REMOTE SENSING TOOLS TO QUANTIFY ECOLOGICAL IMPACTS OF SEA LEVEL RISE ON BARRIER ESTUARIES

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Presented at the 20th NSW Coastal Management Conference, 3-6 November 2011, Tweed Heads, NSW, Australia.

Abstract

Changes in sea level may result in extensive changes to estuarine ecosystems. Increases in sea level are predicted to impact bathymetry, sediment transport, water quality and estuarine macrophyte distribution and abundance. Barrier estuaries provide a model system for studying sea level rise impacts, and can be used in comparative studies, as changes to the estuary entrance impact the same estuarine processes as expected with the sea level increase with undisturbed barrier estuaries to illustrate the ecological impact of sea level changes. Trends in water quality, bathymetry and macrophyte extent can be estimated from archival satellite data which is available in a range of sensor types and spatial resolutions. Previously a cost-effective spatially comprehensive, standardised and objective approach to analysing historical satellite data was developed for a semi-pristine barrier estuary, Wallis Lake, NSW. This project demonstrated that monitoring of estuarine systems across multiple sensors and multiple resolutions is possible. Retrospective analysis of archival satellite imagery illustrates changes in estuarine macrophyte extent. Demonstrating the potential for satellite image analysis to use estuarine macrophytes as indicators of estuarine ecology and morphology change expected with increasing sea-level rise.

Background

Estuarine macrophytes, such as seagrass, saltmarsh and mangroves, are critical components of estuarine ecology due to their role in providing habitat, improving water quality, contributing to the food chain and binding sediments, therefore stabilizing the estuarine morphology. Future climatic change, particularly sea-level rise (SLR), resulting in enhanced estuarine marinization, is expected to have a range of repercussions to environmental processes occurring in NSW estuaries. Higher sea levels may lead to changes in bathymetry, sediment transport, water quality and estuarine macrophyte distributions, particularly in areas of restricted tidal flow (Day et al. 2008).

Enhancement of the tidal head from SLR will result in increases in water salinity higher up in the headwaters of the estuary. This will lead to the redistribution of aquatic vegetation habitats to stay within salinity tolerance zones, such as the replacement of freshwater vegetation with seagrass (Short et al. 1999). It may also lead to die-back of less salt tolerant vegetation. For example, at the Gippsland lakes, increases in water salinity has resulted in the die-back of fringing reed-beds, thus leading to accelerated shore erosion by exposing the highly erodible shoreline substrate to wave-action (Dept. of Sustainability and Environment, 2003).

Changes in tidal height, -range and –velocity will have a dramatic effect on the estuarine ecology. Tidal height will lead to increases in water depth, which will reduce the amount of light reaching existing seagrass beds, thus reducing productivity (Short and Neckles 1999). Tidal velocity will change the sediment mobilization and transport regime of the estuary. This can cause, for example, the erosion of seagrass beds or deposition, allowing seagrass colonization. The ensuing increases in estuarine turbidity will reduce light reaching submerged macrophytes, thus reducing productivity. Changes in water motion will affect biomass, pollination and larval recruitment in seagrass beds (Williams 2003).

Changes in tidal ranges will affect the positions of the landward and seaward extremities of macrophyte zones. Howe et al. (2010), for example, observed that increases in the tidal range in an estuary of the Hunter river, which had the natural tidal flow reinstated, changed the vegetation zonation significantly over a period of 12 years. Habitats at the upper end of the topographic gradient (pasture and saltmarsh) shifted upslope while mudflat habitats at the lower end of the topographic gradient shifted downslope due to erosion. Mangroves colonized the intermediate elevation range. If accretion rates are outpaced by the rate of SLR, saltmarshes will be inundated and become vulnerable to mangrove encroachment, which will lead to reduced habitat diversity and wetland productivity (Day et al. 2008). Saltmarsh habitat may also undergo accelerated erosion due to loss of stabilizing vegetation.

Monitoring changes in the position and extent of key estuarine macrophytes can serve as an indicator for estimating changes in estuarine ecology and morphology expected with SLR. Several techniques exist to monitor the changes in the extent and locations of estuarine macrophytes, including *in situ* observations, aerial photographic interpretation (API) and satellite image-based analysis.

Although API has been extensively used in the past to map the cover of estuarine macrophytes in NSW (West et al. 1985, Saintilan and Williams 1999, 2000, Evans and Williams 2001), work was reported to take at least 10 person years to produce (West et al. 1985, West et al. 2008). Some trends in macrophyte distribution have been reported but several discrepancies in the trend analysis have been attributed to interpretation errors and operator bias (Williams and Meehan 2004, Meehan et al. 2005, Kelleway et al. 2007).

Improved data access coupled with advances in sensor and computing technology have increased quantitative use of airborne and satellite remote sensing data for synoptically assessing the extent of estuarine macrophytes. Multispectral satellite image classification (recently reviewed in Lu and Weng 2007) offers a viable alternative to API for estuarine monitoring. Previous work has shown that satellite image classification can be successfully implemented for retrospective change detection of estuarine macrophytes (e.g. Dekker et al. 2005, Gilman et al. 2007, Paling et al. 2008, Akumu et al. 2010).

Satellite monitoring of submerged areas is often hampered by water quality, governed by a range of different types of water column constituents. Additionally, there are many environmental variables that impact the quality of satellite imagery, such as atmospheric and water-surface conditions. Without sophisticated methodologies, such as glint correction and atmospheric correction, the imagery can be difficult to use for reliable trend assessments.

High spatial resolution satellite imagery offers a new paradigm in monitoring coastal resources over large areas, as the spatial coverage can provide contextual information that would be very expensive to otherwise obtain (Mumby and Edwards, 2002). Although high resolution imagery offers data on an environmentally relevant scale (Lim et al. 2009), it requires a standardised approach to monitor temporal change. An objective methodology is required to reliably assess changes within the environment (Saintilan and Williams, 1999, Dekker et al. 2005, Mishra et al. 2006, Saintilan et al. 2006, Anstee et al. 2009), as human interpretation error or operator bias is one of the most difficult error sources to quantify (Lunetta et al. 1991).

By implementing trend analysis from historic and current satellite image data, the ecological impacts of SLR on NSW barrier estuaries can potentially be forecasted by monitoring changes induced by enhancements to estuarine entrances. Manipulation of barrier estuary entrances can provide an artificial history of sea level rise impacts on estuarine macrophytes, especially the effects of increasing tidal range, higher tidal velocities and changes in water quality.

For this purpose, Wallis Lake, an immature barrier estuary with extensive areas of estuarine macrophytes (Evans and Williams 2001, Witter et al. 2009), located ~360km north of Sydney on the central coast of New South Wales, was selected. The entrance to Wallis Lake was enhanced in 1966 by the installation of a second training wall. A dramatic increase in tidal ranges was observed following the construction of the training wall, resulting in an unstable scouring of the entire estuarine system (Nielsen and Gordon 1980). This instability has not reached a new equilibrium yet (Nielsen and Gordon 2008), providing us with the opportunity to observe and track the impacts of this increased tidal range over time, in lieu of SLR.

Methodology

Field data collection

A field campaign was conducted at representative sites around the Lake in May and September 2008 (Figure 1). Field spectra and geographic location (using a differential-global positioning system) of targets were recorded. Targets included representative saltmarsh, mangrove, macroalgae and seagrass species as well as abiotic backgrounds such as sand and mud. Georeferenced field observations were recorded on species found covering a minimum area equivalent to that of a QuickBird pixel (2.6m x 2.6m).

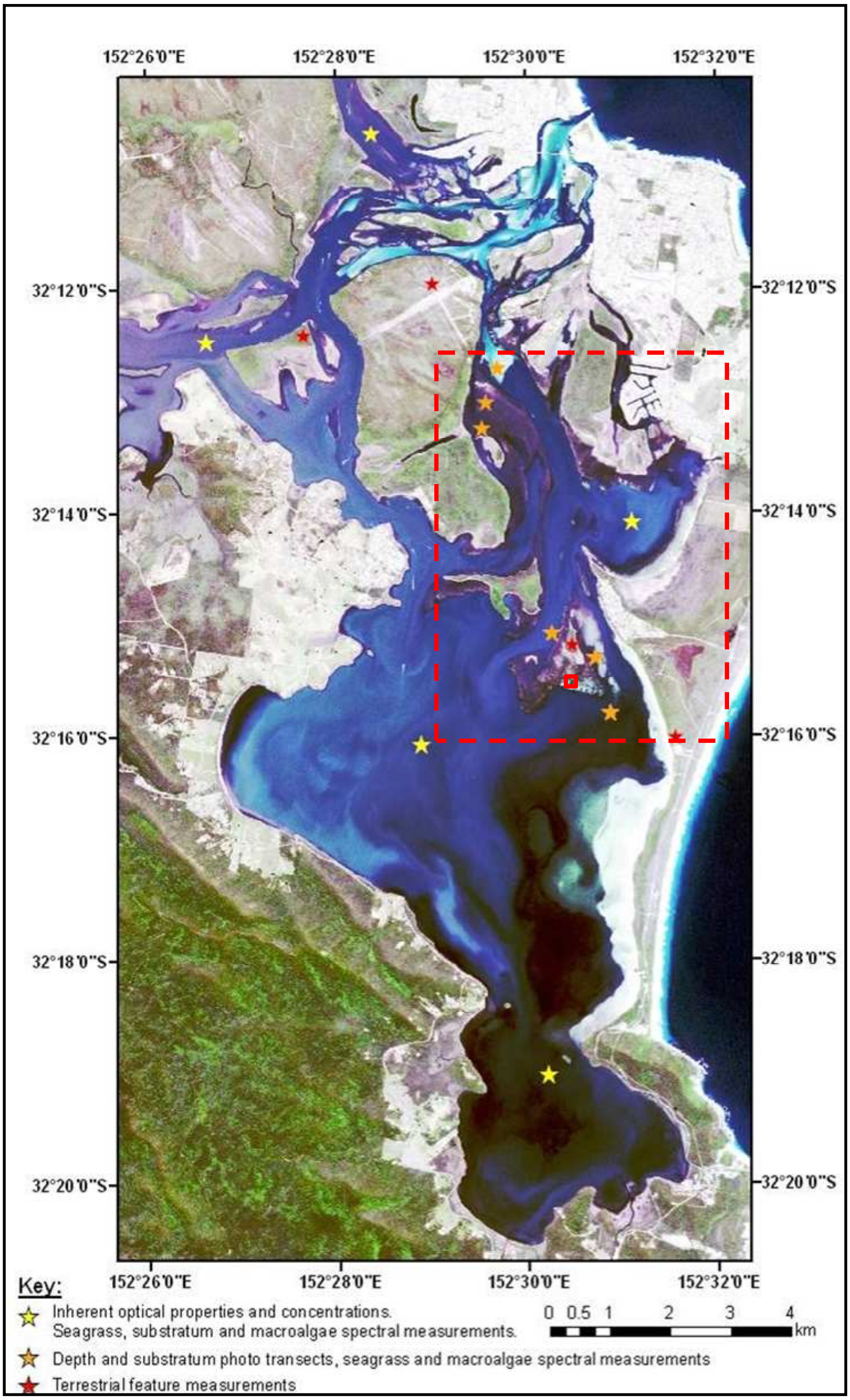

Figure 1 Location of Wallis Lake and the field sites visited in May and September 2008. Dashed red line represents the area presented in this study. Solid red square represents the location of a small shallow subset, selected to demonstrate differences in spatial resolution in macrophyte mapping.

Image acquisition and processing

Satellite images with a range of spectral and spatial resolutions, spanning a period between 2002 and 2010, were available for the study area (Anstee et al. 2011).

Table 1 Specifications of the archival satellite images available for multitemporal analysis of Wallis Lake macrophytes

Sensor	Date	Spatial resolution	Spectral resolution	Source	Comments
Landsat ETM+	12/09/2002	30m	4 VNIR, 1 SWIR, 2 ThIR	CSIRO image archive	Sensor striping
QuickBird	24/01/2003	2.6m	4 VNIR	CSIRO image archive	Partial coverage
ALOS AVNIR-2	5/01/2007	10m	4 VNIR	CSIRO image archive	Good quality
QuickBird	23/02/2008	2.6m	4 VNIR	Specifically tasked for the project	Partial coverage, lake glint affected, land good quality
QuickBird	16/09/2008	2.6m	4 VNIR	Specifically tasked for the project	Good quality
Landsat TM5	20/09/2008	30m	4 VNIR, 1 SWIR, 2 ThIR	USGS image archive	Good quality
WorldView-2	20/08/2010	1.8m	8 VNIR	DigitalGlobe	Good quality

To ensure accurate trend assessment across multiple images, all images underwent pre-processing (co-registration, atmospheric correction, glint and air/water interface corrections as required, segmentation) following a standardized processing pathway to allow for comparisons between satellite data (Figure 2).

As a first step a physics-based atmospheric correction procedure, the ‘coastal Waters and Ocean MODTRAN-4 Based ATmospheric correction’ (‘c-WOMBAT-c’) was implemented to retrieve both surface and subsurface remote-sensing reflectance (Brando and Dekker, 2003, Phinn et al. 2005). c-WOMBAT-c applies a full MODTRAN-4 atmosphere parameterisation, based on radiosonde data from the Australian Bureau of Meteorology Station at Williamtown Airport (approximately 85km to the south-west of the study site) and ozone content obtained from the Total Ozone Mapping Spectrometer (TOMS) database (http://toms.gsfc.nasa.gov/ozone/ozone.html). Adjacency effects were corrected for by using an averaged surface radiance for the surrounding region. This spatially weighted image is generated by convolving the input radiance imagery with a one square kilometre spatial weighting function (Adler-Golden et al. 1998).

To ensure that each pixel is collocated in each image, it was necessary to co-register each image to a base image. The September 2008 QuickBird image was chosen for this purpose. A full description of the pre-processing pathway and reporting of image registration accuracy can be found in Anstee et al. (2009).

Submerged vegetation can only be mapped where the substratum is visible below the water column. For this reason, the aquatic portion of the area of interest was segmented into optically deep (no bottom visibility) and optically shallow (substratum visible) sections and only the optically shallow areas was analysed for subtidal macrophytes.

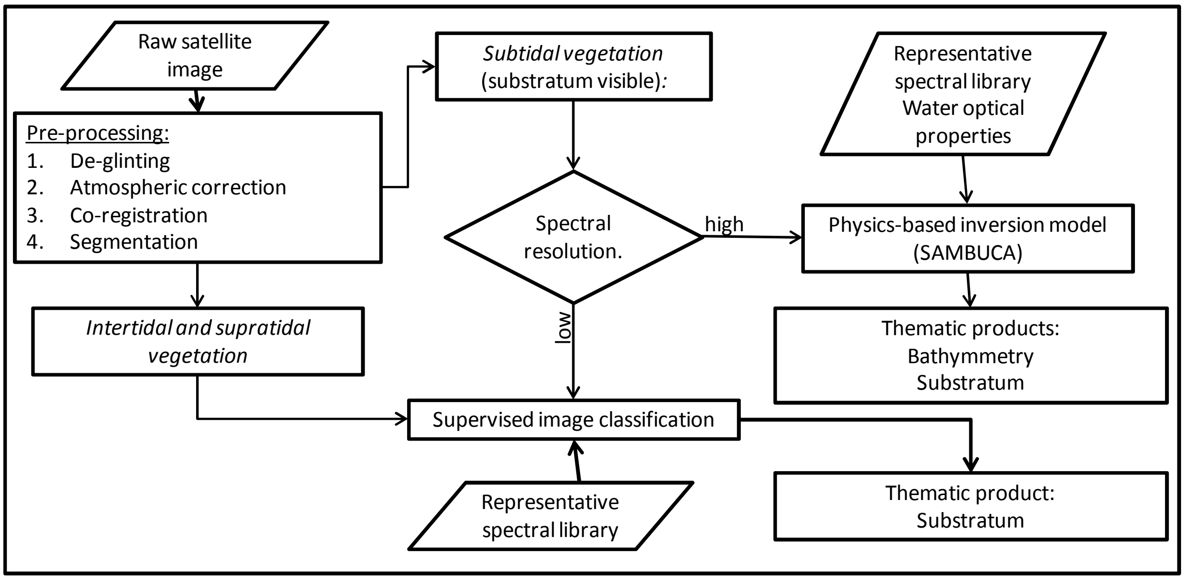

Figure 2 Standardized image processing pathway implemented on all satellite data for retrospective analysis

Benthic Substrata Classification

After removal of atmospheric and glint effects and applying an air/water interface correction, the optically shallow subtidal regions for the target area was analysed with either a semi-analytical model for bathymetry un-mixing and concentration assessment (SAMBUCA) (Brando et al. 2009) or a standard K-means classification (Lu and Weng 2007). Only the higher spectral resolution of the WV-2 image was sufficient for the application of SAMBUCA while the more limited spectral resolution of the other archival satellite images was suitable only for the application of a K Means classifier (Figure 2).

In order to retrieve substratum cover type and bathymetry using a physics-based approach, it is necessary to model the water's optically active constituents (that are measureable water properties) which can be estimated from remotely sensed data. SAMBUCA is an enhanced implementation of the inversion/optimization algorithm by (Lee et al. 1998, Lee et al. 1999, Lee et al. 2001) which utilises the remote-sensing reflectance, r_{rs}, acquired after radiometric and atmospheric correction of the remotely sensed imagery. SAMBUCA was applied to retrieve simultaneous outputs of: 1) the water's optically active constituent concentrations (chlorophyll-a, CDOM and NAP), 2) the percentage substratum cover type (either as homogeneous or mixtures of 2 different types) and 3) metrics to assess the reliability of the retrieval.

The water's optically active constituents which can be estimated from remotely sensed data are modelled in SAMBUCA. For each of the optically active constituents, the absorption and backscattering coefficients are modelled and the algorithm retrieves the concentration for each constituent.

To demonstrate the standardized image processing pathway to capture temporal change in seagrasses, a small shallow subset was selected (Figure 1). The subsetted region was important as it contained *Posidonia* beds known to be extremely stable over extended periods of time (Laegdsgaard 2001, West 1985). The unsupervised K-means classification implemented, this clusters the pixels into classes using a minimum distance technique (Tou and Gonzalez, 1974). Changes in *Posidonia* extent within the small subsets were determined using a standard post-classification change-detection technique.

Intertidal and supratidal substratum classification

The intertidal and supratidal regions of each image underwent a standard Spectral Angle Mapper (SAM) supervised classification. SAM is an automated method that permits rapid mapping of spectral similarity of image spectra to field spectra (Yuhas et al. 1992, Kruse et al. 1993, Maritorena et al. 1994). This technique is relatively insensitive to illumination and albedo effects. SAM compares the angle between the endmember spectrum vector and each pixel vector in n-dimensional feature space, where n is the number of spectral bands. Smaller angles represent closer matches to the reference spectrum. Endmember spectra were extracted directly from each image as Region of Interest (ROI) average spectra, based on geo-located field observations where relatively uniform coverage of each vegetation class could be found. Classification accuracy was estimated using standard post-classification statistical analysis tools (confusion matrix), implemented on a validation dataset of georeferenced *in situ* observations.

The API map of the distribution of estuarine vegetation, prepared in 1985 (West et al. 1985), was used as a basis for assessing trends in saltmarsh and mangrove extent in the study area. Two archival satellite images, (a QB image, acquired in January 2003, and a WV-2 image, acquired in August 2010) were used to demonstrate the ability of the standardized image processing pathway to capture temporal change in estuarine vegetation cover. Changes in saltmarsh and mangrove extent within the polygons of the 1985 map were determined using a standard post-classification change-detection technique (confusion matrix).

Results

Benthic substratum classification

Small changes in the sizes of the *Posidonia* beds could be seen within the QuickBird 2003-2008 images and with the higher WorldView-2 spatial resolution and were mapped accordingly (Figure 3). The changes in the coarser ALOS data could not be as easily assessed. Statistical analysis of region shows a 4.5% loss of *Posidonia* from the February 2008 to the September 2008 but an increase of 0.5% from September 2003 until September 2008 (Table 2). Looking at only the QuickBird imagery, over the medium term (five and a half years), *Posidonia* has shown to be relatively stable with a slight increase of 0.5% and seasonally a small decrease 4.5%.

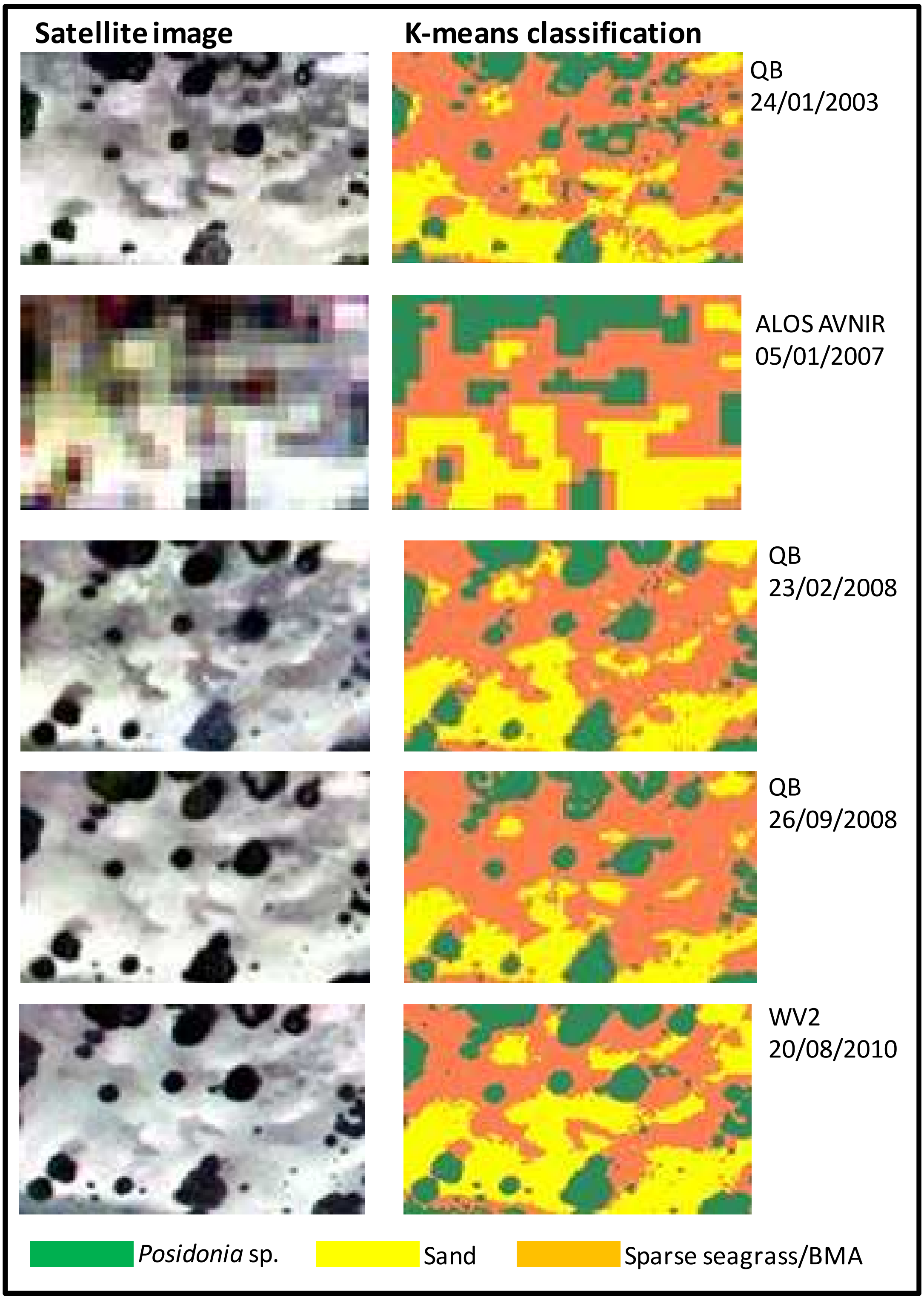

Figure 3 True colour satellite images showing a subset of patches of *Posidonia* and sand (located within the solid red square on Figure 1), extracted from archival QB, WV2 and ALOS AVNIR satellite data of Wallis Lake and the K means unsupervised classification mapping of the *Posidonia*, sand and a sparse seagrass/BMA sand class of each image.

Table 2 Summary of statistical analysis, showing percentage of change in each class, calculated for the K Means classification of image data presented in Figure 3. Results in red are to be treated with caution.

Interval	**Initial image**	**Final image**	**Change in target (%)**			**Comments**
			Posidonia	**Sparse Seagrass/BMA**	**Sand**	
Medium term (5 years)	QB Jan 2003	QB Feb 2008	+3.2	-6.9 (becomes sand and *Posidonia*)	+18.8	Cover of *Posidonia* is stable, but some SS/BMA becomes sand
Medium term (5.5 years)	QB Jan 2003	QB Sept 2008	+0.5	+6.4	-14.3	Cover of *Posidonia* is stable, but some SS/BMA becomes sand
Seasonal (7 months)	QB Feb 2008	QB Sept 2008	-4.5	+14.5 (becomes sand)	-28.1	Some short term reduction in size of bed
Short term (1 year)	ALOS AVNIR Jan 2007	QB Feb 2008				results pending
Medium term (1.5 years)	ALOS AVNIR Jan 2007	QB Sept 2008				results pending
Medium term (4 years)	QB Jan 2003	ALOS AVNIR Jan 2007				results pending

Intertidal and supratidal substratum classification

The saltmarsh class in this study was mapped with high accuracy in most of the available archival satellite images (75% - 95% accuracy, Anstee et al. 2009) while the mangrove vegetation class was confused spectrally with swamp oak and other forest classes in the SAM classification in some of the classified images (Anstee et al. 2009). The lower spatial and spectral resolution of the imagery used in the assessment contributed to the mixing (mixels) of the spectral response.

Saltmarshes are mostly restricted to the upper intertidal environment, generally between the level of the mean high tide and the mean spring tide (Saintilan, 2009). The boundary between mangrove and saltmarsh zones is normally sharp and mangrove seedlings are rarely found in within the saltmarsh zone in NSW (Clarke and Myerscough, 1993). Saltmarshes are vulnerable to mangrove encroachment in an unstable estuarine environment where the accretion rate is outpaced by the rate of SLR. Therefore, changes in saltmarsh extent and position can be used as an indicator of environmental changes attributed to SLR.

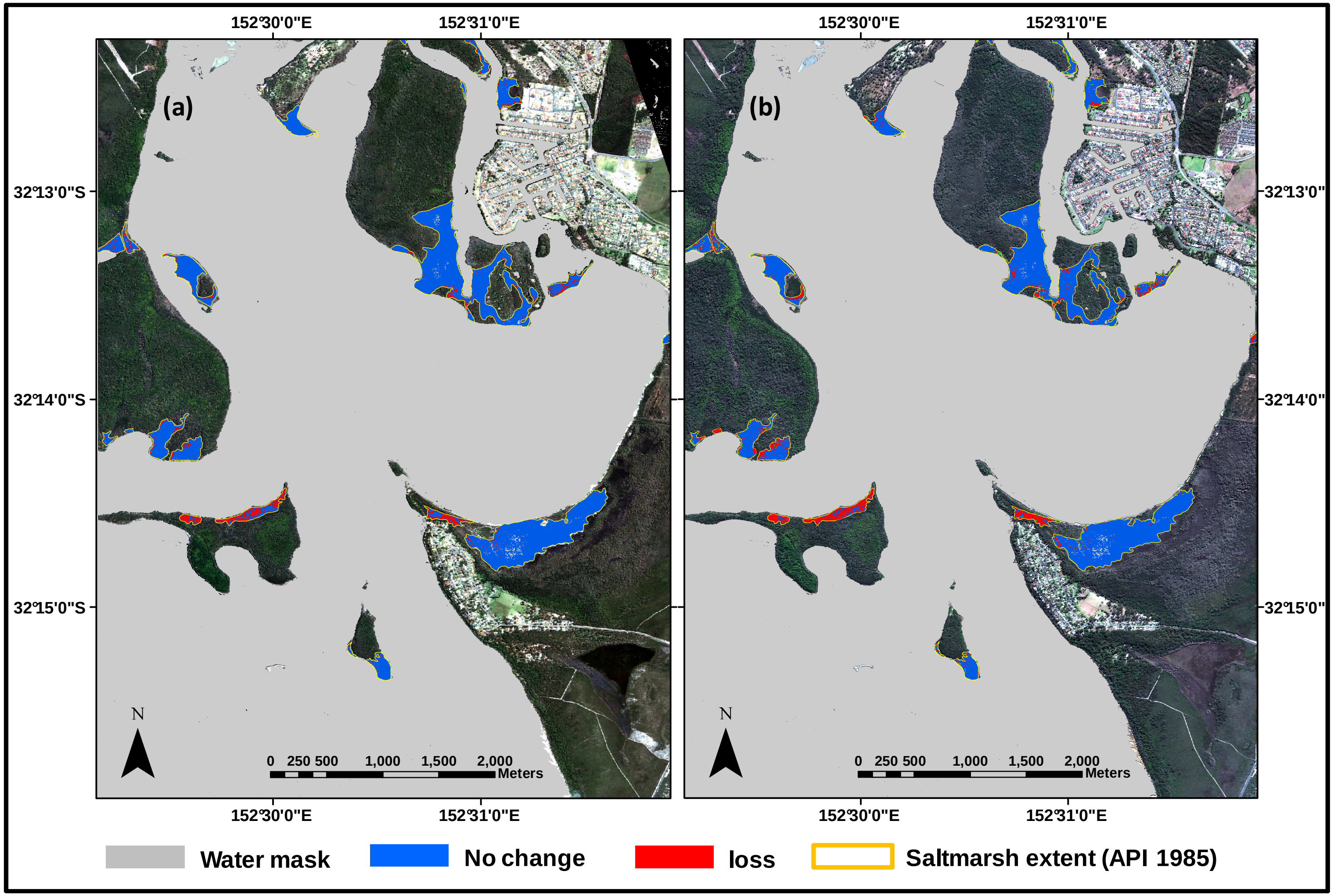

Figure 4 Changes in saltmarsh extent of a subsection of Wallis Lake since 1985, when a map was prepared from aerial photographs (black polygons), captured with (a) a QuickBird image, acquired in January 2003 and (b) a WorldView-2 image, acquired in September 2010.

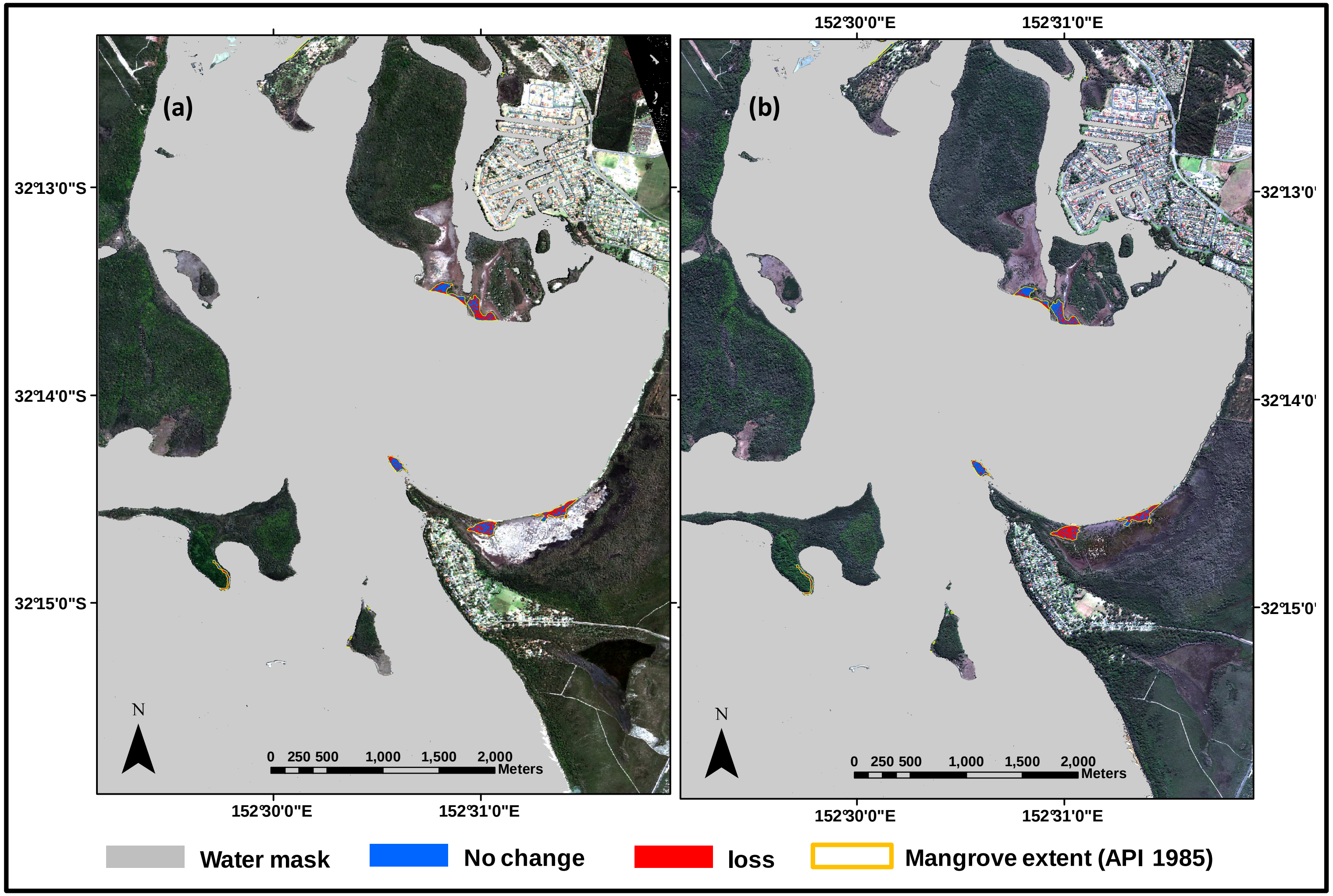

Figure 5 Changes in mangrove extent of a subsection of Wallis Lake since 1985, when a map was prepared from aerial photographs (black polygons), captured with (a) a QuickBird image, acquired in January 2003 and (b) a WorldView-2 image, acquired in September 2010

The API baseline data, used in this presentation (West et al. 1985), were created with the *camera lucida* technique by analysing aerial photos taken in the late 1970s and early 1980s. This methodology, while appropriate for its time in relation to its management objective, had systemic operational errors that reduced its accuracy. These errors are discussed by Williams et al. (2003) and Meehan et al. (2005), and relate to factors such as scale of aerial photos (1: 25,000, 1: 16,000), and width of the drawing implement used to trace polygons from the photos to an adjacent map sheet (1: 25,000). Any apparent changes in saltmarsh and mangrove extent observed in Figure 4 and Figure 5 should thus be interpreted in this light.

From the results it appears that there are areas around the shores of Wallis Lake where saltmarsh disappeared and was replaced by trees (see Figure 4) while areas where mangroves were apparently historically present are now absent (see Figure 5). Kelleway et al. (2007) highlighted the difference between API mapping and field surveys for the presence and distribution of saltmarsh wetlands in the Parramatta River. They found that saltmarsh extent was not suitably captured on a micro-scale in the 1985 API map due to the quality of the black and white aerial photographs and the interpretation techniques available at the time. It is thus possible that results presented in figures 4 and 5, may not be real changes from 1985 to 2003 and 2010 but an artefact of the mapping technique implemented at the time. Medium-term changes (five-year scale), based on satellite image analysis, will also be presented at the conference.

Discussion and Conclusions:

Retrospective analysis of archival satellite imagery illustrates changes in estuarine macrophyte extent. Demonstrating the potential for satellite image analysis to use estuarine macrophytes as indicators of estuarine ecology and morphology change expected with increasing sea-level.

Moderate resolution imagery was found to be limited in the determination of extent, type or health of NSW estuarine macrophytes due to the lack of spatial resolution. However it does have the unique advantage of presenting a decade-scale archive for establishing the rate and extent of broad-scale vegetation change over time. Fine-scale aerial photography has been used to determine the extent of estuarine macrophytes, implementing the *camera lucida* technique. However this proved to be time-consuming and prone to operator bias.

Fine spatial scale QuickBird imagery used in conjunction with a supervised image classification technique was evaluated in this study as an additional source of data for the preparation of maps of estuarine macrophytes. Furthermore, higher spectral and spatial resolution images from WorldView-2 shown that species level discrimination of seagrasses are possible with almost twice the classification accuracy of QuickBird (Anstee et al. 2011).

The coarser spatial resolution Landsat (25m resolution) and ALOS (10m resolution) images in initial trials performed similarly in terms of classification accuracy with the finer scale QuickBird (2.4m resolution) images (final results pending). This may indicate that higher spatial resolution is not always required for trend assessment. Each future mapping project has to be assessed individually to determine what spatial scale will provide the optimal information needed for decision-making purposes whilst minimizing costs.

It is evident that the spectral resolution of existing and historical satellite sensors limits the reliability of the supervised classification technique implemented in this study. However, sensors with higher spectral resolution have shown to enhance the spectral separability of the estuarine macrophytes. Inclusion of pattern recognition may enhance the effectiveness of mapping estuarine macrophytes from existing multispectral satellite images, especially from high spatial resolution data (Lu and Weng 2007, Brando et al. 2009).

We propose to apply this method to replicated enhanced barrier estuaries and compare the observed trends with undisturbed barrier estuaries in NSW to forecast possible changes in bathymetry, water quality and macrophyte extent in estuaries state-wide.

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