 with 33% and also had the highest percent coral cover in 2002 with 15%. The lowest percent coral habitat in 1984 was found at Sombrero Reef (Middle Keys) with 7.8%, but was only the second lowest in 2002 (4.5%). The lowest percent coral habitat in 2002 was Conch Reef (Upper Keys) with 3.4%.

The Upper Keys reef sites yielded the greatest decline in coral habitat with 62% (3.5%/year), followed by the Lower Keys with 59% (3.3%/year) and finally, the single Middle Keys reef site with 45% (2.5%/year). The decline in the Upper Keys was stable among the sites, with only 12% change variability. Western Sambo showed the highest relative loss in coral habitat at 72% (1988–2002) and Looe Key Reef the lowest at 38% (1984–2002).

3.5. Comparison with CREMP coral cover data

For all concurrent data points (n=32, with only four concurrent years for eight sites), there was no significant difference (paired t-test, p=0.468) between the CREMP percent coral cover loss and the Landsat-derived percent coral habitat loss. The correlation was high with r²=0.704 (Fig. 6).

Relative percent change rank between data sets was examined. In other words, we compared at two different ecological scales (habitat vs. coral cover) how each reef site compared to the others in terms of percent change over time. This comparison showed that two sites agreed in relative ranking (including the highest and lowest percent changed), two sites differed by one rank and two other sites differed by two ranks (Table 4). The two datasets were also consistent for the four reef sites that exhibited the highest rates of change and the four reef sites that exhibited the lowest rates. For each site, an analysis of covariance (ANCOVA, Sokal & Rohlf, 1981) was performed to compare the trend lines across time for the two data sets (Fig. 7). Considering data from all eight reef sites taken together, there was no significant difference in slopes (i.e., rates of change) between Landsat-derived

![Correlation of Landsat percent coral habitat versus CREMP percent coral cover for all concurrent data points (1996–2002) in the Upper (■), Middle (▲) and Lower (○) Keys, r²=0.704 (n=32).](image)

Table 3

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<td>17.7</td>
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<tr>
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<td>7.0</td>
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Empty cells are occurrences when no image data were available and values in bold are those found to be ecologically inconsistent.

Table 4

<table>
<thead>
<tr>
<th>CREMP</th>
<th>Reef site</th>
<th>Landsat</th>
<th>% Change (1984–2002)</th>
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<tbody>
<tr>
<td>1</td>
<td>Western Sambo</td>
<td>1</td>
<td>~71.9*</td>
</tr>
<tr>
<td>2</td>
<td>Molasses Reef</td>
<td>4</td>
<td>~59.2</td>
</tr>
<tr>
<td>3</td>
<td>Carysfort Reef</td>
<td>3</td>
<td>~71.5</td>
</tr>
<tr>
<td>4</td>
<td>Conch Reef</td>
<td>2</td>
<td>~66.3</td>
</tr>
<tr>
<td>5</td>
<td>Sand Key Reef</td>
<td>5</td>
<td>~60.5</td>
</tr>
<tr>
<td>6</td>
<td>Sombrero Reef</td>
<td>7</td>
<td>~42.3</td>
</tr>
<tr>
<td>7</td>
<td>Grecian Rocks</td>
<td>6</td>
<td>~57.8</td>
</tr>
<tr>
<td>8</td>
<td>Looe Key Reef</td>
<td>8</td>
<td>~38.4</td>
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</tbody>
</table>

Ranking is highest percent change (1) to lowest (8). Also shown is the total Landsat-derived coral habitat change between 1984 and 2002 (*Note that Western Sambo percent change is from 1988–2002).
Fig. 7. Percent coral habitat for Landsat (■) and CREMP (○) full-timeline data (y-axis) per year (x-axis) with linear regression line added for each dataset, for selected Florida Keys reef sites.
change in coral habitat and CREMP change in coral cover ($F$-test, $p = 0.030$). Individually, Sombrero Reef and Sand Key Reef were the only two reef sites where the slopes were significantly different ($F$-test, $p = 0.001$ and $p = 0.034$, respectively).

The pooled CREMP Stony coral cover data showed a loss of 52% (8.7%/year) between 1996 and 2002, whereas the Landsat-derived coral habitat data showed a decline of 37% (6.2%/year) for the same time period. CREMP data showed the highest decline in the Middle Keys, and systematically lower declines in the Upper Keys and Lower Keys, respectively. This is in contrast with the Landsat-derived coral habitat data, in which the single Middle Keys reef site examined had the lowest decline.

Both time series showed the highest relative decline per time period (2 years) occurring between 1998 and 2000 (CREMP=37%, Landsat=32%). The lowest decline occurred between 2000 and 2002 for both data sets (CREMP=2.4%, Landsat=3.7%). CREMP percent coral cover data and Landsat-derived percent coral habitat data matched geographically between the two datasets (2002): the highest percentages were found in the Lower (CREMP=9.5%, Landsat=8.8%), Upper (CREMP=5.3%, Landsat=7.7%) and Middle Keys (CREMP=3.2%, Landsat=4.1%), respectively.

4. Discussion

4.1. Trends for the Florida Keys

The concurrent data between CREMP loss of coral cover and the Landsat loss of coral habitat yielded a correlation, $r^2 = 0.704$, showing a good agreement between the datasets. This indicates that for these reef sites, although the information was acquired at different spatial and ecological scales, they provide a comparable measurement of reef benthic evolution and status. The good agreement does not mean that Landsat can be used to measure coral cover. Here, it is an indication that coral cover and abundance of coral habitat are related across two different spatial scales, with similar magnitude. This similar magnitude should not be generalized to other sites, or considered as a rule to scale-down Landsat interpretation to coral cover measurements.

The similarity in data sets was further corroborated by the relative percent change over time for each individual site (Table 4). In all cases the rates of decline were greater for CREMP data, with the slopes for Grecian Rocks being very close (CREMP, $m = -1.043$ and Landsat, $m = -1.035$). Taken individually, only two sites had slopes significantly different, Sombrero Reef and Sand Key Reef. Both datasets identified 2000−2002 as the time period of lowest change (Landsat, 4% and CREMP, 3%). The single greatest decline per time period for both datasets occurred between 1998 and 2000. The rates of decline derived from CREMP (37%) are comparable to that of Landsat (32%) for that period. For the CREMP data this can be attributed to extensive bleaching, and subsequent mortality, which occurred during the summer of 1998. However, at the scale of coral habitat and Landsat resolution, there is no clear cause. We suggest that the habitat dynamics highlighted by the Landsat time series reflect the long-term influence of the variety of stressors that impact Florida reefs. Coral colonies may react quickly to a perturbation (e.g. dying after a few weeks of bleaching or disease) but habitats have a much slower dynamic (Scópelitis et al., 2007).

The comparison between satellite-derived and in situ benthic data provides a strong argument for continuing the use of remote sensing for monitoring coral reef habitats in the Florida Keys, using Landsat if the time series is maintained. Of greater interest may be using IKONOS or Quickbird data that provide similar spectral resolution as Landsat but with enhanced spatial resolution. However, the generalization of these recommendations to other coral reef sites of the planet warrants further investigation. Parallel trends in relative loss can be expected. Here, there is also an agreement in terms of absolute values, which is probably a local effect that can not be taken as a generic rule.

The availability of IKONOS imagery helped understanding the local patterns observed on Landsat classifications along the backreef, reef crest and shallow forereef for each site (Fig. 4). Coral habitats were found on the backreef, an area that traditionally has very low live coral cover. Unfortunately, very few studies, including CREMP, precisely monitor the backreef rubble zones. However, Wheaton and Jaap (1988) found whole and fragmented colonies of *A. palmata* and *A. cervicornis* in the backreef of Looe Key Reef, which they hypothesized were likely transported inshore from the shallow forereef. *Porites astreoides* was also found in the backreef of Looe Key Reef (Wheaton & Jaap, 1988). If conditions are favorable, transported pieces of coral and new recruits can thrive in the backreef zone. Coral colonies from four different species were found in the shallow ($z = 1 \text{ m}$) reef flat of Carysfort Reef in 1981–1982 (Dustan & Halas, 1987). *A. palmata* colonies were described by Shinn et al. (1989) for Grecian Rocks. However, *Palythoa* spp. are common and can dominate in shallow-water high energy hardbottom environments (e.g., reef crest and reef flat) of the Florida Keys and throughout the Caribbean (Acosta, 2001; Haywick & Mueller, 1997). Dustan and Halas (1987) documented *Palythoa* spp. in ten of eleven transects in 0–4 m water depth at Carysfort Reef. Surveys performed at Looe Key Reef in 1983 found an abundance of *Palythoa* spp. in 0–7 m water depth (Wheaton & Jaap, 1988). *Palythoa* spp. accounted for 11% of all cnidarians sampled in 0–11 m water depth by Wheaton and Jaap (1988). Finally, a recent survey revealed that *Palythoa* spp. cover 15% of Sombrero Reef backreef (Palandro, unpublished data collected in March 2007). Since *Palythoa* spp. have essentially the same spectral signal as scleractinian corals, it is likely that the coral habitat classification witnessed in the backreef may be largely due to *Palythoa* spp.

By performing a visual qualitative assessment on the progression of each habitat on the classified images, it was possible to detect a number of misclassifications. The highest occurrence of likely misclassification was between covered hardbottom and bare hardbottom. This can be attributed to the similarity between the two classes. Bare hardbottom has similar benthic constituents, but less abundance. The second highest rates of misclassification occurred between sand and bare hardbottom. This appears logical since these classes have the highest reflectance (Fig. 3). Conversely, there were no observed occurrences of coral habitat and sand misclassifications.

Several images produced more inconsistencies than others within the time series. For example, the 1998 image results were inconsistent for four of the five reef sites covered by that image (no image data were available for Conch Reef due to cloud cover). This may be due to image-scale environmental conditions that made the pre-processing (atmospheric and water-column correction) less effective. The 1998 image data has the highest *K*ₜ variance among reef sites. We further discuss the possible influence of the Florida Bay waters on the processing and results.

The Lower Keys sites showed the highest abundance of coral habitats, followed by the Upper and Middle Keys sites, respectively. This observation is in agreement with both historic (Shinn et al., 1989) and current studies (Beaver et al., 2006). The latitudinal pattern is attributed to the relative influence of Florida Bay on the three regions. The outflow of nutrient-rich and turbid waters from Florida Bay has traditionally explained the lower coral cover (Porter et al., 1999; Shinn et al., 1989) at Sombrero Reef (Middle Keys) and Conch Reef (the southernmost Upper Keys reef, thus easily influenced by Florida Bay waters).

4.2. Observations for individual reefs

As presented above, we observed trends for the entire Florida Keys. However, the results also show a large range of individual coral cover values and percent change over time, even for sites within a single region or on close geographic proximity. We also noticed six individual instances of coral habitat increases, but none were
significant. Western Sambo had the second highest percent coral habitat (22%) in 1988. This positively biased the overall percent coral habitat for that year, resulting into an increase compared to 1986. However, with the hindcast correction applied for Western Sambo coral habitat cover for 1984 and 1986, the overall change between 1986 and 1988 appears negative.

Looe Key Reef showed the lowest loss coral habitats (38%). Miller et al. (2002) analyzed loss in acroporid coral species (A. palmata and A. cervicornis) on Looe Key Reef from 1983 to 2000 and found a decline of 93% and 98%, respectively. However this was solely for the A. palmata and A. cervicornis, which were already scarce in 1983 (6.8% and 2.7% of total percent cover, respectively) (Wheaton & Jaap, 1988).

Porter and Meier (1992) showed average loss on Looe Key Reef to be 29% for all coral species between 1984 and 1991, with an average loss per year of 4.1%. CREMP data showed a decline in coral cover of 26% (3.3%/year) loss of coral cover between 1996 and 2004. Both estimates are in the range of the Landsat-derived coral habitat loss of 2.1%/year. This rate is the lowest among reef sites. Due to Looe Key Reef's location, adjacent to a large gap between Keys, it should be one of the most affected reef sites in the Lower Keys. Similarly, Sombrero Reef (Middle Keys) has the second lowest rate of coral habitat loss (45%) while also being directly influenced by Florida Bay. Conversely, Western Sambo (Lower Keys) displayed the highest loss (72%) from 1984 to 2002. From this, it is difficult to point to Florida Bay discharge as the main driver of the changes. The complex patterns may be due to environmental gradients (e.g., sedimentation) acting on scales larger than the reef sites but smaller than the regions (Murdoch & Aronson, 1999).

In situ studies found a decline in coral cover at Carysfort Reef averaging from 20% (2.9%/year) for 1984–1991 (Porter & Meier, 1992) to 72% (9.0%/year) for 1996–2004 (CREMP) to 90% (3.5%/year) for 1974–2000 (Dustan, 2003). Remote sensing studies on Carysfort Reef estimated coral habitat loss at 89% for 1981–2000 (4.7%/year) from IKONOS and aerial photography (Palandro et al., 2003a) and 88% (5.5%/year) for 1984–2000 from Landsat (Palandro et al., 2003b). This study estimated coral habitat loss at Carysfort Reef at 72% (4.0%/year). The different results obtained from the two Landsat studies were due to the different thresholds used to define the coral habitat class and different training pixels. The Carysfort Reef studies showed the sensitivity of the analysis to the different methods, in situ and remote sensing based. We found that the results were comparable between studies.

5. Conclusions

Twenty-eight (28) Landsat images from 1984 to 2002 were geographically, atmospherically, radiometrically and bathymetrically corrected to measure habitat changes for eight coral reef sites in the FKNS. Although most of the image data provided a logical ecological succession of classified habitats, some were spatially incoherent and ecologically unlikely. Overall, results confirmed a loss in coral habitat, which was strongly correlated to the in situ CREMP live stony coral cover decline for the same sites. The correlation between the two different datasets showed that changes witnessed at the smaller in situ monitoring scale are mirrored at the larger habitat scale.

The Florida Keys case study cannot be generalized for other sites worldwide. The good agreement between in situ and remote sensing interpretation may not be a rule. However, it suggests that local scale measurements of loss (here coral cover, but possibly also richness) can be set in a spatially broader context of local habitat dynamics and fragmentation. Time series of images are scarce at decadal scales. At best using Landsat, two decades can be captured, and possibly more with local aerial photographs. However, despite the short time scale, this helps establishing an accurate baseline to determine whether the system is degrading, recovering or stable. More work along the lines presented here should be conducted worldwide, and extended in the future with new high resolution sensors.

Acknowledgements

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References
