TWEED RIVER ESTUARY

ESTUARINE VEGETATION MONITORING PROGRAM

FINAL REPORT
DECEMBER 2012

Prepared for NSW Department of Trade and Investment, Crown Lands, and Queensland Department of Science, Information Technology, Innovation and the Arts

Pacific Wetlands Environmental Consultants

Executive Summary

A trend of increasing mangrove extent (11.68%) and decreasing saltmarsh extent (25.93%) was identified in the raw figures for the period 2000-2012, and the changes were deemed to be higher than the error range of the methodology. The increase in mangrove is consistent between years and common to all subsections of the estuary, representing an annual increase of 0.97%. This increase is most probably the result of estuarine sedimentation and elevated water levels, and is common in most estuaries in the region (Saintilan 2003; Saintilan and Rogers in press). Saltmarsh decline has been measured at a rate of 2.16% per annum (25.93% in total), and over the whole estuary was constant between mapping periods. Though saltmarsh loss to mangrove encroachment is common within estuaries in NSW (Saintilan and Williams 1999) and predated the sand bypassing project, this rate of loss is on the high end of decline in the region and on current trends saltmarsh will be lost from the Tweed by 2060.

The proliferation of mangroves in the Tweed River has been documented by Saintilan (1997, 1998), and forms the background against which results of this survey are to be interpreted. Saintilan (1997) found, in contrast to other sites in NSW, little evidence of upslope encroachment of mangrove upon saltmarsh. However, local occurrences of this trend are evident on the Tweed. Wilton (2002) found from aerial photographs of Ukerebagh Island dating from 1948 to 1998, which indicate between 1961 and 1998 a trend of mangrove encroachment on saltmarsh, with saltmarsh declining by 20% and mangrove increasing extent by 20% over the same period. Saltmarsh decline on Ukerebagh has continued at the same absolute rate, though this represents a proportional increase as the area of saltmarsh has diminished.

Seagrass was found to have increased in extent in the period 2000-2012 by approximately 20.08%, a figure well in excess of the estimated error of mapping in relation to seagrass in this survey. Increase occurred between every survey, though most of the increase was in the period 2005-2007 with a slight decline to 2010 and a slight increase since. Such an increase is indicative of stable climatic, hydrological and geomorphic conditions. Seagrass extent was stable between 2002 and 2005. Minor flood events in June 2005, January 2008 and January 2012 did not cause significant declines in seagrass extent. The increase in seagrass extent is most likely the result of stable conditions in the estuary, following several years of reduced rainfall and storminess.

Analysis of vegetation structure from quadrats located within the saltmarsh zone between 2000 and 2012 indicate a substantial decline in the cover of the dominant saltmarsh species *Sporobolus virginicus*. *S. virginicus* declined in 2001, possibly as a consequence of drought, and had recovered by December 2002 and was again high in 2005 and 2007. There was a noticeable decline in percent cover in the 2010 survey to levels below those observed in 2001, and consistent thinning in 2011 and 2012, whereby a threshold seems to have been crossed. The cause of this thinning appears to be elevated water levels. The decline has corresponded to an eroding trend observed at the same location in recent surveys, and the SET platforms in the saltmarsh are indicating that the saltmarsh elevation is lagging behind water level

trends in the estuary. An assessment of long-term sedimentation trends on Ukerebagh Island is recommended, to determine whether this trend may be associated with reduced sedimentation rates in recent years.

Surface Elevation Tables (SETs) and feldspar marker horizons were installed in 2000 in the mangrove and saltmarsh zone to measure rates of subsidence and sedimentation. Surface elevation trends in the mangrove zone can be characterised by three periods of elevation gain (2000-2002, 2006-2008, 2010-2012), and two periods of elevation loss (2002-2006, and 2008-2010). Overall the trend has been one of elevation gain in the mangrove at a rate of 1.3 mm per year. Saltmarsh elevation changed by less than 1mm in 12 years. Elevation in the saltmarsh did not change significantly between 2000 and 2007, though increased to the 2008 survey before declining to 2012. Average increase in saltmarsh elevation for the period 2000-2012 was approximately 0.07 mm per year. By October 2007 there was no significant difference between mangrove and saltmarsh in the annualised rate of accretion, both environments averaging approximately 1.5mm per year. Following this, the mangrove environment accreted strongly to 2010 (reaching more than 20 mm above 2000 levels), while the saltmarsh eroded to approximately 8 mm above 2000 levels. Overall, mangrove seems to be responding well to increased water levels in the estuary, while saltmarsh has lagged behind, and this seems to have precipitated the decline of saltmarsh on Ukerebagh.

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1 Introduction

An Environmental Management System - Operations (EMS - Operations) was developed for the Tweed River Entrance Sand Bypassing Project to address environmental management issues associated with the ongoing operation of the sand by-passing system. The EMS contains a range of Environmental Management Plans (EMPs) that address key environmental performance issues. The EMS - Operations is currently being implemented by the NSW Department of Trade and Investment, Queensland Department of Science, Information Technology and the Arts and the Tweed River Entrance Sand Bypassing Company Pty Limited.

The EMS - Operations Sub-Plan B16 Tweed River Entrance and Lower Estuary Management Plan, has identified a number of monitoring requirements for the lower Tweed estuary including monitoring the distribution and health of wetlands in the lower estuary through the use of aerial photography mapping and periodic quadrant sampling, if required. The monitoring result will be compared to baseline data collected prior to the Bypassing system operation. As part of the monitoring program, aerial photography of the lower Tweed estuary was captured every six months from 2001 to 2003 and annually thereafter.

Pacific Wetlands were commissioned by the Tweed River Entrance Sand Bypassing Project to undertake mapping of mangroves, saltmarsh and seagrass communities within the lower Tweed estuary using aerial photography. The following report by *Pacific Wetlands* describes the methods by which these photographs have been utilised in determining the rates of change in mangrove, saltmarsh and seagrass vegetation cover, and comparisons of these rates to those determined in previous decades by Saintilan (1997), and previous years by Rogers et al. (2003; 2006). Further, the proposal describes high-resolution on-ground measurements of vegetation structure and wetland sedimentological process established on Ukerebagh Island in 2000, and continuing during the time of the establishment of the sand bypassing system. The Ukerebagh site is one of a network of sites monitored over the same time period in NSW and Victoria.

Aerial photograph interpretation is a relatively coarse tool for the assessment of vegetation change in mangrove and saltmarsh, particularly at the temporal scale of months to years. An advantage of quadrat vegetation sampling over, or in addition to, aerial photo interpretation is the capacity of on-ground measures to assess seedling and juvenile recruitment. This is of particular interest in the present situation, as successful recruitment of mangroves in the saltmarsh environment might be an early indicator of changes to the vegetation structure following alterations in tidal conditions in the estuary. These changes are not detected on the spatial scale at which aerial photograph interpretation is made.

For this reason, the consultants established a series of six vegetation plots in the saltmarsh environments of Ukerebagh Island in November 2000. Each plot contained a series of 6 randomly selected vegetation quadrats of 5 $\times$ 5 metres square. These plots were revisited in November 2001, December 2002, January 2006, October 2007, December 2010, November 2011 and October 2012, during which times replicate measures were taken of species composition, mangrove height, girth and crown foliage diameter, the number of mangrove seedlings and the percent cover of each saltmarsh species in each quadrat.

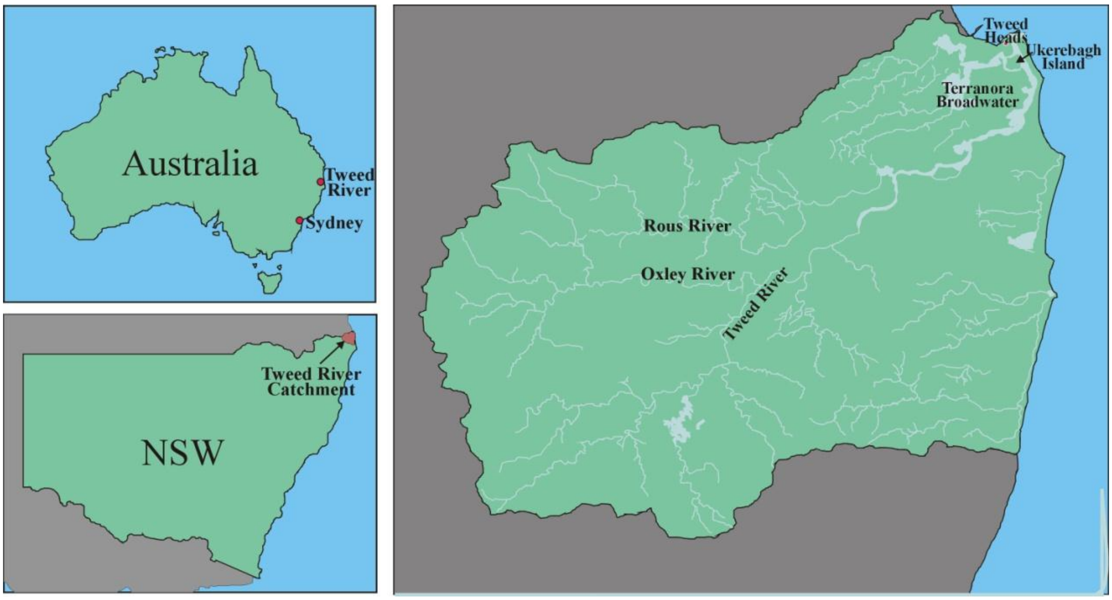

Figure 1: Location of the Tweed River and Ukerebagh Island.

Similar quadrats have been installed in the estuaries cited in Table 1. Rates of vegetation change on Ukerebagh Island can therefore be compared to 12 other sites in NSW and Victoria over the same period, providing a comprehensive context for the interpretation of the Ukerebagh Island data. The cost of this work was supported by Environment Australia under the Coasts and Clean Seas Initiative, and Tweed Shire Council.

Table 1: Study sites incorporated in the wider monitoring program undertaken by Pacific Wetlands

Region	Study Sites
Northern NSW	Ukerebagh Island, Tweed River
Central NSW	Kooragang Island, Hunter River
Sydney	Berowra Creek, Hawkesbury River Marramarra Creek, Hawkesbury River Homebush Bay, Parramatta River Towra Point, Botany Bay
Southern NSW	Currambene Creek, Jervis Bay Minnamurra River Cararma Inlet, Jervis Bay
Victoria	Kooweerup, Westernport Bay Rhyll, Westernport Bay Quaill Island, Westernport Bay French Island, Westernport Bay

2 Methods

The primary method undertaken as part of this study is photogrammetric mapping. Methods used in the wider monitoring program, undertaken by Pacific Wetlands, were also incorporated into this study and include vegetation sampling using a series of plots and quadrats and measures of surface elevation and sedimentation using surface-elevation table (SET) techniques.

2.1 Photogrammetric Mapping

The consultants used the protocols established in Wilton and Saintilan (2000) for the mapping of mangrove and saltmarsh communities in eastern Australia. The following procedures were applied:

1. Aerial photographs covering the area to be mapped were loaned to the consultants for the duration of the project. This area includes all mangrove, saltmarsh and seagrass areas on the Tweed River north of Barneys Bridge, for the 13 May 2000, 19 April 2002, 8 June 2003, 5 March 2005, 12 June 2007, 5 April 2010 and 3 April 2012 series. These air photographs were scanned and imported into the ArcView Geographic Information System (*ESRI Inc.*) at a resolution of at least 300 dpi.

2. Aerial photographs were georectified to correct distortions of scale caused by the varying distances of photographed objects from the camera lens. A minimum of 6 ground control points per photograph were used, all derived from recognisable fixed points on the CMA topographic maps of the region.

3. The on-screen digitising functions of ArcView (*ESRI Inc.*) were used to create polygons of discrete mangrove, saltmarsh and seagrass areas. The following criteria were used when mapping vegetation communities

   – Mangrove, saltmarsh, mixed mangrove and saltmarsh, and seagrass were differentiated on the basis of colour, texture, and the geomorphic and geographical context of the vegetation, as described at length in Wilton and Saintilan (2000).

   – Interpretations were cross-checked with the earlier surveys of West *et al.* (1985), Saintilan (1997) and by comparison of the 2000, 2002, 2003, 2005, 2007 and 2010 series.

   – Saltmarsh was defined as intertidal vegetation where a gap of greater than 30 metres exists between mangrove crowns. Higher crown densities were classified as mangrove.

2.1.1 Ground-truthing of intertidal vegetation communities

Ground-truthing is the process of determining the accuracy of mapped vegetation units by comparison with field observation. This is particularly important if different vegetation units present similar textures and colours in the air photograph, and the geomorphic context does not allow easy differentiation.

Along the intertidal gradient, it is often difficult to distinguish between saltmarsh and adjacent pasture grasses. Similarly, it is at times difficult to distinguish from aerial photographic evidence alone the similar textures of mangrove, Casuarina and Eucalypt forests. These problems were previously noted by Saintilan (1998).

Visiting pre-determined sites following the production of draft maps is a simple and cost-effective way of ensuring the accuracy of the boundaries and, therefore, the validity of conclusions drawn concerning the rates of change in the vegetation. For this reason, ground-truthing was considered an essential part of the mapping exercise.

Two days were spent checking interpretations of vegetation communities at approximately 30 points at four key locations on the Tweed River for the 2003 survey (Rogers et al. 2003). An additional day was spent ground-truthing four locations on the southern shores of the Terranora Broadwater. These points were determined following production of draft maps of the wetland vegetation. Further ground-truthing was conducted in January 2006, October 2007, December 2010, November 2011 and October 2012.

2.1.2 Methods of Analysis

The total area of mangrove, saltmarsh and seagrass were calculated for each of the geomorphic divisions of the Tweed River described in Saintilan (1997) (Figure 2), including the Cobaki Broadwater, Fluvial Channel, Terranora Broadwater and the Terranora Tidal Channel. These analyses were performed digitally using the Spatial Analyst application in ARCView.

Charts were drawn for 2000 to 2012 in comparison with the results of Saintilan (1997) for the five decades preceding 1995 for the Terranora Tidal Channel, and the combined broadwater areas of Cobaki Broadwater and Terranora Broadwater. Since the Fluvial Channel was not mapped to the same extent as Saintilan (1997), comparisons were not made.

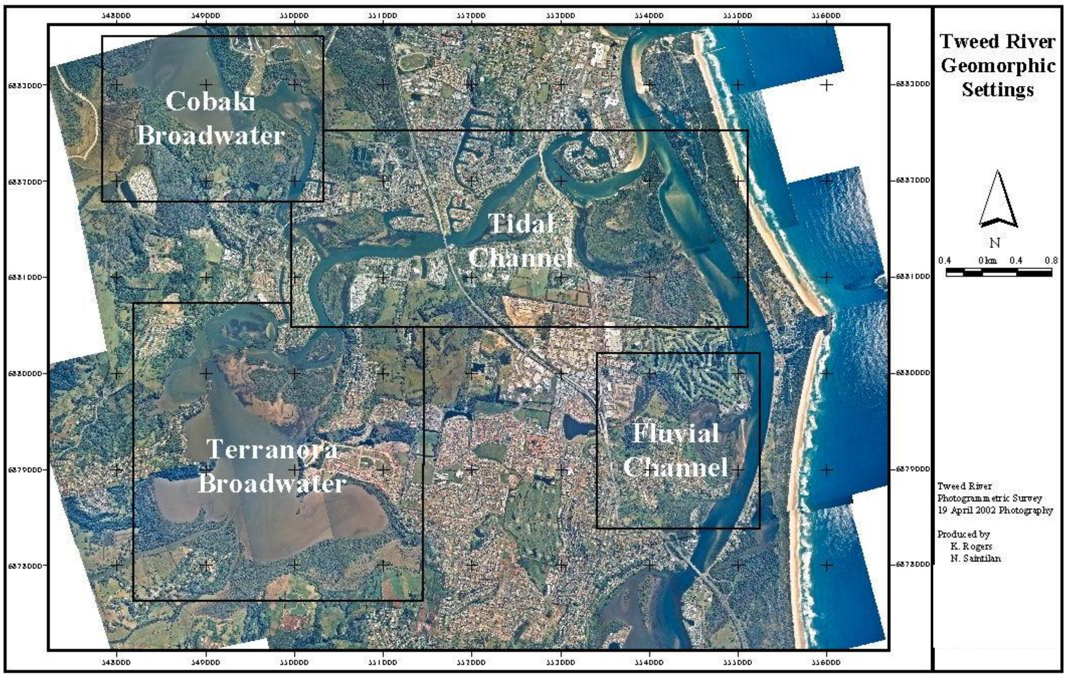

Figure 2: Geomorphic settings used for mapping comparisons, adapted from Saintilan (1997).

2.2 Vegetation Sampling

Prior to the commissioning of this study the consultants were engaged in monitoring vegetation and surface elevation characteristics of Ukerebagh Island for Environment Australia and the Tweed Shire Council. This study commenced in 2000 prior to the establishment of the Tweed River Entrance Sand Bypassing system. Results from this study bear directly upon the interpretation of data in this report, and so the methodology employed in this on-site monitoring is reproduced here.

Changes in estuarine wetland vegetation health are best assessed using plots and quadrats in a stratified manner to measure species composition and structural properties (Watkinson 1998). Based on the methods of Clarke (1993), plots were established in a stratified manner within the mangrove and saltmarsh vegetation at each study site. Six plots of 20m x 50m were established in November 2000 beside the SET monitoring stations. Within each plot, six random 5m x 5m quadrats were established. Monitoring plots within the mangrove and saltmarsh zone were resampled in December 2001, November 2002, January 2006 (saltmarsh plots only), October 2007 (saltmarsh plots) December 2010 (saltmarsh plots), November 2011 (saltmarsh plots) and October 2012 (saltmarsh plots).

2.3 Surface Elevation

To analyse surface elevation, sedimentation-erosion table techniques were employed on Ukerebagh Island by Rogers *et al.* (2002). The sedimentation-erosion table (SET) is an instrument originally created by Boumans and Day (1993) for the United States Geological Survey (USGS) to make high precision measurements of change in the surface elevation of intertidal and shallow sub-tidal environments. The confidence interval of the SET is ±1.5mm. The SET acts as a benchmark in space from which relative changes in the surface elevation can be determined (Figure 3).

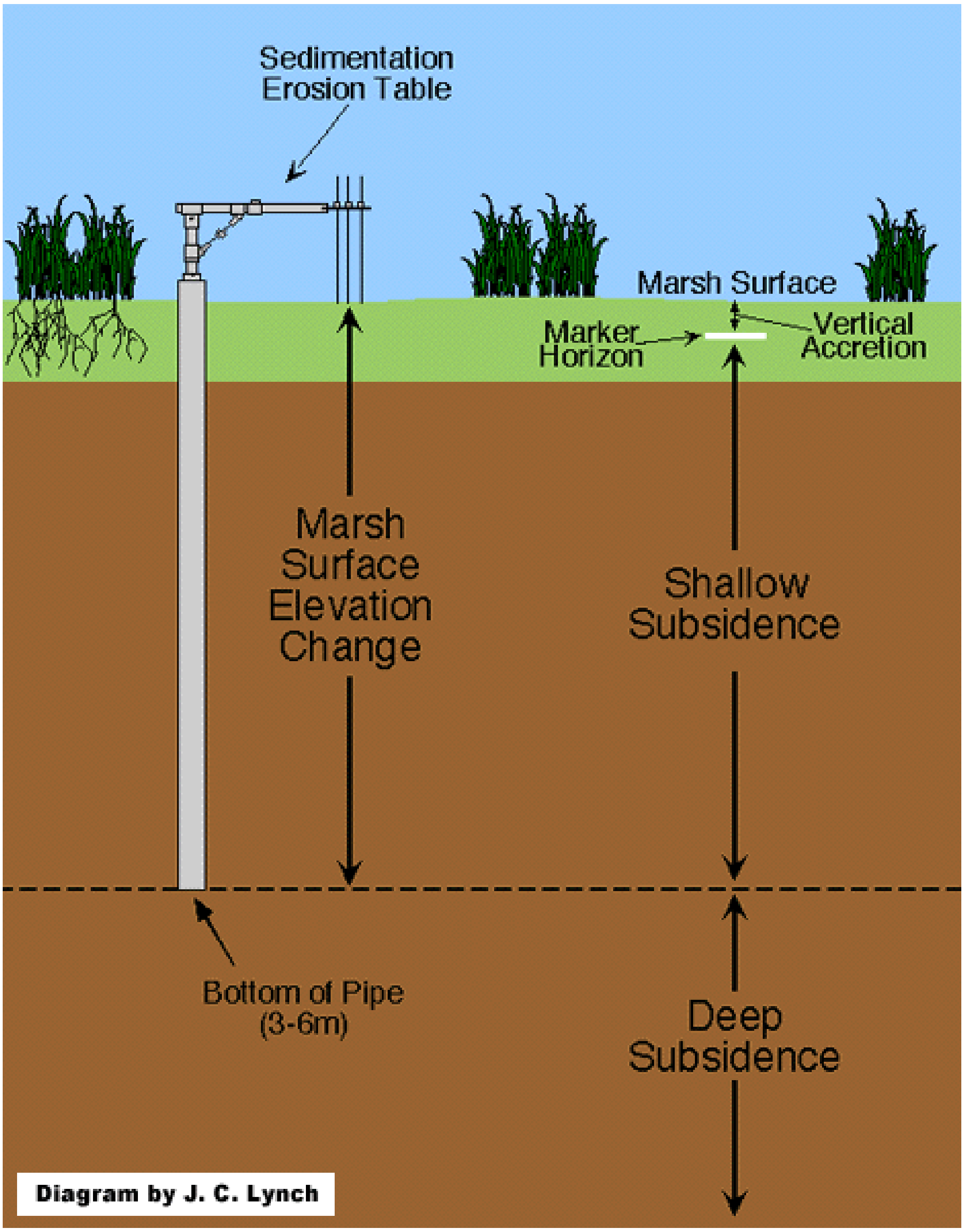

Figure 3: Overview of the Sedimentation-Erosion Table.

Before installation, the site was prepared so that disturbance of the marsh surface was minimised. A platform of treated pine was built from which installation was performed and all subsequent measurements taken. A six-metre aluminium pole was driven into the marsh surface to the point of refusal, to act as a permanent benchmark from which measurements of surface elevation are taken (Figure 4). The pole extends approximately 25cm from the wetland surface and an insert pipe is cemented at the top to provide a junction for the vertical arm of the SET. The SET fits onto the insert pipe at fixed positions so that at least four replicate sets of readings can be taken (Figure 5).

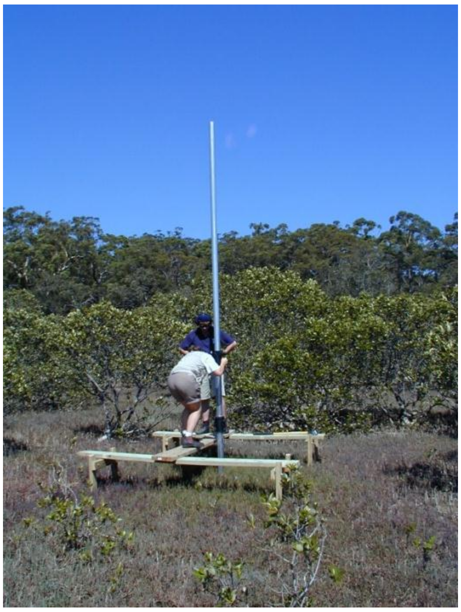

Figure 4: SET installation.

Figure 5: SET readings.

The SET was positioned on the insert pipe and the horizontal arm was levelled. The compass direction of the horizontal arm of the SET was recorded. Nine pins were lowered to the marsh surface and the length of each pin above the horizontal table was measured to the nearest millimetre using a ruler. This procedure was repeated for the nine pins and in four directions to yield a maximum of 36 elevation measurements per SET.

SET monitoring stations were employed in replicated sets of three to characterise the sedimentation and elevation properties of both the mangrove and saltmarsh vegetation of the Tweed River. A total of 6 SET's were installed on Ukerebagh Island, with three being established within the mangrove vegetation and three in the saltmarsh vegetation.

2.4 Sedimentation

In conjunction with the SET installation, feldspar marker horizons were sprinkled on the wetlands surface at the perimeter of each SET. These horizons serve as a sedimentation marker against which vertical accretion in measured. Subsidence or uplift was then determined based on the difference between the degree of vertical accretion measured from the feldspar markers and the extent of surface elevation change measured with the SET.

Three 0.5 m x 0.5 m (0.25 m^2) feldspar marker horizons were sprinkled on the wetland surface at the perimeter of each SET at the time of installation (Figure 6). Mini-cores were removed from each feldspar marker horizon in December 2001, November 2002, January 2006, October 2007, and December 2010, and the degree of sedimentation was measured using a ruler (Figure 7). Feldspar could not be retrieved from the saltmarsh in 2012 due to erosion, or from the mangrove due to bioturbation.

Figure 6: Feldspar marker horizon application.

Figure 7: Mini-core removal and measurement.

3 Results

3.1 Photogrammetric Mapping

3.1.1 Changes in Extent

Changes in extent within the entire mapped area at 13 May 2000, 19 April 2002, 8 June 2003, 5 March 2005, 12 June 2007, 5 April 2010 and 3 April 2012 are shown in Figure 8 to Figure 14. Changes in extent within the four geomorphic settings are shown in Figure 15 to Figure 42 (See Appendix 1 for larger versions).

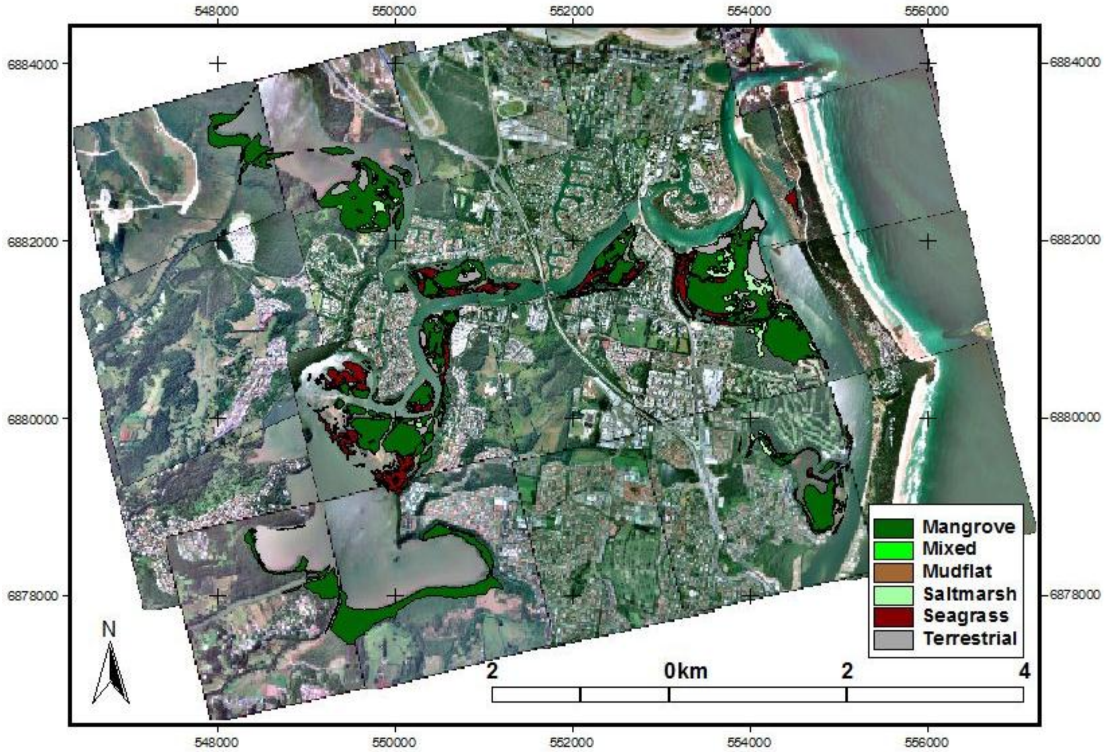

Figure 8: Tweed River estuarine vegetation extent on 13 May 2000.

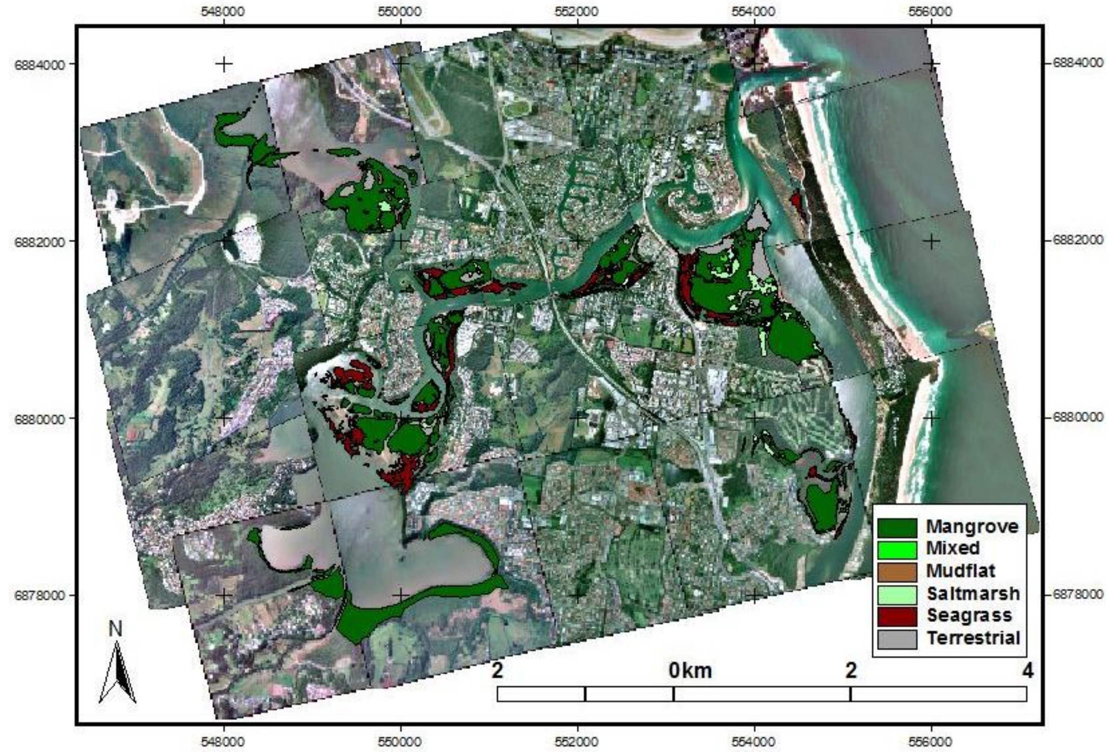

Figure 9: Tweed River estuarine vegetation extent on 19 April 2002.

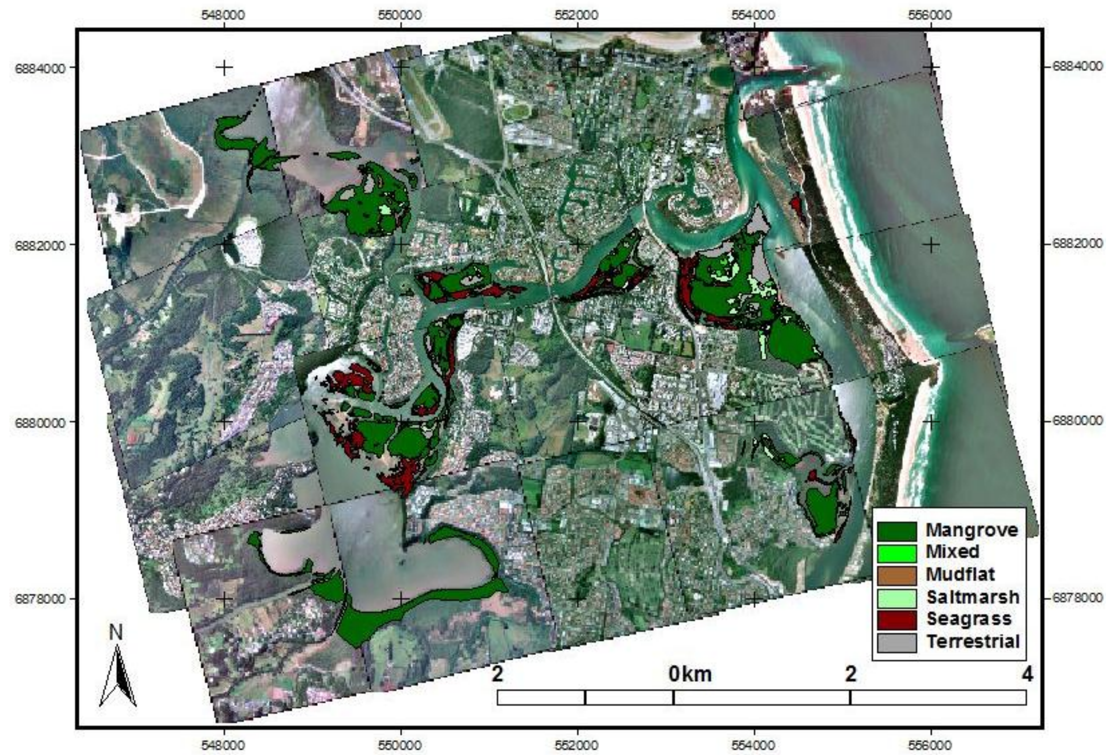

Figure 10: Tweed River estuarine vegetation extent on 8 June 2003.

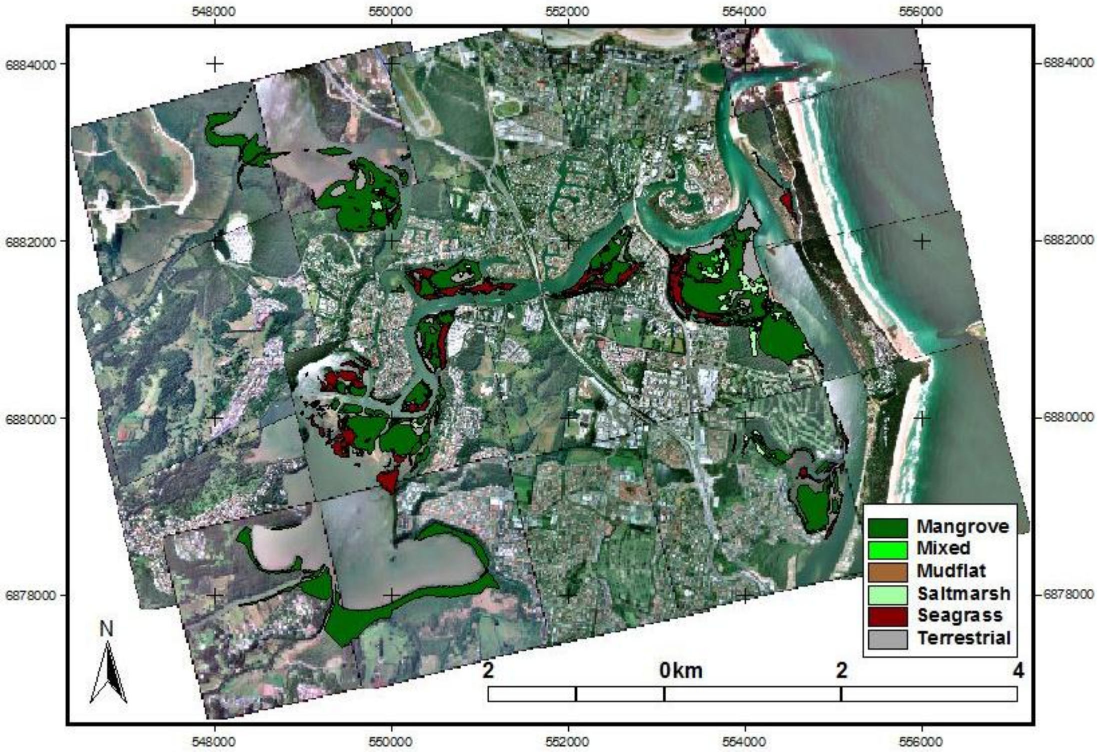

Figure 11: Tweed River estuarine vegetation extent on 5 March 2005.

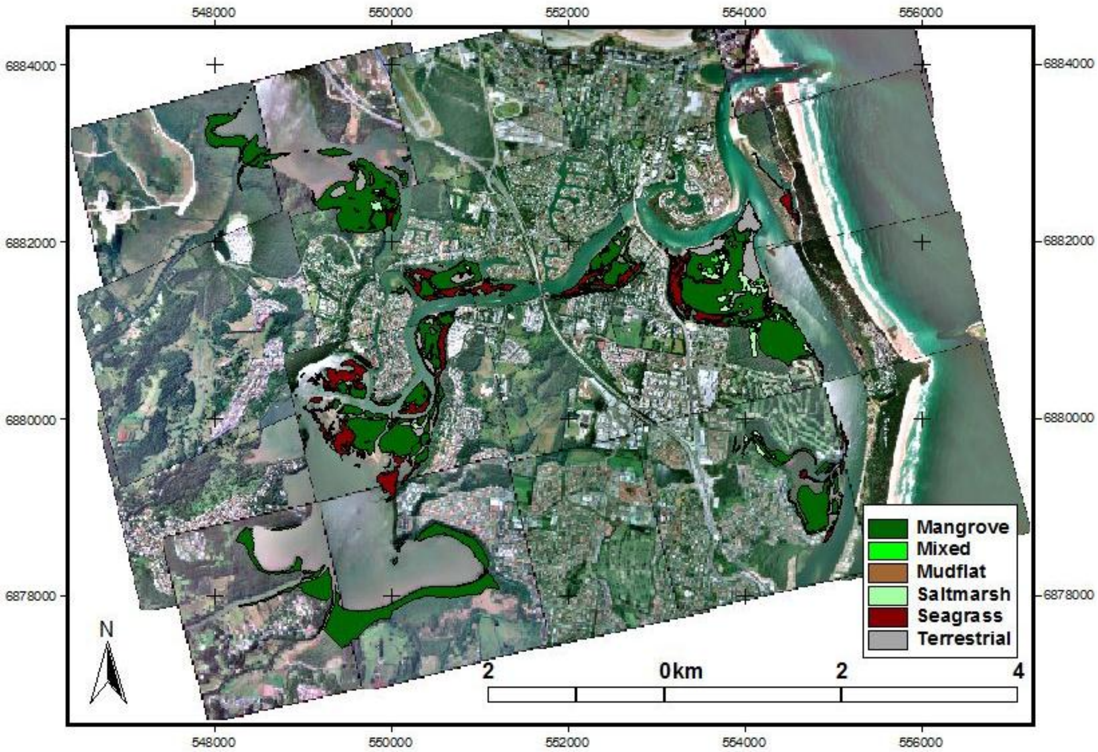

Figure 12: Tweed River estuarine vegetation on 12 June 2007.

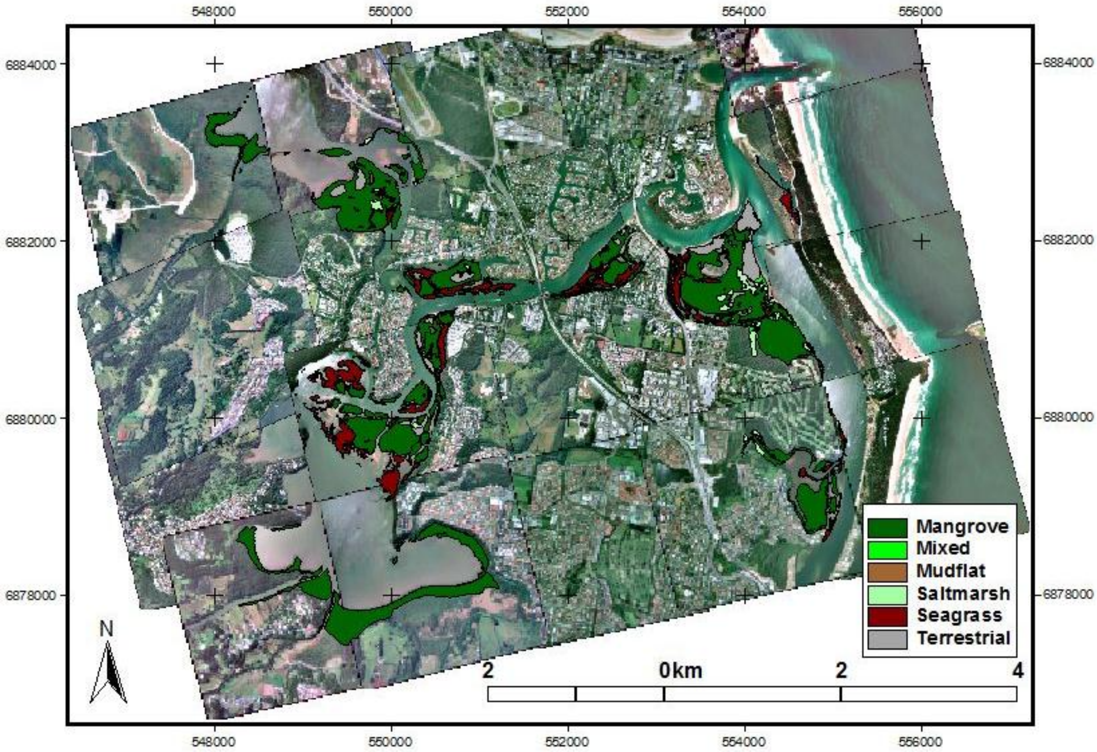

Figure 13: Tweed River estuarine vegetation on 5 April 2010.

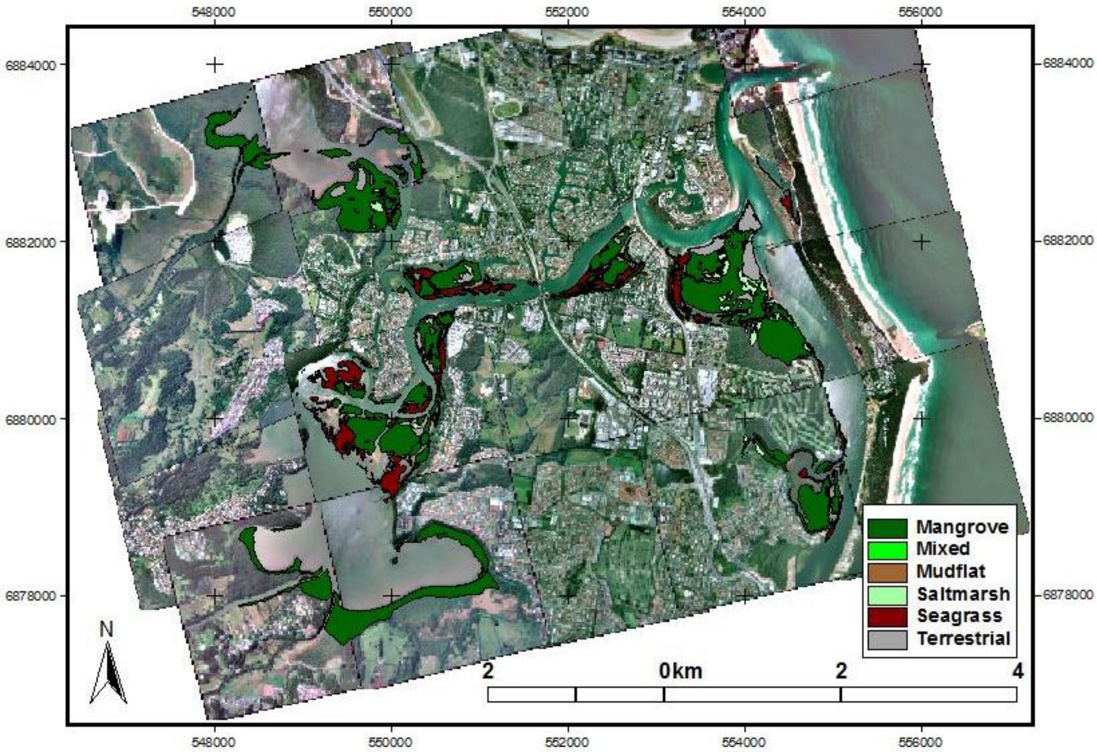

Figure 14: Tweed River estuarine vegetation on 3 April 2012.

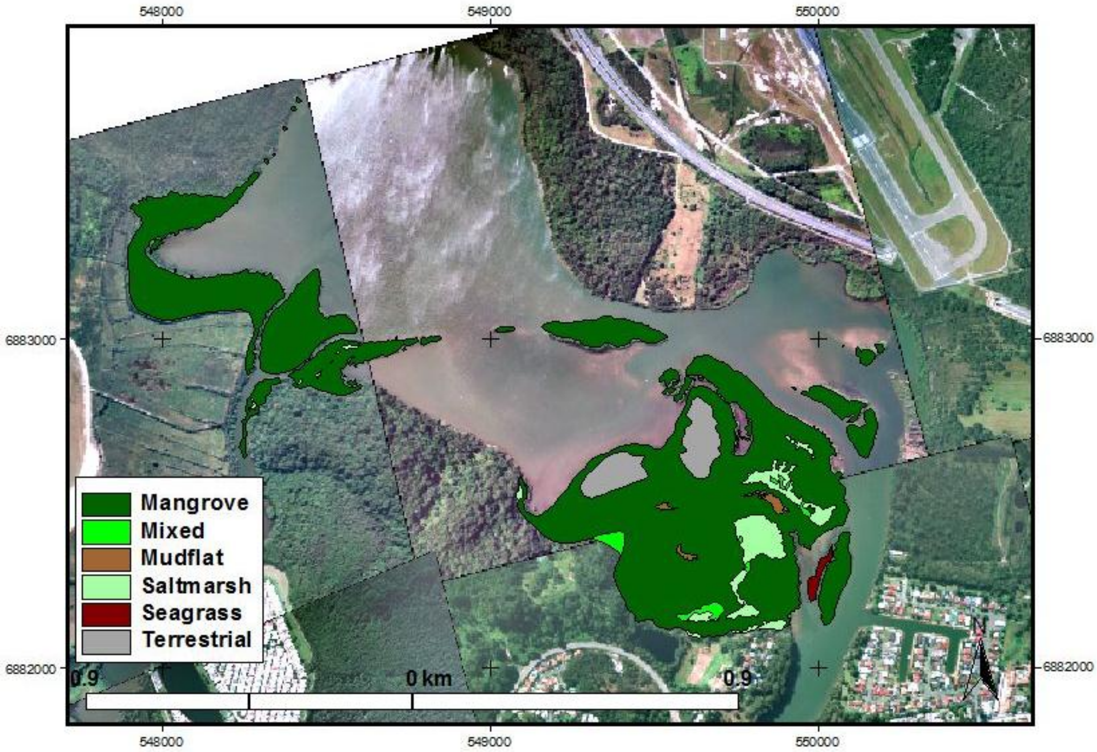

Figure 15: Cobaki Broadwater estuarine vegetation extent on 13 May 2000.

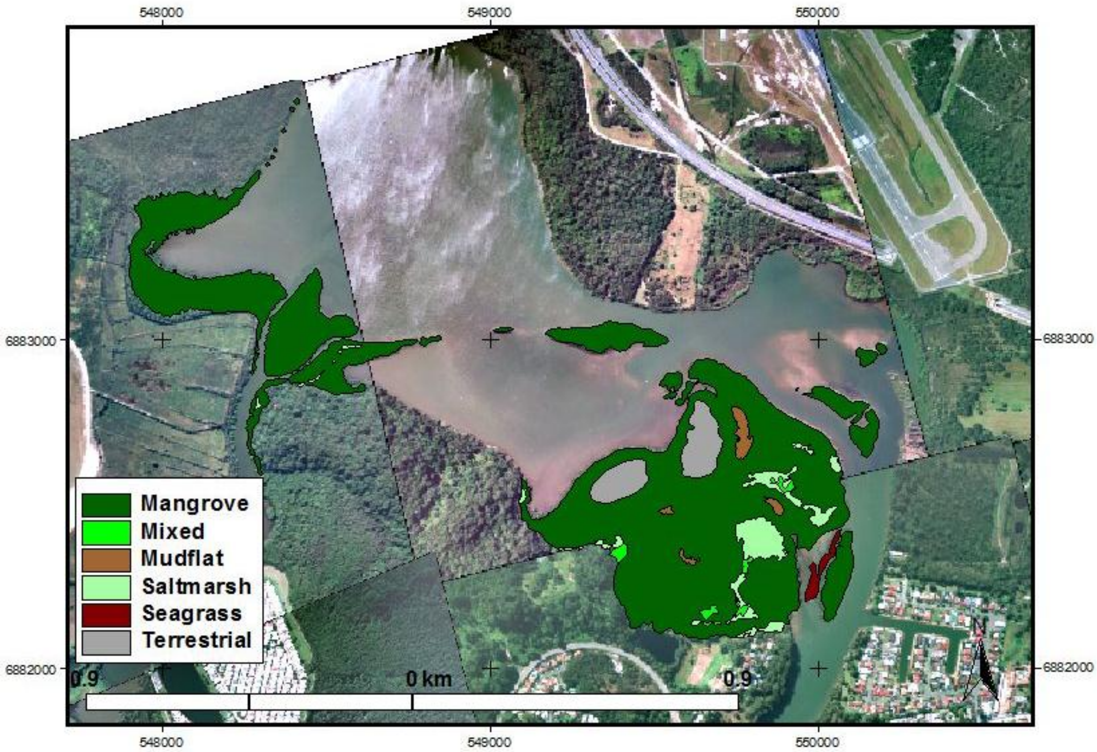

Figure 16: Cobaki Broadwater estuarine vegetation extent on 19 April 2002.

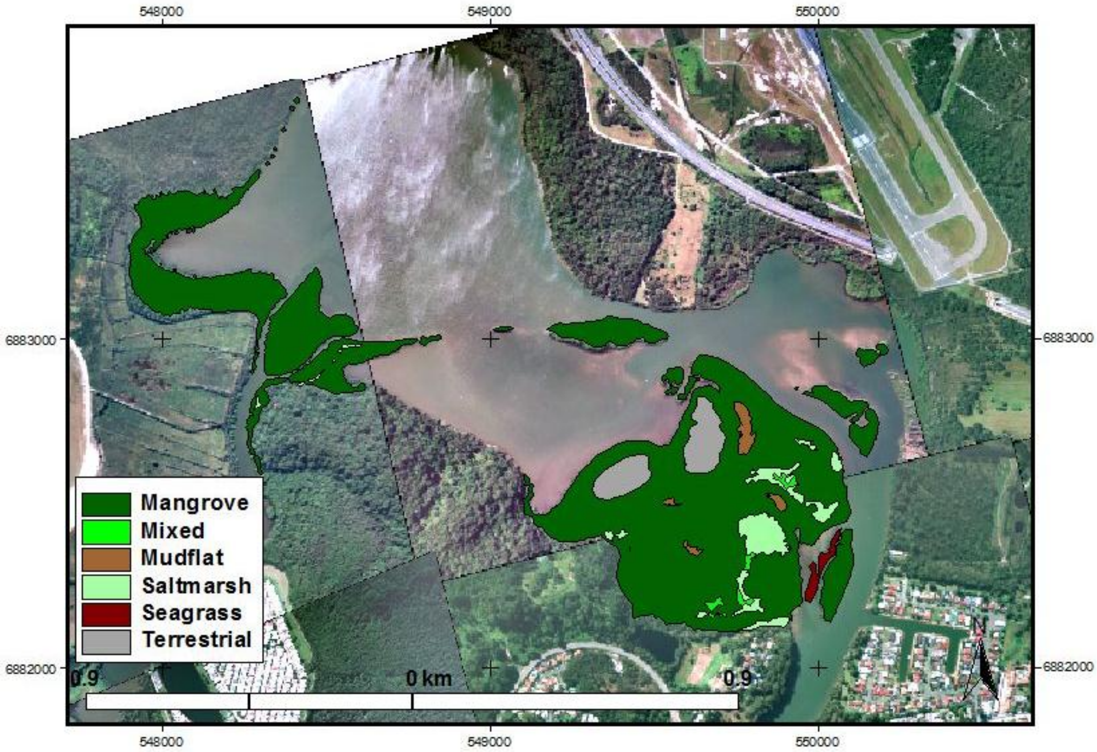

Figure 17: Cobaki Broadwater estuarine vegetation extent on 8 June 2003.

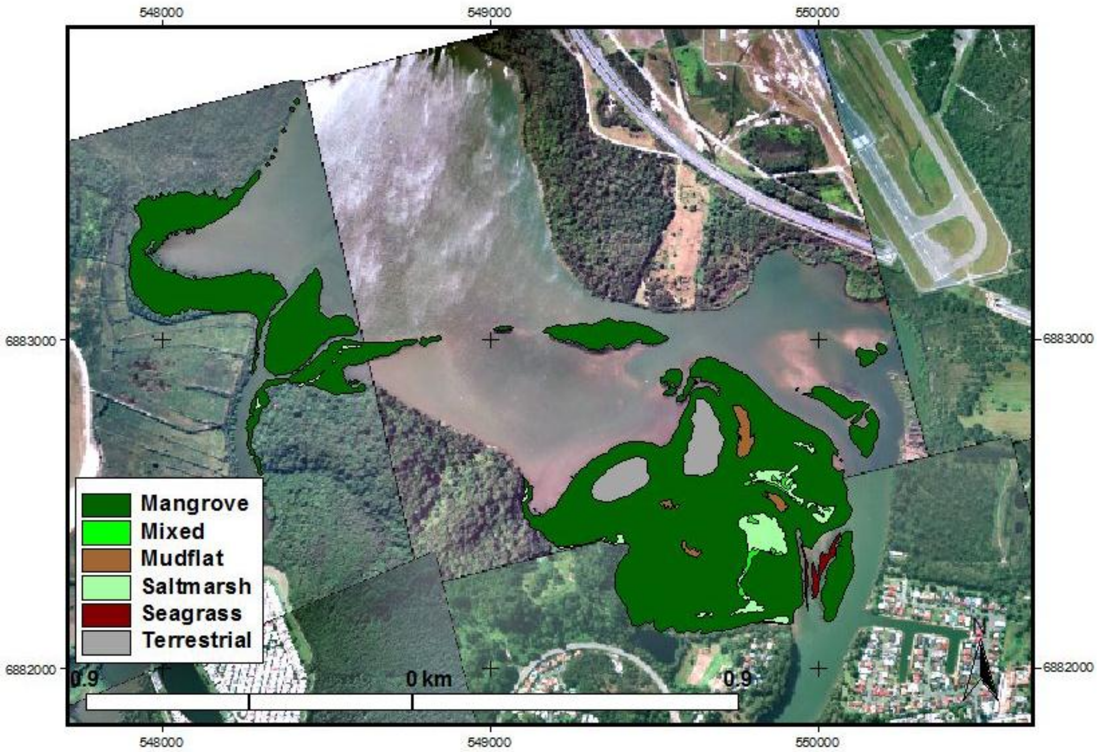

Figure 18: Cobaki Broadwater estuarine vegetation extent on 5 March 2005.

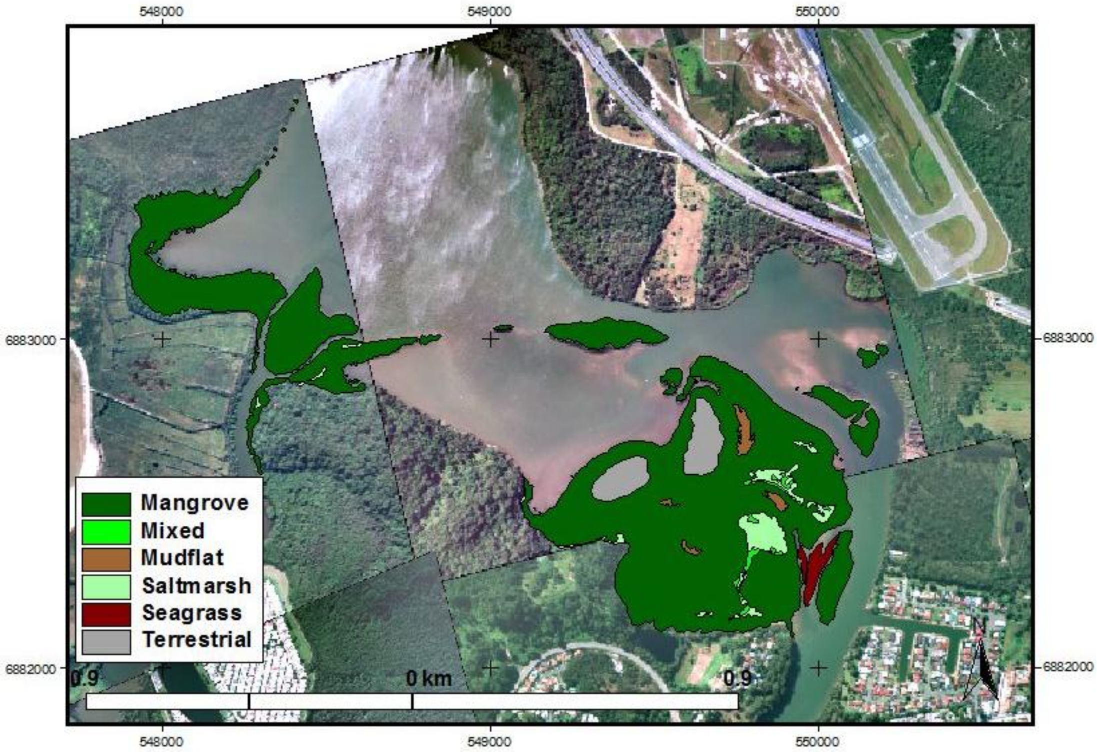

Figure 19: Cobaki Broadwater estuarine vegetation extent on 12 June 2007

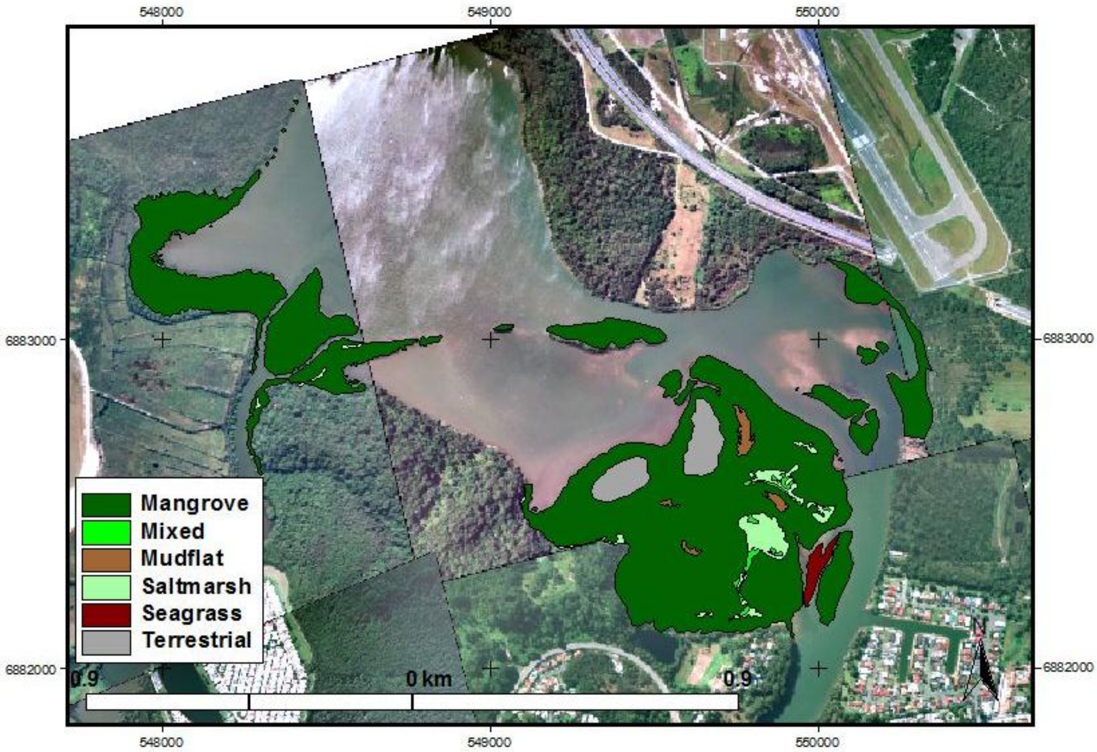

Figure 20: Cobaki Broadwater estuarine vegetation extent on 5 April 2010.

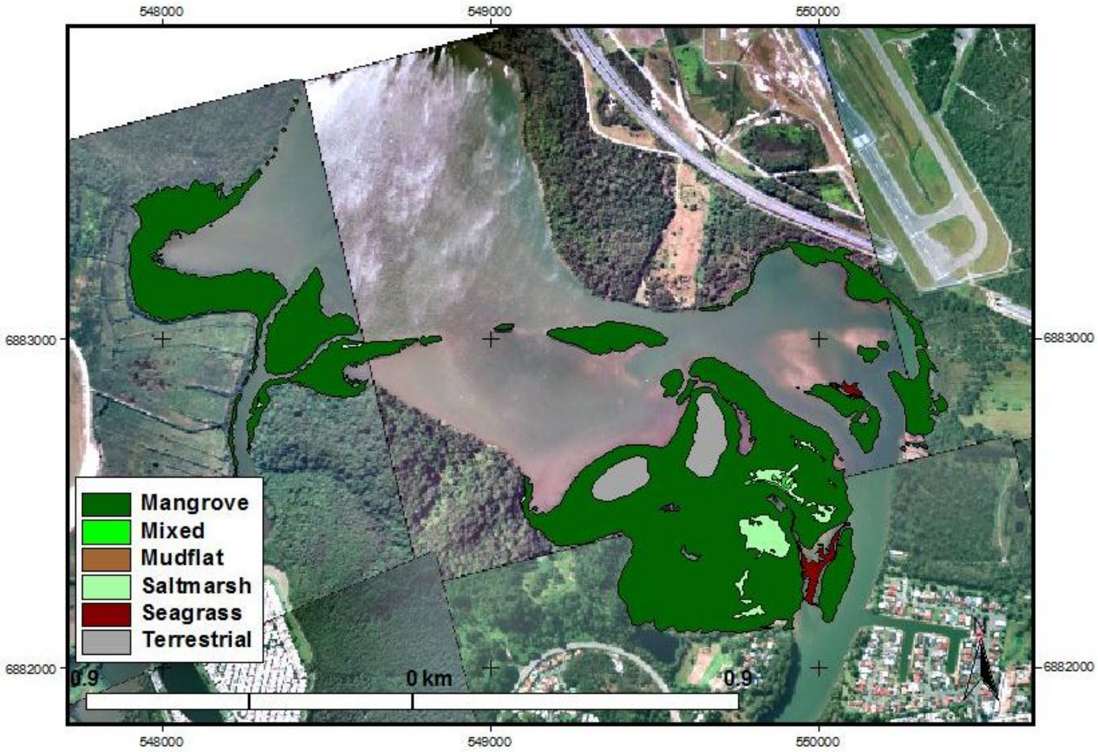

Figure 21: Cobaki Broadwater estuarine vegetation extent on 3 April 2012.

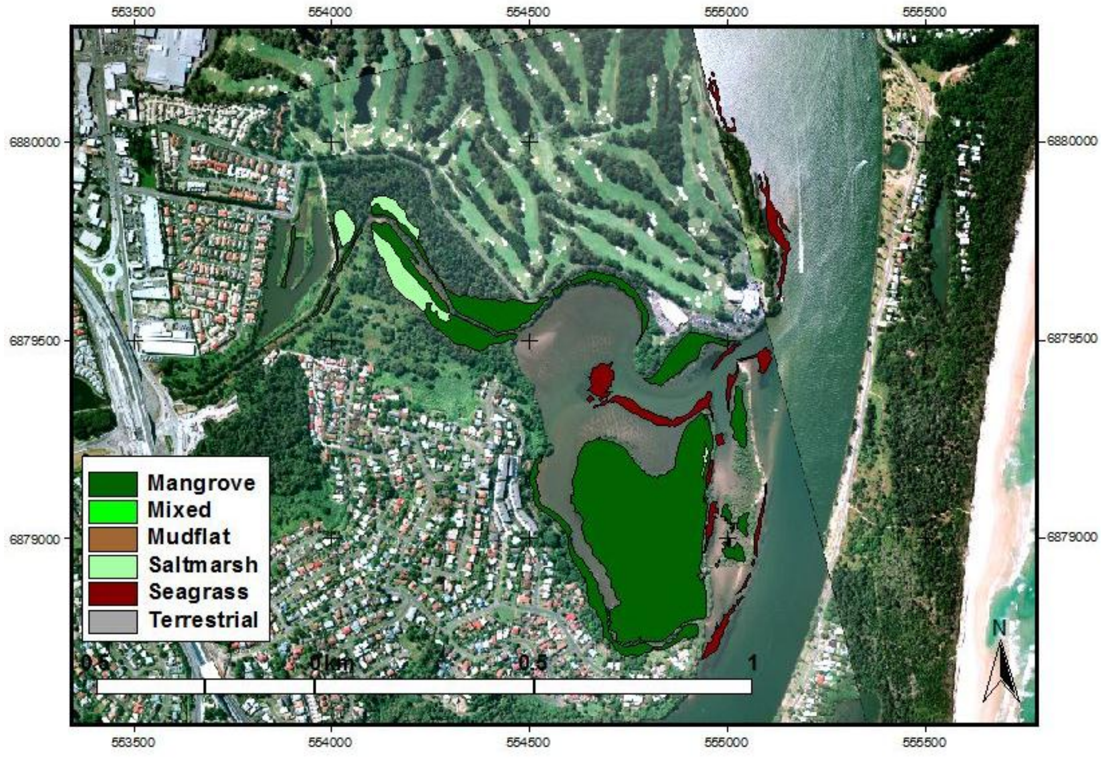

Figure 22: Fluvial Channel estuarine vegetation extent on 13 May 2000.

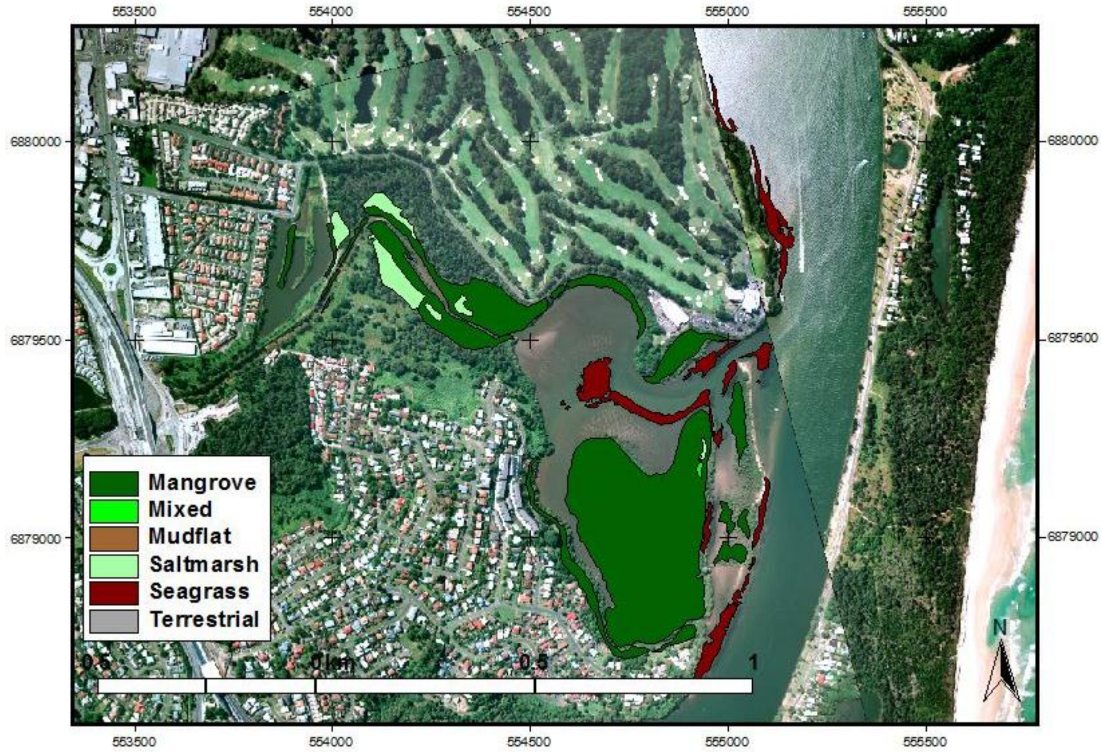

Figure 23: Fluvial Channel estuarine vegetation extent on 19 April 2002.

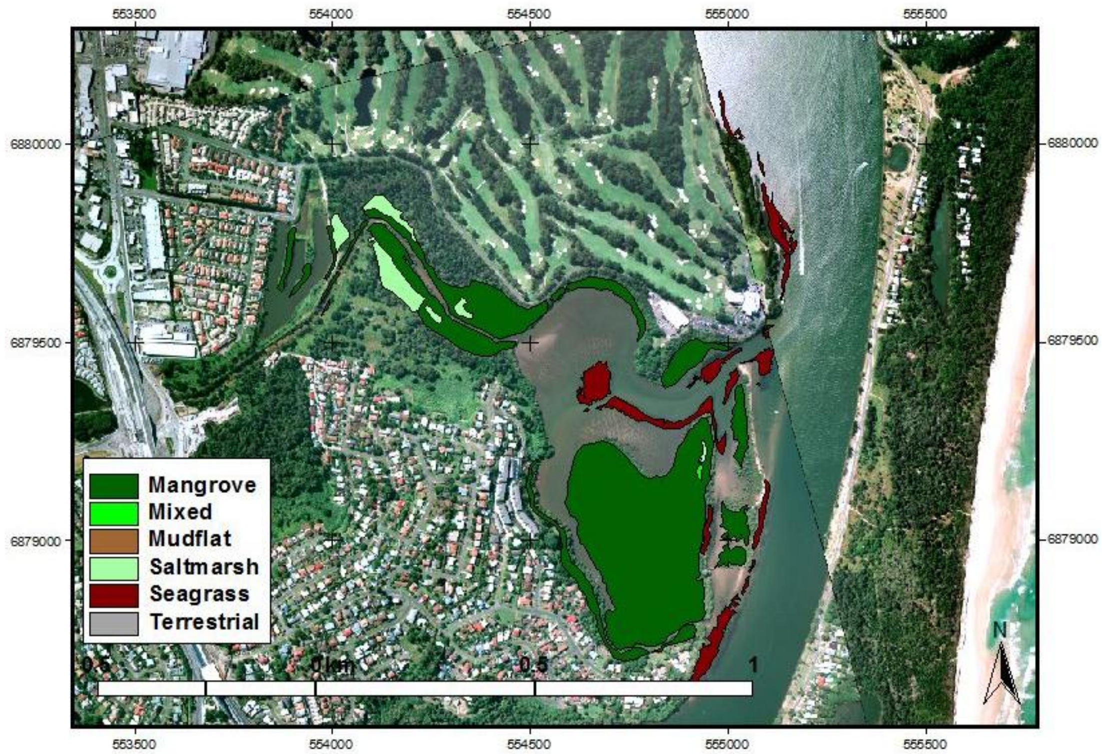

Figure 24: Fluvial Channel estuarine vegetation extent on 8 June 2003.

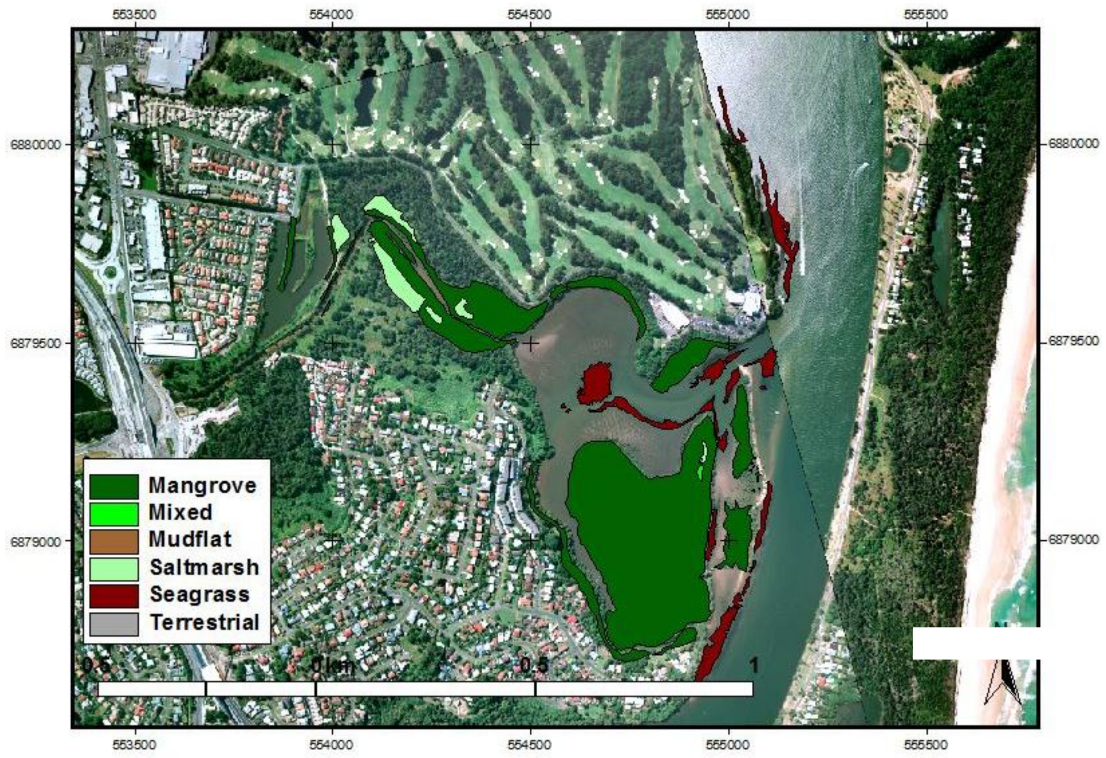

Figure 25: Fluvial Channel estuarine vegetation extent on 5 March 2005.

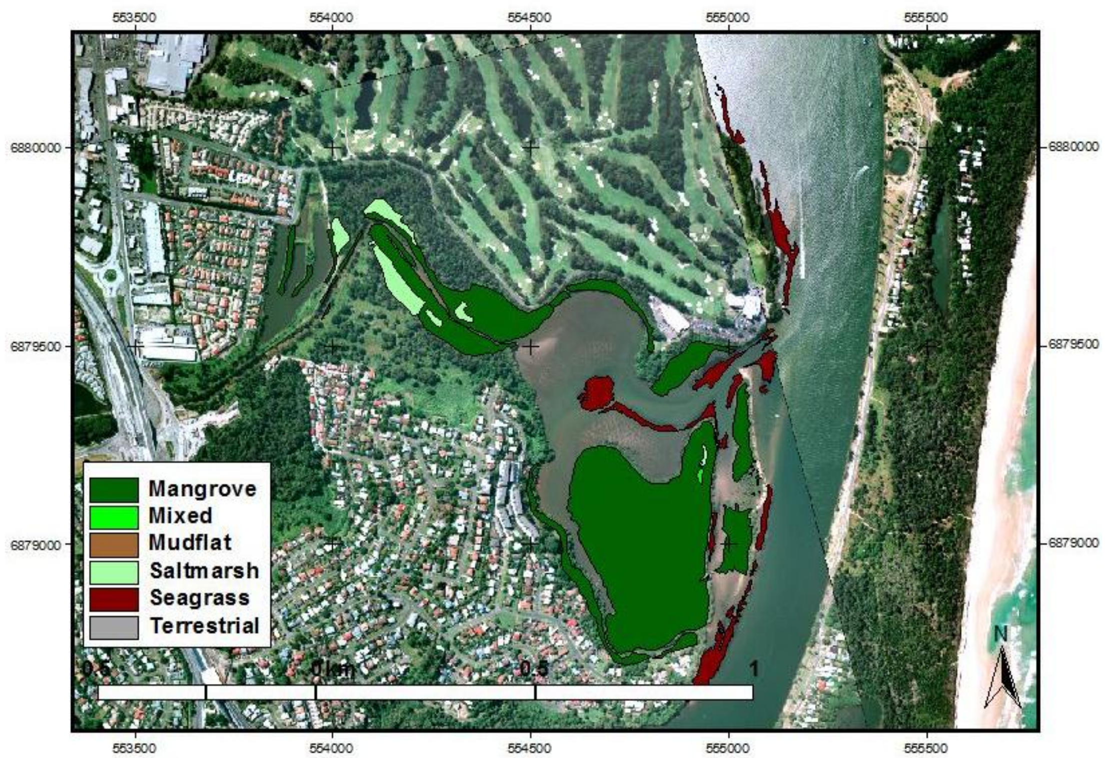

Figure 26: Fluvial Channel estuarine vegetation extent on 12 June 2007.

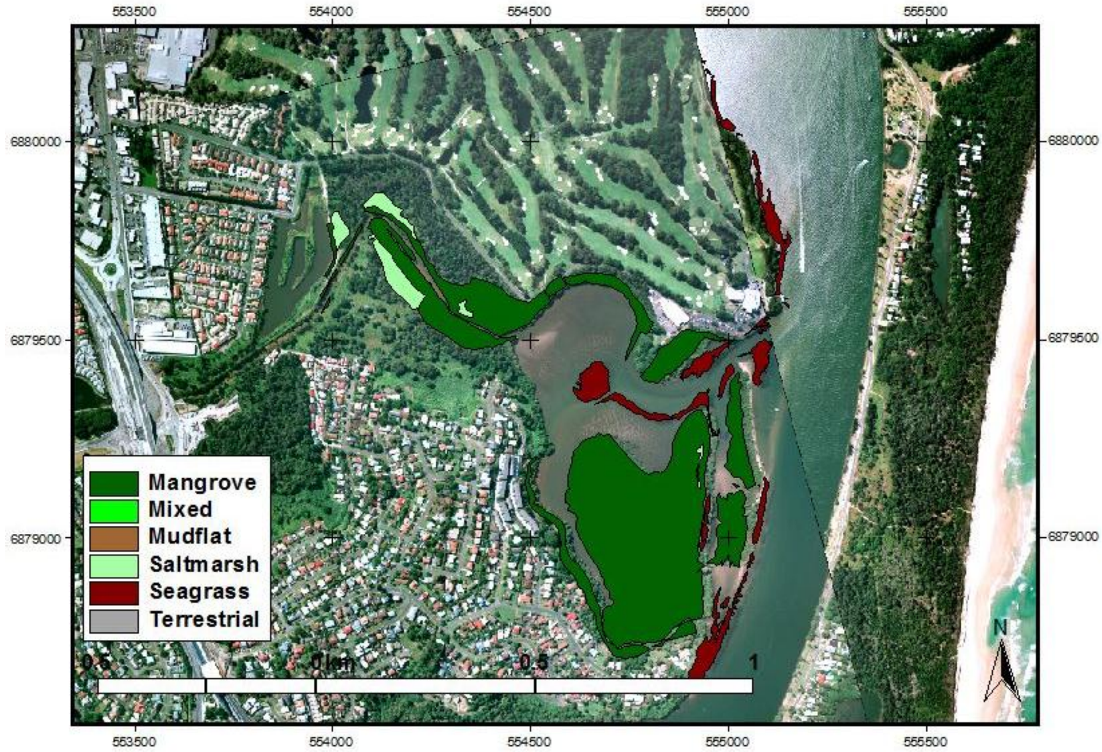

Figure 27: Fluvial Channel estuarine vegetation extent on 5 April 2010.

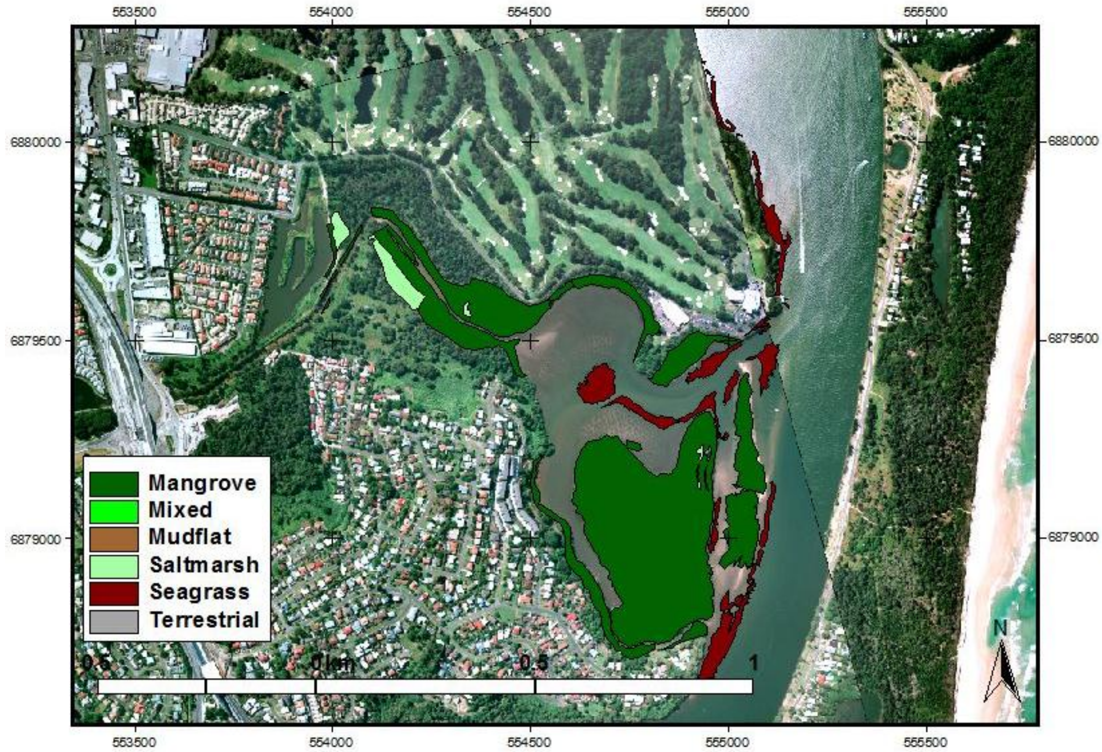

Figure 28: Fluvial Channel estuarine vegetation extent on 3 April 2012.

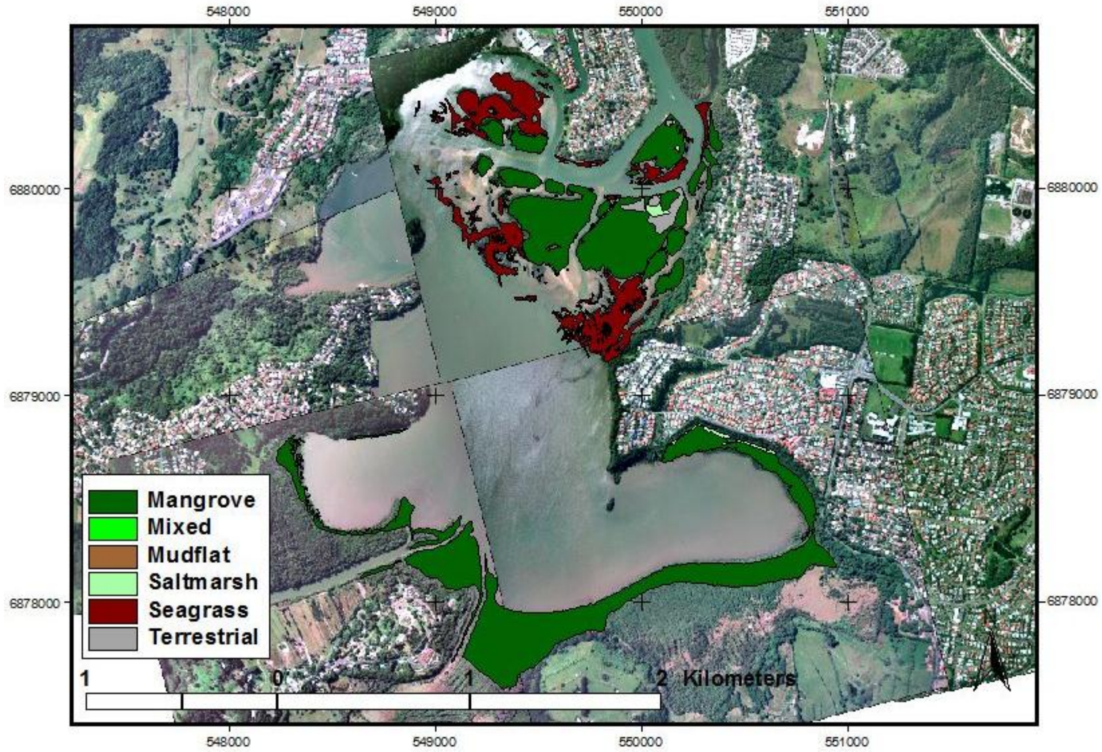

Figure 29: Terranora Broadwater estuarine vegetation extent on 13 May 2000.

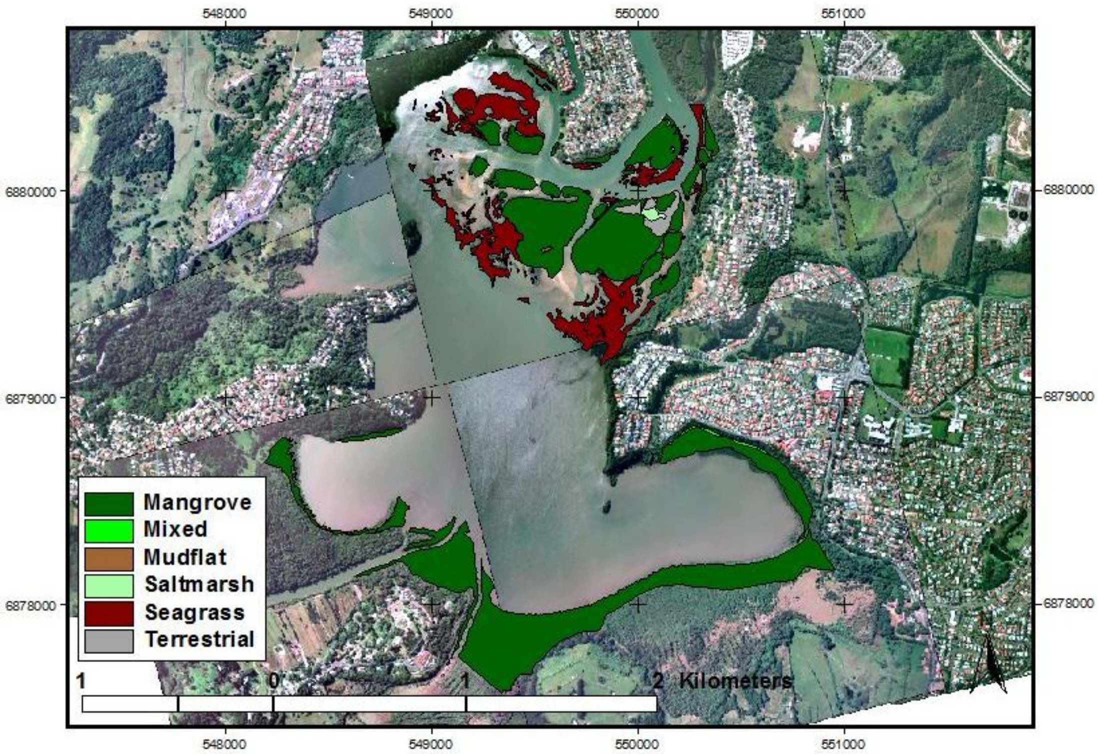

Figure 30: Terranora Broadwater estuarine vegetation extent on 19 April 2002.

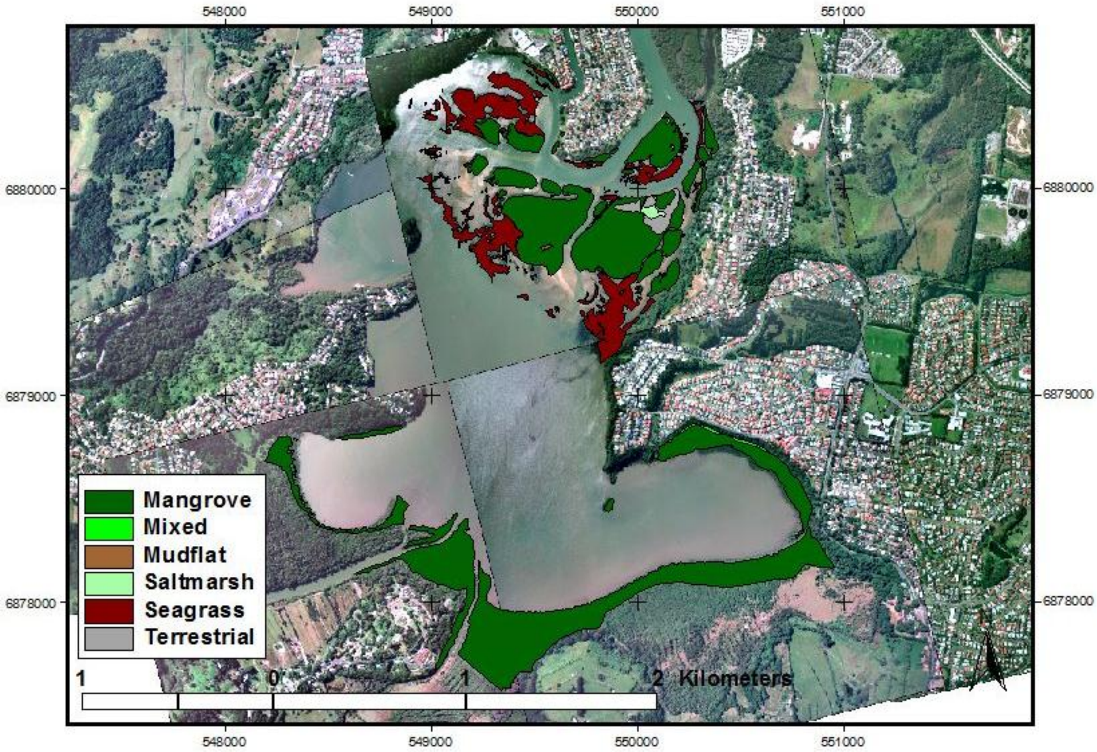

Figure 31: Terranora Broadwater estuarine vegetation extent on 8 June 2003.

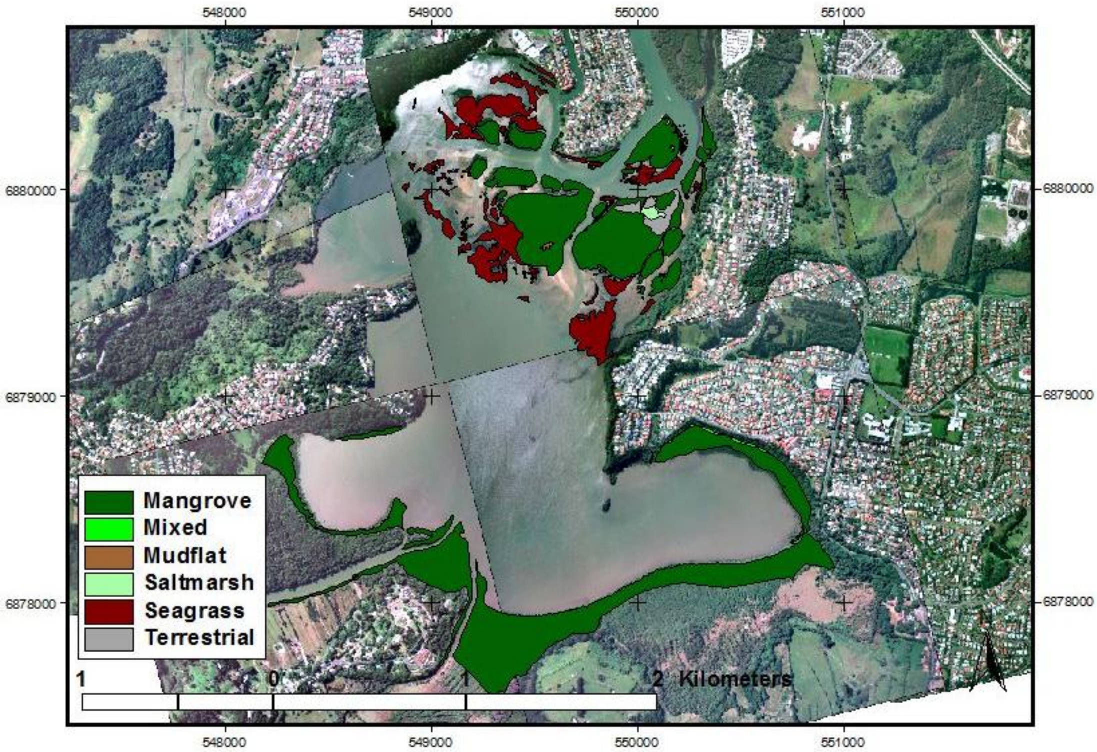

Figure 32: Terranora Broadwater estuarine vegetation extent on 5 March 2005.

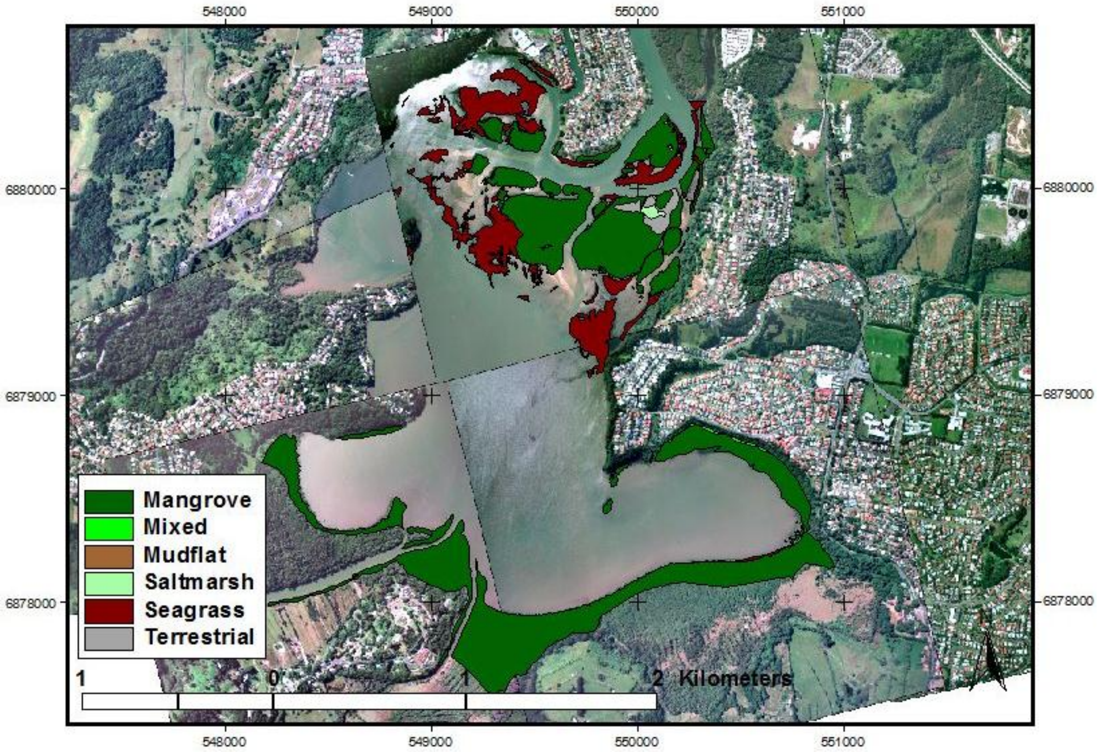

Figure 33: Terranora Broadwater estuarine vegetation extent on 12 June 2007.

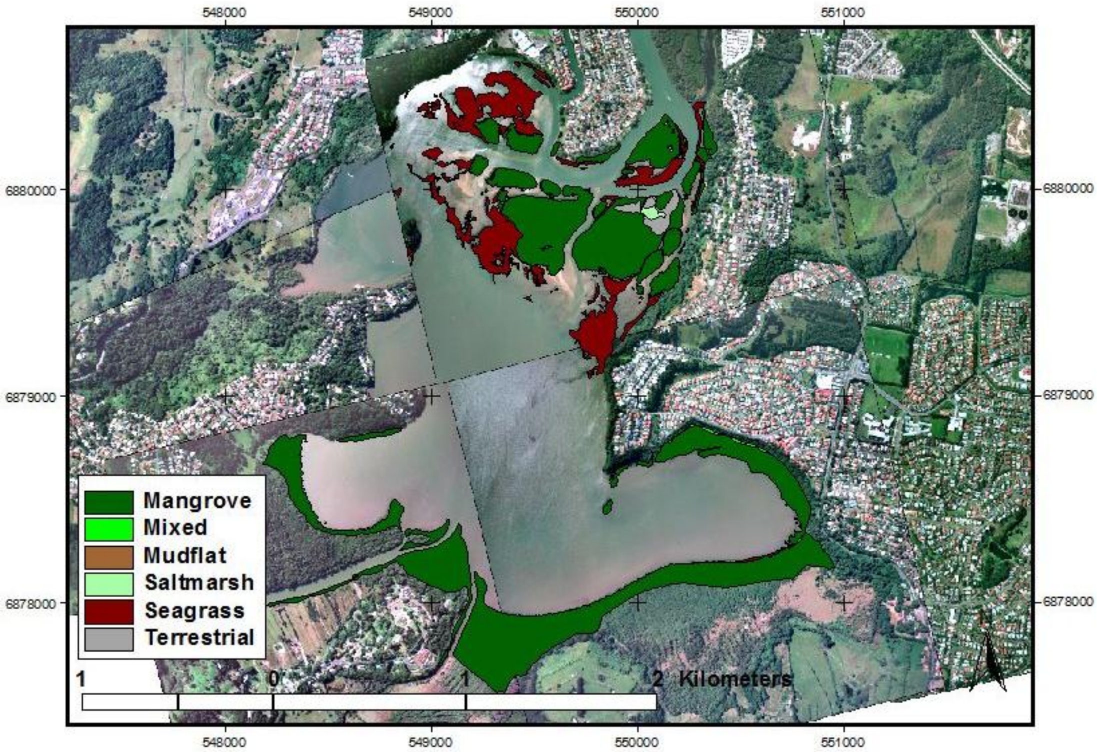

Figure 34: Terranora Broadwater estuarine vegetation extent on 5 April 2010.

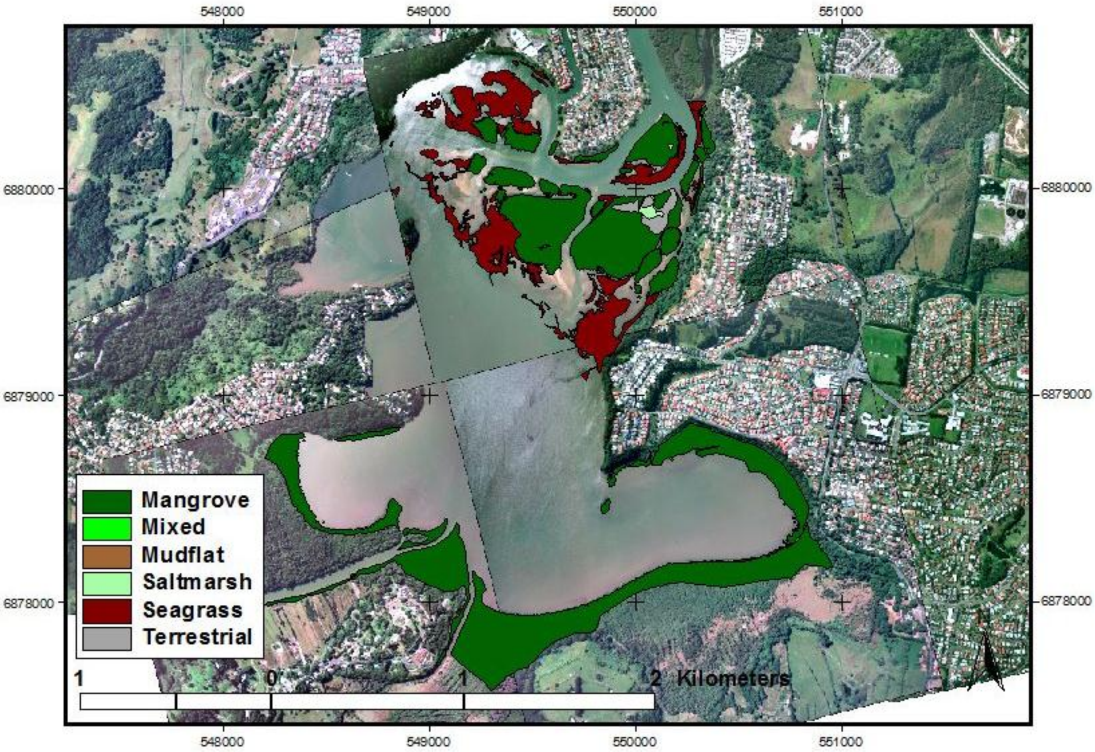

Figure 35: Terranora Broadwater estuarine vegetation extent on 3 April 2012.

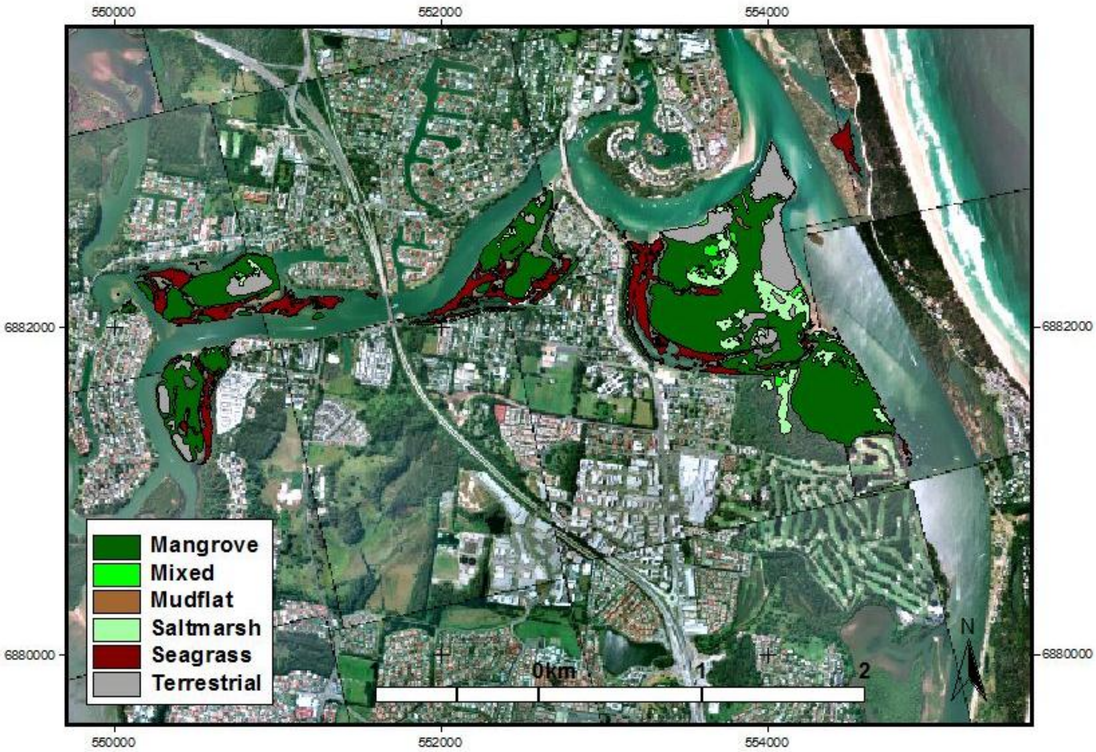

Figure 36: Tidal Channel estuarine vegetation extent on 13 May 2000.

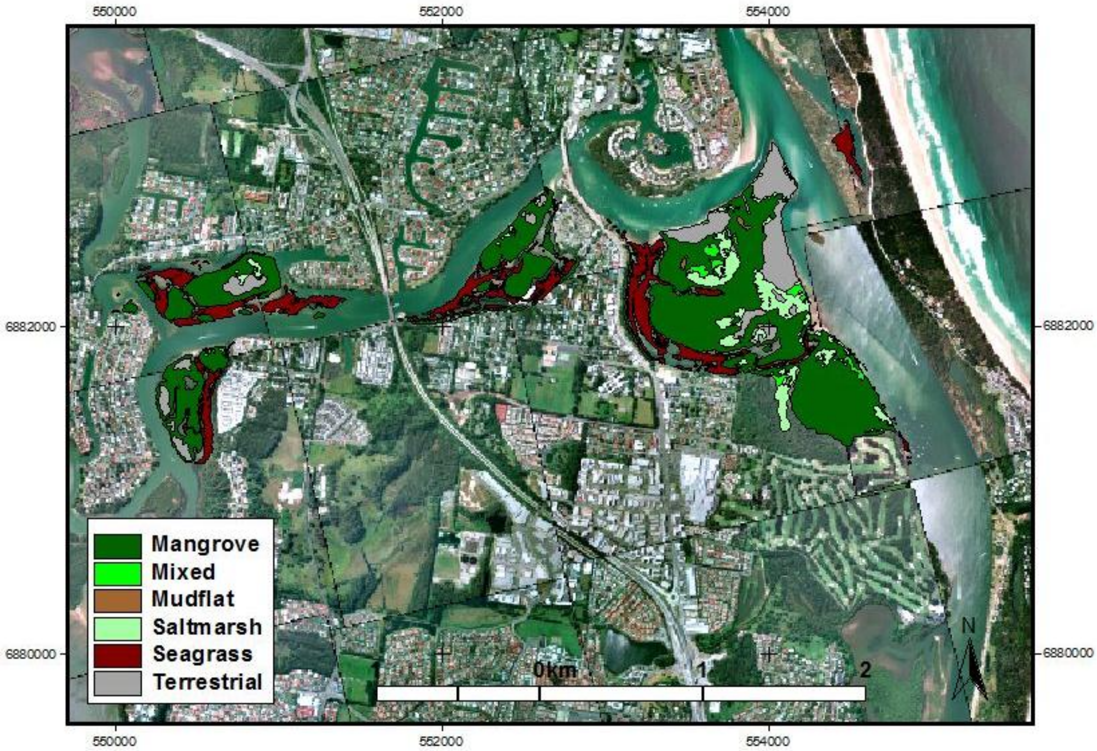

Figure 37: Tidal Channel estuarine vegetation extent on 19 April 2002.

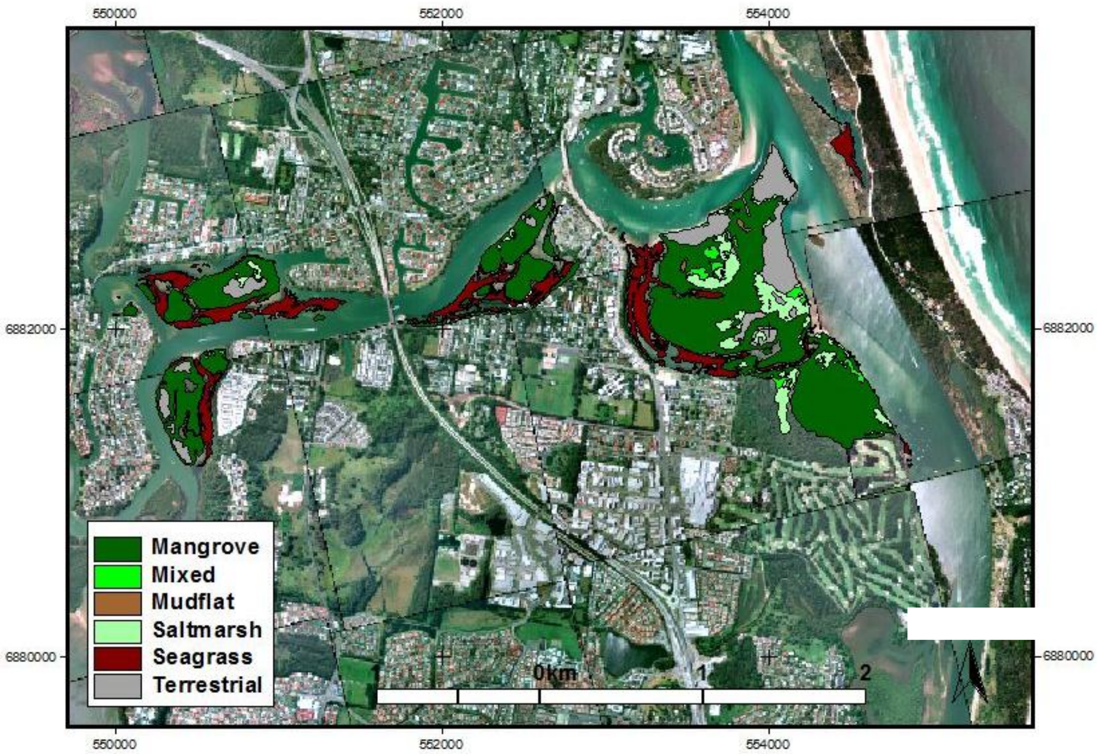

Figure 38: Tidal Channel estuarine vegetation extent on 8 June 2003.

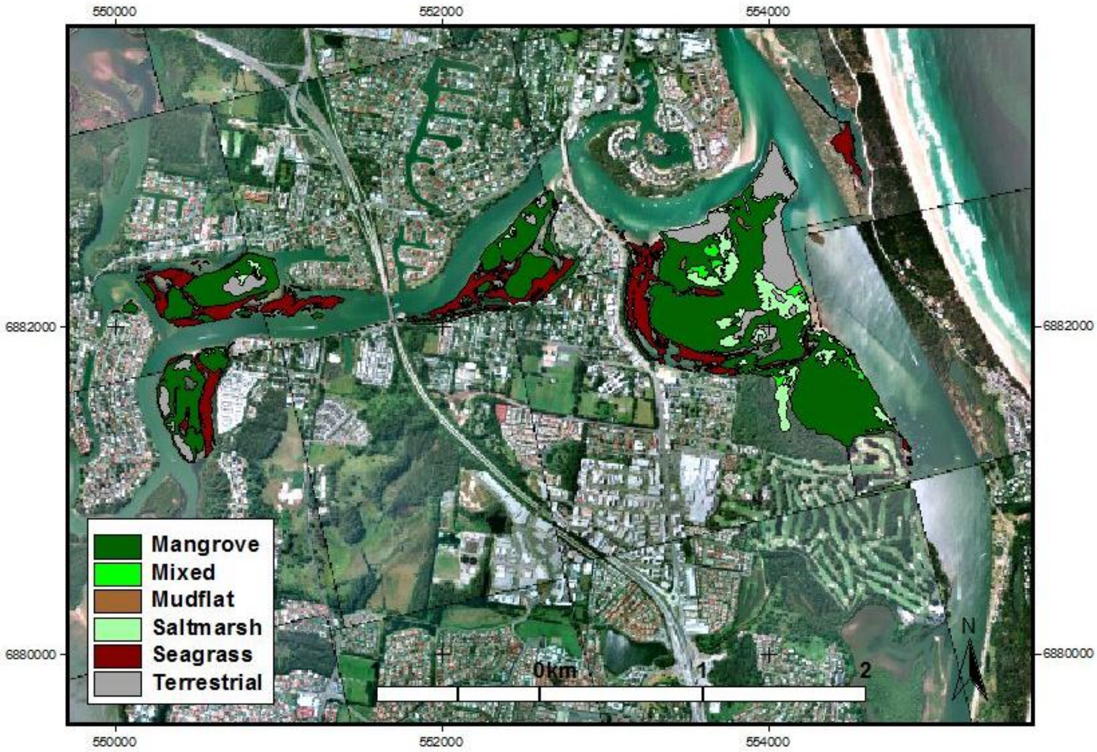

Figure 39: Tidal Channel estuarine vegetation extent on 5 March 2005.

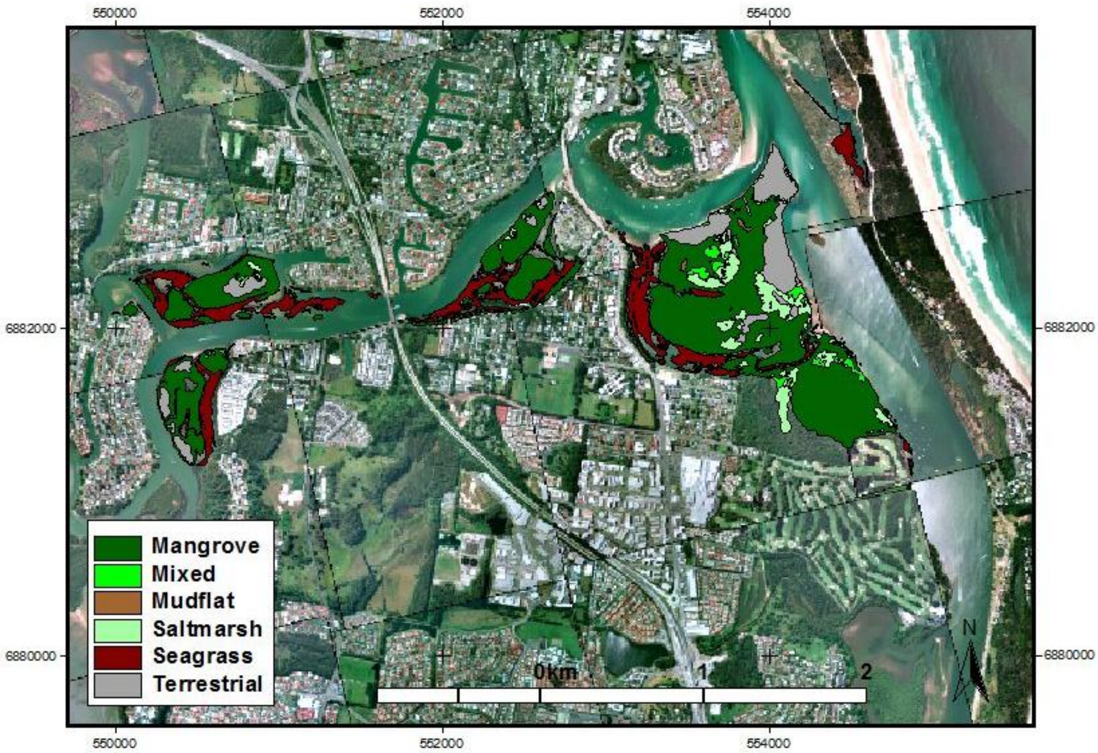

Figure 40: Tidal Channel estuarine vegetation extent on 12 June 2007.

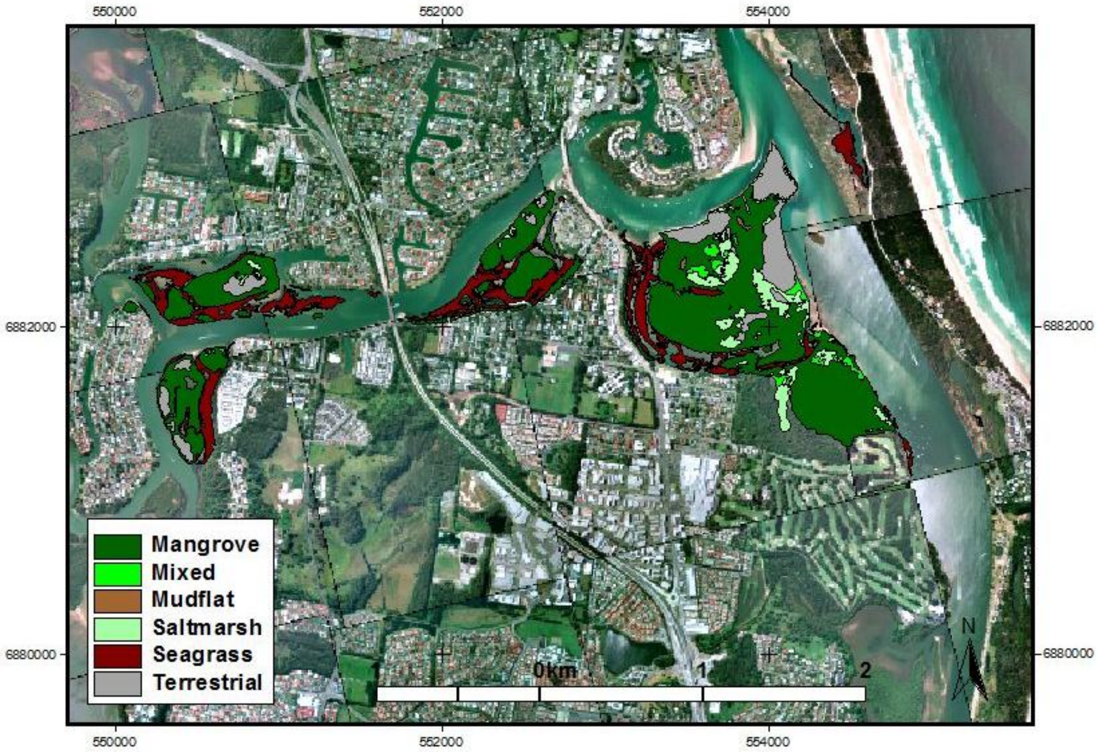

Figure 41: Tidal Channel estuarine vegetation extent on 5 April 2010.

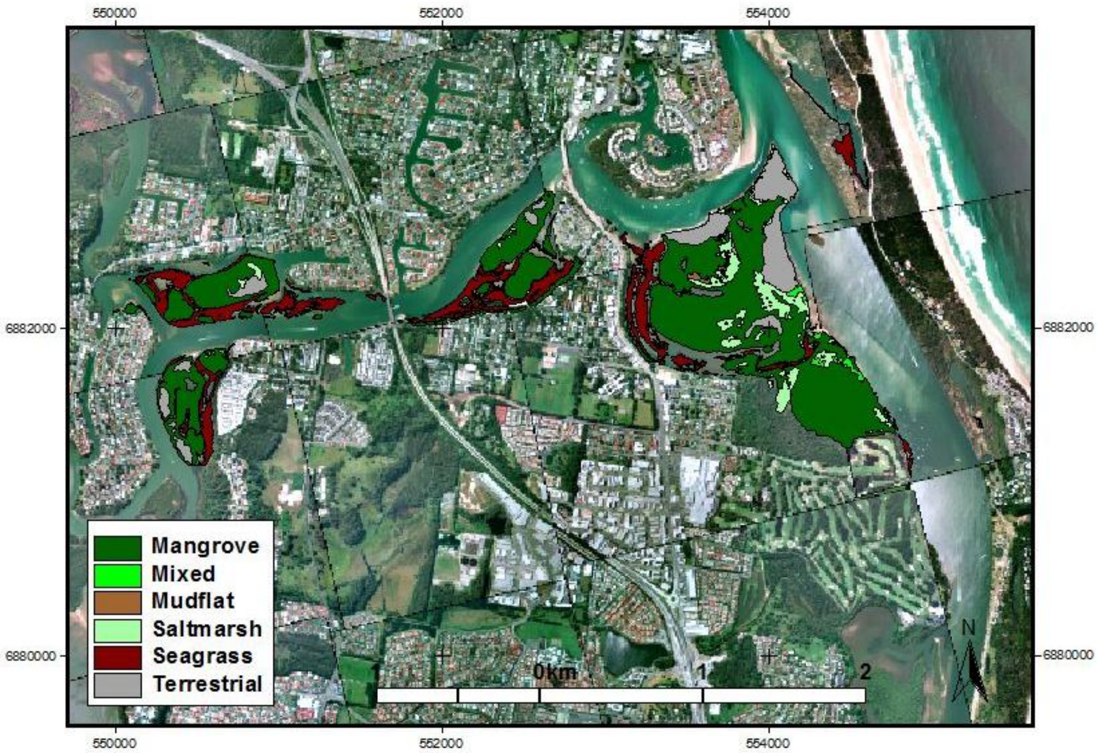

Figure 42: Tidal Channel estuarine vegetation extent on 3 April 2012.

3.1.2 Changes in Area

Mangrove area increased by approximately 11.68% and seagrass area increased by approximately 20.08% between 13 May 2000 and 3 April 2012 within the entire mapped area (Figure 43). Both of these increases were higher than the estimated digitising mapping errors shown in Table 8. The increase in seagrass area is likely to be significant, given the small error in the digitising process. The decrease in saltmarsh area of 25.93% over the same period is of concern, though consistent with historic trends.

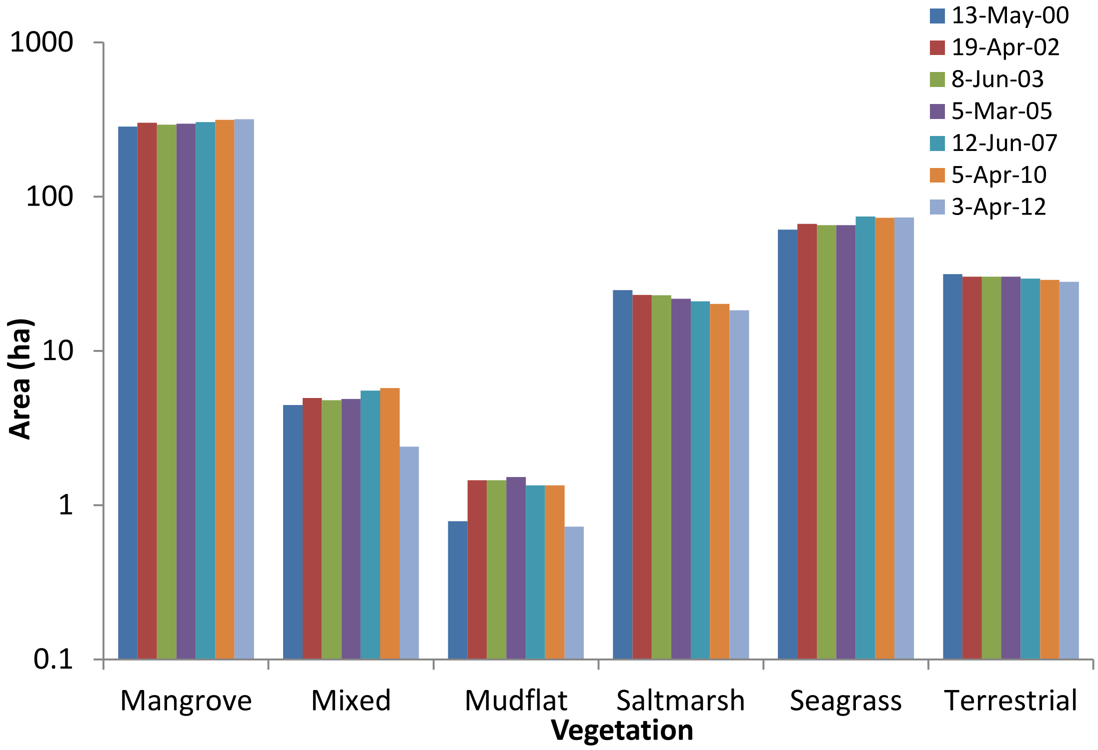

Figure 43: Change in the area of estuarine vegetation within the mapped section of the Tweed River.

The area of mangrove and seagrass in each geomorphic setting increased over the study period. Alternatively, saltmarsh area decreased in all geomorphic settings between 13 May 2000 and 3 April 2012 (Figure 44 to Figure 47; Table 2 to Table 6).

Table 2: Percentage change in area of estuarine vegetation in Cobaki Broadwater, Fluvial Channel, Terranora Broadwater and the Terranora Tidal Channel between 13 May 2000 and 3 April 2012.

Vegetation Type	Cobaki Broadwater	Fluvial Channel	Terranora Broadwater	Terranora Tidal Channel	All
Mangrove	13.43	25.06	13.00	4.03	11.68
Saltmarsh	-32.40	-38.71	-53.23	-18.61	-25.93
Seagrass	150.20	60.55	21.65	13.95	20.08

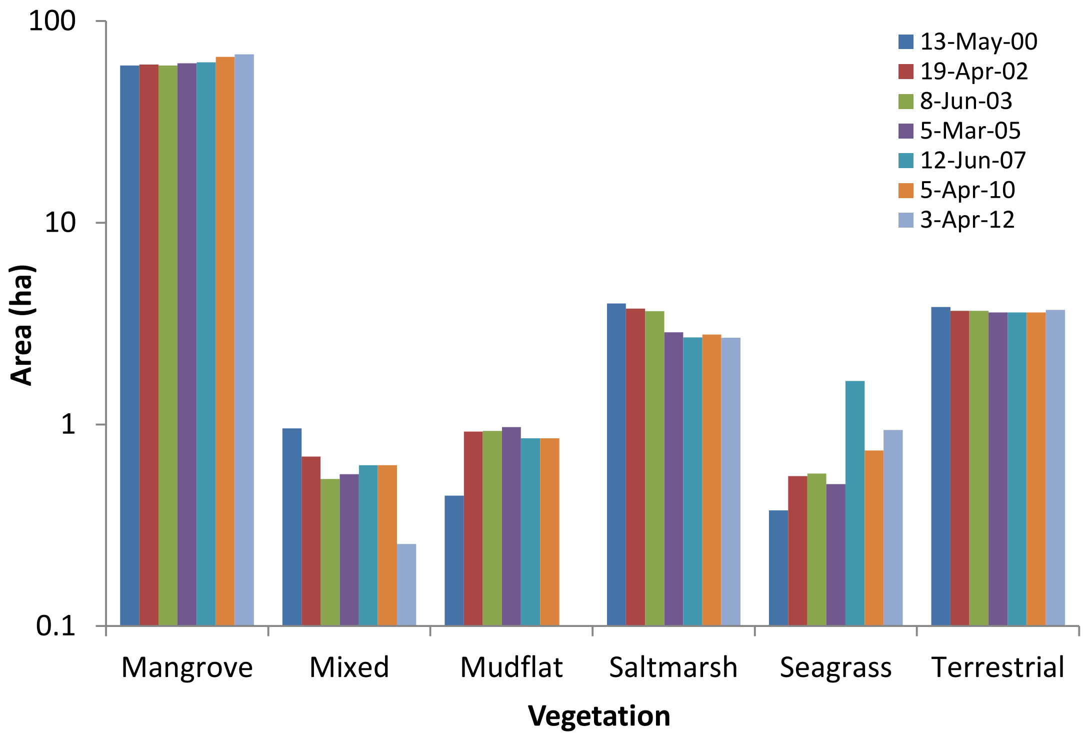

Figure 44: Change in area of estuarine vegetation within the Cobaki Broadwater.

Table 3: Area (ha) of estuarine vegetation within the Cobaki Broadwater between 13 May 2000 and 3 April 2012.

Vegetation	13-May-00	19-Apr-02	8-Jun-03	5-Mar-05	12-June-07	5-Apr-10	3-Apr-12
Mangrove	60.30	60.88	60.29	61.77	62.50	66.37	68.40
Mixed	0.96	0.69	0.54	0.57	0.63	0.63	0.26
Mudflat	0.44	0.92	0.93	0.97	0.85	0.85	
Saltmarsh	3.98	3.75	3.65	2.86	2.70	2.79	2.69
Seagrass	0.38	0.56	0.57	0.51	0.90	0.74	0.94
Terrestrial	3.83	3.66	3.66	3.59	3.59	3.59	3.70

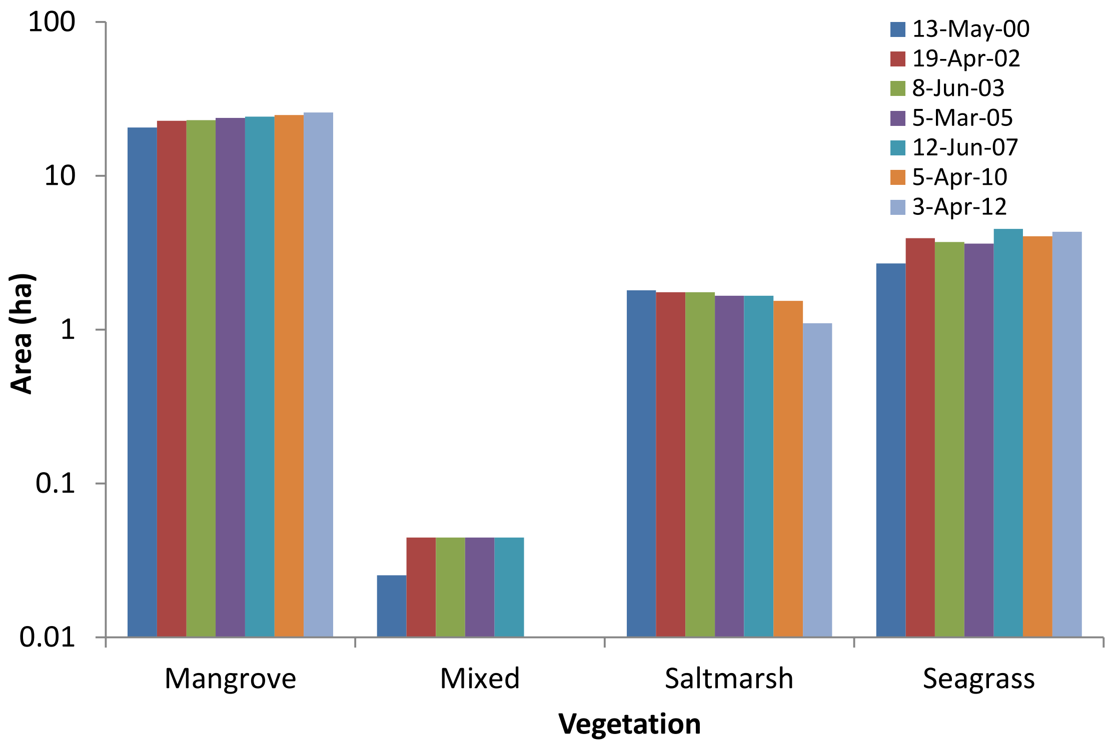

Figure 45: Change in area of estuarine vegetation within the Fluvial Channel.

Table 4: Area (ha) of estuarine vegetation within the Fluvial Channel between 13 May 2000 and 3 April 2012.

Vegetation	13-May-00	19-Apr-02	8-Jun-03	5-Mar-05	12-June-07	5-Apr-10	3-Apr-12
Mangrove	20.62	22.78	23.05	23.82	24.23	24.81	25.78
Mixed	0.03	0.04	0.04	0.04	0.04		
Saltmarsh	1.80	1.76	1.76	1.66	1.66	1.54	1.10
Seagrass	2.69	3.94	3.72	3.63	4.53	4.05	4.32

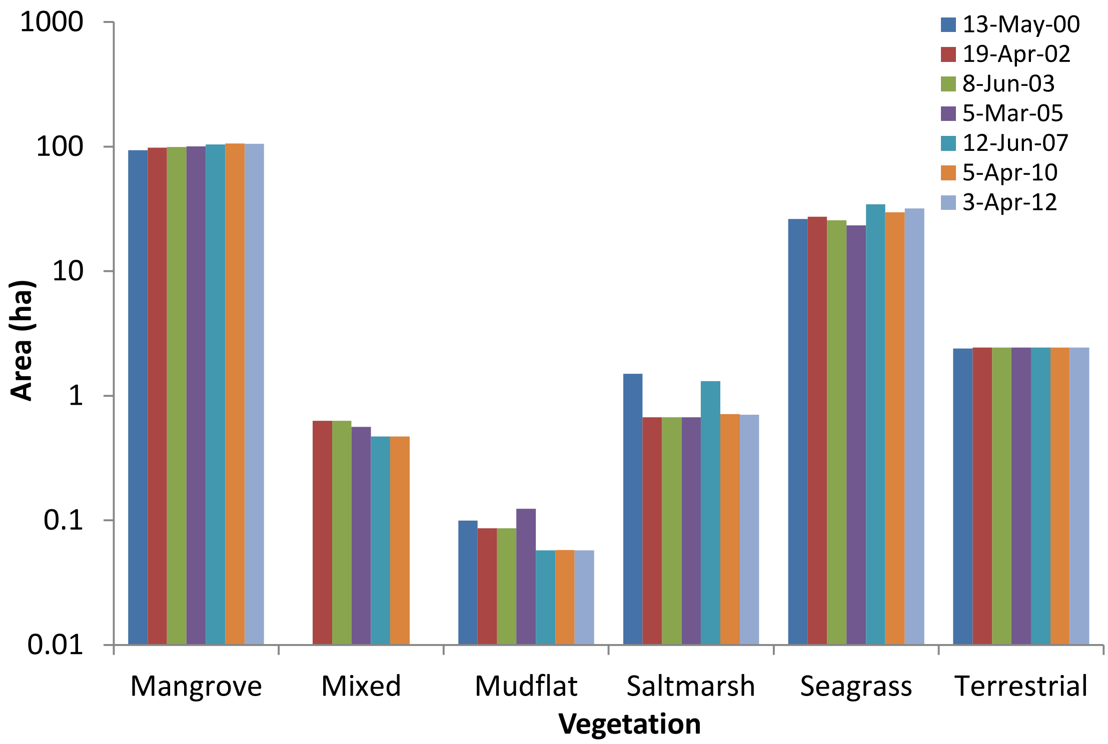

Figure 46: Change in area of estuarine vegetation within the Terranora Broadwater.

Table 5: Area (ha) of estuarine vegetation within the Terranora Broadwater between 13 May 2000 and 3 April 2012.

Vegetation	13-May-00	19-Apr-02	8-Jun-03	5-Mar-05	12-June-07	5-Apr-10	3-Apr-12
Mangrove	93.30	97.89	99.21	100.49	103.92	106.15	105.43
Mixed		0.63	0.63	0.56	0.47	0.47	
Mudflat	0.10	0.09	0.09	0.12	0.06	0.06	0.06
Saltmarsh	1.50	0.67	0.67	0.67	1.31	0.71	0.70
Seagrass	26.30	27.42	25.59	23.39	34.38	29.80	31.99
Terrestrial	2.39	2.44	2.44	2.44	2.44	2.44	2.44

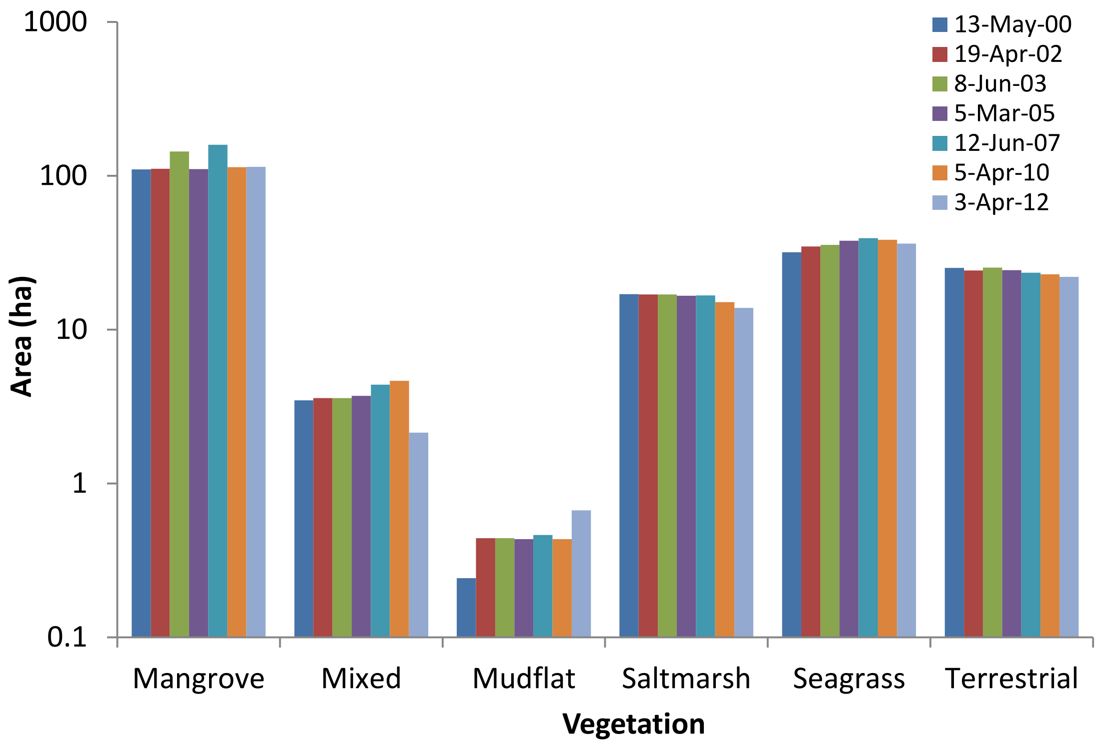

Figure 47: Change in area of estuarine vegetation within the Terranora Tidal Channel.

Table 6: Area (ha) of estuarine vegetation within the Terranora Tidal Channel between 13 May 2000 and 3 April 2012.

Vegetation	13-May-00	19-Apr-02	8-Jun-03	5-Mar-05	12-June-07	5-Apr-10	3-Apr-12
Mangrove	110.26	111.16	144.02	110.67	159.39	113.92	114.70
Mixed	3.48	3.59	3.59	3.71	4.39	4.65	2.14
Mudflat	0.24	0.44	0.44	0.43	0.46	0.43	0.67
Saltmarsh	17.04	16.90	16.90	16.62	16.71	15.14	13.87
Seagrass	31.80	34.78	35.64	37.84	39.32	38.45	36.24
Terrestrial	25.24	24.22	25.35	24.34	23.50	22.90	22.01

3.1.3 Mapping Errors

Some component of the estimated changes in the extent of mangrove, saltmarsh and seagrass will be due to variations in photograph quality between periods of capture, inherent errors within aerial photograph images, georectification error and digitising error.

Georectification is the process whereby the geometry of an image is made planimetric based on the co-ordinates on a map (Jensen, 1996). In doing so a root mean square (RMS) error is produced for each ground control point and is then averaged for each image. Based on the protocols of Wilton and Saintilan (2000) an RMS error of 5 metres or less was considered appropriate. Due to the topographic variability within the areas surrounding the broadwaters, it is expected that the error is greater in areas with a high elevation (>20m). Since estuarine vegetation is not located within high elevation areas, the need for correction within these regions was not necessary.

The Tweed Heads and Currumbin topographic maps were used as the base maps for georectification. Topographic maps are produced based on aerial photography and therefore produce an error. This error should be accounted for in the calculation of mapping error based on these maps. The Tweed Heads map is based on 2002 aerial photography and on the horizontal axis, "90% of well defined areas is within 12.5 metres of true position". The Currumbin topographic map is based on 1972 imagery and was revised in 1983. On the horizontal axis the map error is 10 m.

Human error is unavoidable in any manual procedure. To estimate the human error, each individual involved in the mapping process remapped a small section and the variability was determined based on differences in vegetation area. This variability is represented as a percentage error for each vegetation category in Table 7.

Table 7: Percentage errors associated with interpretation of vegetation polygons, by vegetation category, for the 2002 photo series.

Vegetation Type	Area 1	Area 2	Percentage Error
Mangrove	109996	111772	1.59
Saltmarsh	82361	83433	1.28
Seagrass	94693	92729	2.07
Terrestrial	25004	28138	11.14

3.2 Vegetation Sampling

The saltmarsh plain of Ukerebagh Island is dominated by *Sporobolus virginicus.* A number of *Avicennia marina* seedlings and juveniles have begun to encroach on the saltmarsh plain. The cover of saltmarsh showed significant differences over time ($p<0.001$, Figure 48), due largely to a decline in cover at the height of the drought in 2001, and a more extensive decline between the 2008 and 2012 surveys. The increased density of *Avicennia marina* individuals in saltmarsh plots does not explain the decline, in that the decline in vigour of *S virginicus* predates encroachment of mangrove. This is unusual in that within NSW mangrove encroachment is normally the cause of saltmarsh decline (Saintilan and Williams 1999). The period of decline does correspond to a period of erosion identified by the feldspar marker horizons. Further monitoring is recommended to determine whether *S. virginicus* recovers from this decline, though this seems unlikely given the large bare areas now resulting (Figures 51 and 52).

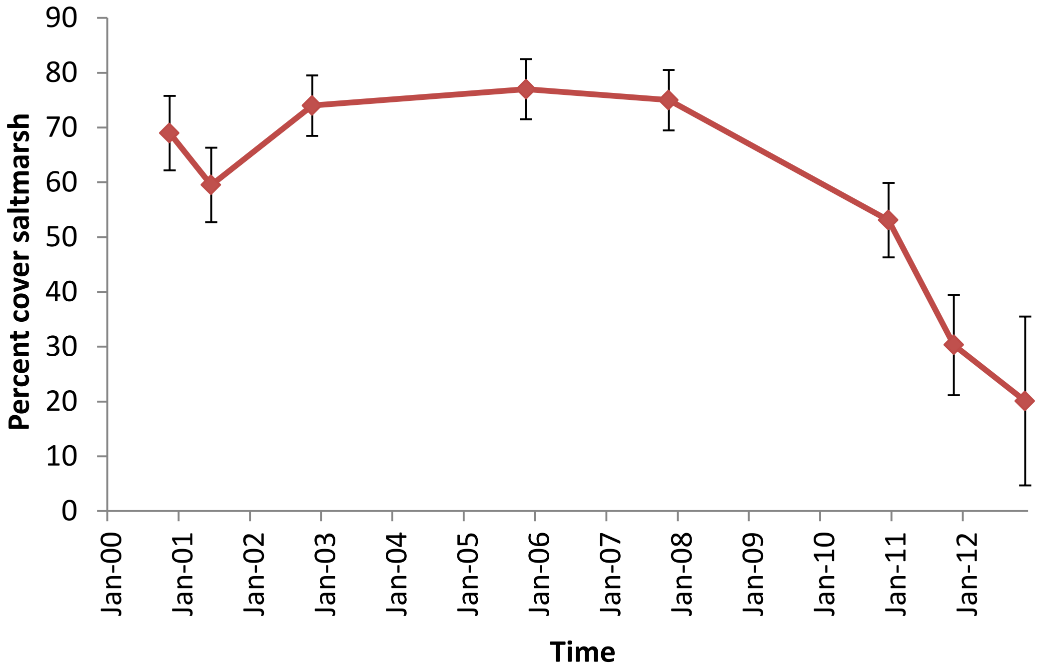

Figure 48: Changes in cover (%) of *Sporobolus virginicus* in saltmarsh plots at Ukerebagh Island.

3.3 Surface Elevation

Surface elevation increased in the mangrove zone over the study period by 18.94 mm and increased at a mean rate of 1.59 mm y^{-1} (Figure 49). The mangrove trajectory was characterised by decreases in surface elevation in the periods of October 2003 to January 2006, and September 2007 to June 2010. There was a marked increase in surface elevation between June 2010 and October 2012. A similar trend was evident within the saltmarsh zone with surface elevation declining in the periods of October 2003 to September 2007 and August 2009 to June 2010, though a decline in surface elevation was evident between June 2010 and October 2012. Surface elevation increased in the saltmarsh zone over the study period by 0.79 mm and increased at a mean rate of 0.07 mm y^{-1}.

Surface elevation change did varied significantly between zones, with mangrove elevation increase higher than saltmarsh, and varied significantly over time ($p<0.01$) with significant time and zone interactions evident ($p<0.001$). This variation over time was particularly driven by changes in the mangrove zone, rather than the saltmarsh zone.

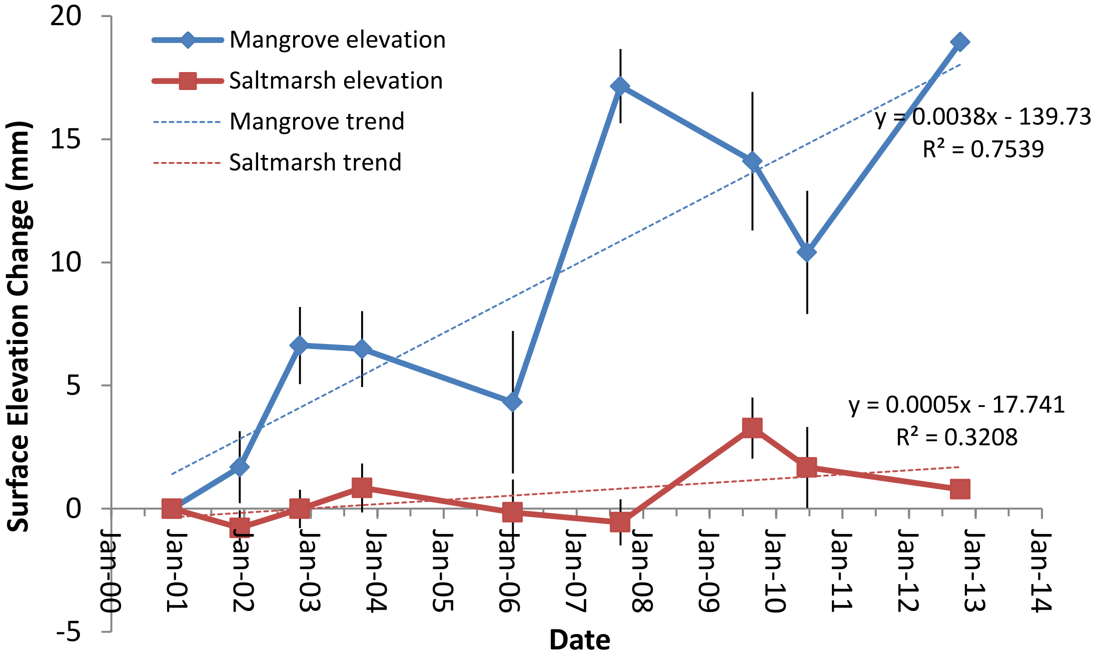

Figure 49: Surface elevation change in the mangrove and saltmarsh zones between 2000 and 2012.

3.4 *Vertical Accretion*

Feldspar marker horizons could not be located in October 2012; consequently estimates of rates of accretion since October 2010 were not generated.

In the first two years of the survey, mangrove accretion exceeded saltmarsh accretion, and the rates of accretion in both environments accounted for the surface elevation trends. There was a departure from this trend following 2004, with significant accretion in the saltmarsh zone, and more modest rates of accretion in the mangrove zone (Figure 50). As a result, accretion in the saltmarsh zone was no longer significantly different to mangrove by October 2007. Between October 2007 and June 2010, accretion returned to the norm established in 2000 with greater accretion occurring in the mangrove zone than the saltmarsh zone. The mangrove zone accreted at a mean rate of 2.04 mm y^{-1} over the study period while saltmarsh accreted at a mean rate of 1.24 mm y^{-1}.

Rates of vertical accretion did vary significantly between zones over the study period ($p=0.0286$) and was generally driven by higher rates of accretion in the mangrove zone than the saltmarsh zone.

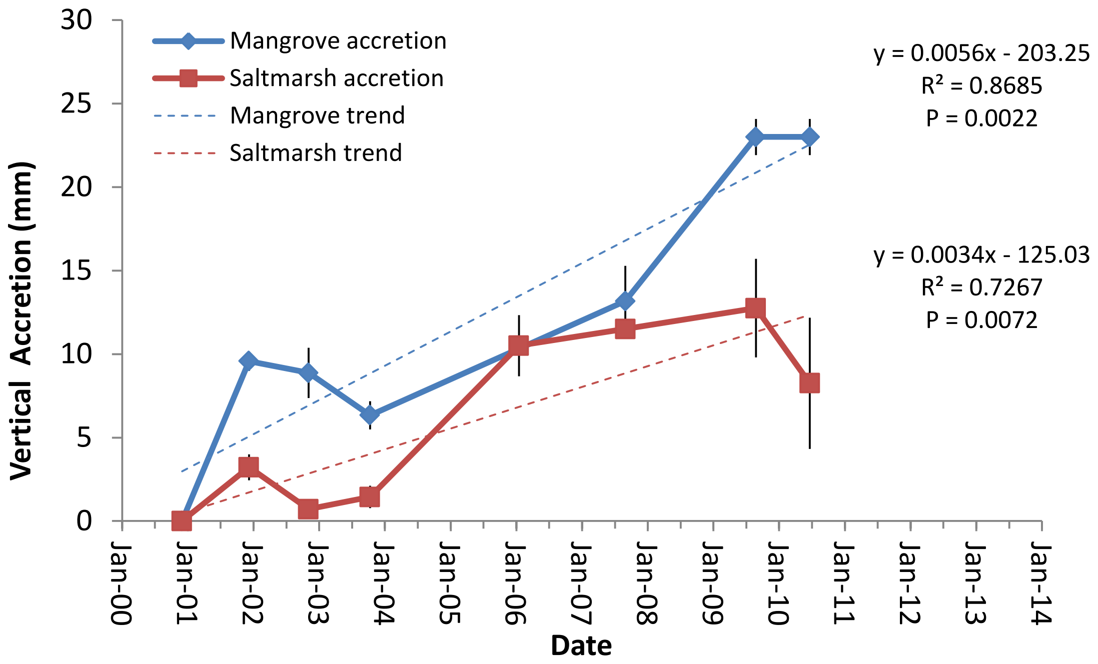

Figure 50: Change in sedimentation in the mangrove and saltmarsh zones between 2000 and 2010.

Measures of surface elevation and vertical accretion between October 2000 and June 2010 were not significantly different (p=0.1024), a trend that was consistent in both the mangrove zone (p=0.1436) and the saltmarsh zone (p=0.1580). The overall trend was for surface elevation change to be driven by accretion.

4 Discussion

4.1 Trends in Wetland Vegetation 1948- 1998- the historical context

The proliferation of mangroves in the estuaries of southeastern Australia since the time of European settlement over 200 years ago is now a well established trend (McLoughlin 2000, Saintilan and Williams 2000, Saintilan 2003). Historical records point to a continuous increase in mangroves along estuarine shorelines (McLoughlin 2000) and historical aerial photographs extending back to the early 1950’s show the landward encroachment of mangroves onto saltmarsh plains in numerous estuaries (Saintilan and Williams 1999). Many hypotheses have been advanced explaining this trend, including elevated sea level (Wilton 1997, Saintilan and Hashimoto 1999, Rogers et al. 2006), marsh subsidence (Burton 1982, Vanderzee 1988, Saintilan 1998), increased sedimentation rates (Wilton 1997, McLoughlin 2000) and elevated nutrient levels (McLoughlin 1987, Wilton 1997, McLoughlin 2000). A network of monitoring sites is established in 11 locations throughout southeastern Australia to examine vegetation change in the context of the above hypotheses.

One estuary particularly affected is the Tweed River (Figure 1). The area of both mangrove and saltmarsh increased in the period 1939-1994 (Saintilan 1997). The increase in saltmarsh area in this period is atypical of estuaries in New South Wales, which are characterised by saltmarsh decline (Saintilan and Williams 1999). The increase in saltmarsh area was not significant in the final decade of this survey

(1984-1994). A more recent survey (Wilton 2002) has demonstrated a decline in saltmarsh on Ukerebagh Island of 16% in the period 1948-1998, and this rate of decline corresponds to the rate of saltmarsh decline documented for the entire estuary in the survey of Rogers et al. (2003).

A study was implemented in November 2000 in response to the proliferation of mangroves in estuaries in southeast Australia (Rogers, Saintilan and Wilton 2002, Rogers et al. 2006). Detailed photogrammetric mapping in this report and the study of *Avicennia marina* community structure at the landward mangrove boundary indicate a trend of mangrove colonisation of saltmarsh. Saltmarsh area declined from 13.8ha in 1961 to 11.1ha in 1998 (20% decline), while mangrove area increased from 37.7ha to 45.2ha (20% increase) over the same period. Juveniles were observed spreading out over the saltmarsh plain in areas perceived by Saintilan (1998) to be relatively stable.

4.2 Vegetation trends in the period 2000-2010

The extent of mangrove within the estuary increased by 33.29 hectares over the study period, a rate of approximately 1% per year. Over the same period, the area of saltmarsh decreased by 6.43 hectares, a rate of decline of 2.18 % per year. The decline of saltmarsh was primarily due to mangrove proliferation in the saltmarsh zone, though there appeared to be some conversion of saltmarsh to unvegetated mudflats on Ukerebagh Island (Figure 51 and Figure 52). The trend of mangrove increase and saltmarsh decline occurred in all segments of the estuary.

Figure 51: Extensive mudflats developing on the saltmarsh at Ukerebagh Island.

Figure 52: Unvegetated mudflats developing at lower elevations on saltmarsh plains of Ukerebagh Island, These mudflats are largely unvegetated and were thick saltmarsh prior to 2010.

These rates of change are consistent with the historical trends described above, as well as changes in the extent of similar communities in estuaries within New South Wales and Victoria (Saintilan and Williams 2000). However, the conversion of saltmarsh to mudflats on Ukerebagh Island is inconsistent with trends observed throughout southeastern Australia, where mangrove tend to rapidly colonise saltmarsh plains and out-compete saltmarsh for resources, such as light. The loss of saltmarsh and the conversion to unvegetated mudflat is an atypical trend for southeastern Australia and further investigation is warranted.

Seagrass showed a consistent increase in the period 2000 to 2007, in all geomorphic settings, declined slightly to 2010 and has since stabilised, showing a slight increase in the present survey. The overall increase in seagrass of 20.08 % is significantly higher than can be attributed to georectification and digitising errors. Factors which promote the increase of seagrass area include water clarity and salinity (which may be associated with drought conditions), and geomorphic stability. There is no indication that the small change in tidal hydraulic conditions within the Tweed estuary is having a detrimental impact on seagrass beds. These rates of increase in seagrass extent are similar to those reported by Hossain (2005) for the Ukerebagh Channel over the period of 1997 to 2001 (Table 7). The year 2000 was mapped twice to determine mapping error.

Table 8: Area of seagrass vegetation in Ukerebagh Channel, Tweed River. (Source: Hossain 2005).

Time	Area (ha)
1997	7.04
1998	7.47
1999	7.05
2000 (a)	7.90
2000 (b)	8.12
2000 (mean)	8.01
2001	8.95

Minor flooding of the estuary in June 2005, January 2008 and January 2012 did not adversely affect seagrass extent. Indeed, the greatest period of increase in seagrass extent was the period 2006-2007. During this period the extent of seagrass increased from 7.5% above the 2000 benchmark to 21.1% above the 2000 benchmark, in spite of local losses in the vicinity of boat-ramps.

4.3 Vegetation quadrats

The percentage cover of the dominant saltmarsh plant *Sporobolus virginicus* declined in December 2001, possibly due to the onset of drought (Figure 48). By November 2002 there has been considerable recovery, with groundcover exceeding the 2000 levels. This increase in cover continued to October 2007. Since 2007 there has been another more substantial and sustained episode of decreasing cover, with *S. virginicus* declining to 20 percent cover across 24 quadrats. The thinning trend was noticeable across Ukerebagh Island, and cannot be attributed in this case to drought. Erosion of the same saltmarsh was detected by the feldspar marker horizons and may be related to the thinning trend, though further monitoring would be required to substantiate this suggestion. There was indication of increased mangrove encroachment in the permanent saltmarsh vegetation plots in 2012, though not prior to this. The decline of *S. virginicus* cover is of ecological significance given the demonstrated importance of this species as a source of nutrition for estuarine crustaceans (Saintilan and Mazumder 2010).

4.4 Surface processes

The primary process controlling surface elevation change within the mangrove and saltmarsh communities of the lower Tweed River estuary is accretion. The degree of surface elevation change was greater in the mangrove zone than the saltmarsh zone; a trend that is consistent with models of accretion being proportional to inundation frequency (Pethick 1981, van Winjen and Bakker 2001) and findings at study sites in southeastern Australia (Rogers 2004, Rogers *et al.* 2005).

However, accretion and corresponding surface elevation change are lagging behind water level changes within the estuary. Water levels at the Letitia 2A tidal gauge increased at a mean rate of 4.6 mm y^{-1} between 1997 and 2012 (Figure 53), while mangrove and saltmarsh surface elevations increased by 1.39 mm y^{-1} and 0.18 mm y^{-1}, respectively. The increase in mean water level within the estuary appears to be largely driven by increases in the mean high water spring tide, estimated to be increasing at 6.2 mm y^{-1}.

Based on current rates of surface elevation change and water level change within the estuary, the mangrove and saltmarsh communities of the lower Tweed River estuary may be vulnerable to submergence from rising water levels. Continued analysis of surface elevation change and vertical accretion in relation to sea-level rise and water level changes within the estuary will enhance our capacity to forecast the vulnerability of mangrove and saltmarsh vegetation in the lower Tweed River estuary to submergence. Supplementing this information with analyses of longer-term sedimentation rates using radiometric dating techniques would provide valuable information regarding changes in rates of sediment delivery pre- and post 2000.

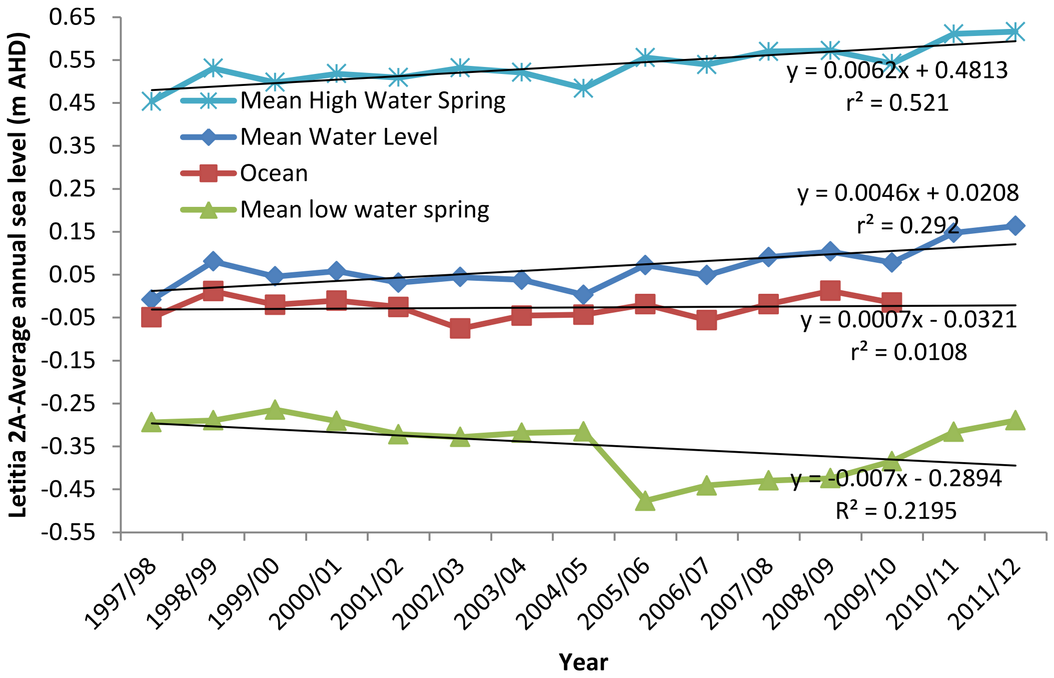

Figure 53: Annualised mean high water spring tide and mean water level at the Letitia 2A gauge within the Tweed River estuary and mean ocean water level from 1997 to 2012.

4.5 Tidal changes

4.5.1 Predicted changes

The TRESBP Environmental Impact Statement/Impact Assessment Study (EIS/IAS) predicts very small tidal changes directly attributed to entrance improvements as a result of the operation of the Tweed River Entrance Sand Bypassing Project (Hyder 1997). The EIS/IAS predicts that a deeper river entrance condition would result in a slight increase in the spring tidal range of about 5 cm at the Letitia 2A tide gauge near the confluence of the Tweed River and Terranora Inlet (Figure 54). This predicted change would reduce with distance upstream of the Letitia 2A gauge and be less for non-spring tides.

Figure 54: Location of the Letitia 2A tide gauge and lower estuary shoals.

The lower estuary marine shoals (Figure 48) are a control on the propagation of tides into the lower Tweed River. These shoals have gradually infilled and reduced tidal exchanges, which have adversely impacted on tidal flushing and water quality since the mid 1970's. The EIS/IAS predicts that the operation of the sand bypassing system will reduce the net infeed of marine sand into the river estuary. This has the beneficial impact of reducing the growth of the lower estuary shoals and lessening their adverse impacts on tidal flushing and water quality.

Were it to occur, severe scouring of the lower estuary shoals by a major flood would lead to an increase in the tidal range of the lower Tweed River estuary. The increased tidal range would improve tidal mixing and flushing of the estuary, but would also impact on estuarine ecology. The reduction of the net infeed of marine sand into the river by the operation of the TRESBP could, under these circumstances, increase the period taken for post-flood recovery of the eroded shoals (Hyder 1997). Thus, the project has the potential to have both beneficial and detrimental impacts on the lower Tweed River estuary.

4.5.2 Monitored changes

The NSW Department of Services, Technology and Administration (formerly Public Works), Manly Hydraulics Laboratory (MHL) has monitored tides within the Tweed River from 1971 to present. MHL analysed and reported on long-term tidal changes from analyses carried out for 1971, 1980 and yearly analyses from 1988 to 1997, on behalf of the Tweed River Entrance Sand Bypassing Project. In addition, the NSW

Department of Environment, Climate Change and water (DECCW) have undertaken ongoing tidal analyses from 1997 to the present on behalf of the TRESBP using data from the Letitia 2A tidal gauge, which is located 1.5 km upstream from the Tweed River entrance.

To date monitoring of the tidal data (Table 9) shows that there has been no significant change in tidal conditions that can be attributed to the operation of the sand bypassing system. The annual spring tidal ranges at Letitia 2A for June 2001 to June 2010 have not exceeded the EIS/IAS predictions (Floyd 2001, 2002, 2003, 2004, 2005, 2006a and 2006b). The observed tidal range has slightly reduced since 2007, with conditions currently approximating pre-TRESBP conditions.

Table 9: Annual tidal analysis for the Letitia 2A tide gauge

	Letitia 2A Annual Spring Tidal Range (m)	Letitia 2A/Average Ocean Spring Tidal Range Ratio
EIS/IAS Pre-existing Conditions (mean 1989-92)	**0.92**	**0.67**
EIA/IAS Predicted Change	**0.97**	**0.71**
Monitoring Results		
2000/01	0.92	0.65
2001/02	0.96	0.70
2002/03	0.97	0.71
2003/04	0.96	0.70
2004/05	0.97	0.70
2005/06	0.97	0.71
2006/07	0.98	0.71
2007/08	0.96	0.68
2008/09	0.94	0.67
2009/10	0.92	0.67
2010/11	0.93	
2011/12	0.91	

There have been no major floods in the Tweed River since the preparation of the EIS/IAS and no significant unexpected changes to date in the Tweed River lower estuary shoals. There have been minor net losses since 1994, but sand volumes currently approximate the EIS/IAS baseline conditions that were surveyed in 1989/1990.

4.6 Recommendations concerning monitoring frequency

In determining an appropriate monitoring frequency, consideration was given to the wetland monitoring program for wetland distribution identified in Section 8.5.1.5 of the TRESBP Stage 2 EIS/IAS and the relative influences of long term impacts associated with the project (i.e., tidal changes) compared to other influencing factors outside the control of the project, such as climatic changes (example droughts and floods). Trends in mangrove and saltmarsh aerial extent continue to be incremental and linear between surveys. For this reason, it is suggested that a four-yearly survey of mangrove and saltmarsh distributions is an appropriate frequency for system-wide vegetation mapping. An adequate period of assessment for the coming decade, therefore, would be 2014 and 2018, with the frequency of survey revisited each time. However, it would be advisable for aerial photography to be captured biennially, in 2014, 2016, 2018 and 2020, to provide greater flexibility in assessing the impacts of discrete events, such as specific floods and storms.

Seagrass distribution, on the other hand, was found to be more variable. Williams and Meehan (2004) working in the Port Hacking estuary also noted a more variable seagrass area than mangrove and saltmarsh, and suggested that seagrass distribution was sensitive to storm and flood events. It is suggested that the seagrass communities be monitored at the same time as the mangrove and saltmarsh communities and that additional surveys might follow significant storms and/or floods.

We have concerns over the declining cover of *Sporobolus virginicus*, (known as Salt Couch Grass), on Ukerebagh Island that was identified in the two most recent surveys (2011, conducted voluntarily by the consultants, and 2012). It appears that *S. virginicus* has crossed a survival threshold on Ukerebagh and that recovery is unlikely. Sediment erosion was detected over the same period and the Surface Elevation Tables are suggesting that the saltmarsh elevation is lagging behind water level trends in the estuary or may be responding to more recent slight reductions in tidal range. For these reasons a return survey of the SET, feldspar and vegetation plots in the saltmarsh on Ukerebagh Island is recommended for 2014. We also suggest that long-term sedimentation rates in the saltmarsh on Ukerebagh Island be assessed using radiometric techniques (such as 210Pb). This would help assess whether contemporary rates of sedimentation in the saltmarsh (2000-2010) measured by this study are a departure from historic trends, as might be the case if sediment source has been interrupted.

Under the project Environmental Management System-Operations, the ongoing monitoring of tide and shoal conditions, particularly following a major flood event, require additional shoal surveys to be undertaken if the operation of the system has a significant impact on the condition or post-flood recovery of the shoals and tides. It is not anticipated that current rates of change will depart from their established trajectories under current operating conditions and in the absence of significant additional drivers, such as flooding, major storm events, or accelerated sea-level rise. Unless monitoring identifies that the operation of the system has had a significant impact on the river tides, there does not appear to be a need to sustain the current mapping frequency to manage the potential impacts on the mangrove, saltmarsh and seagrass communities.

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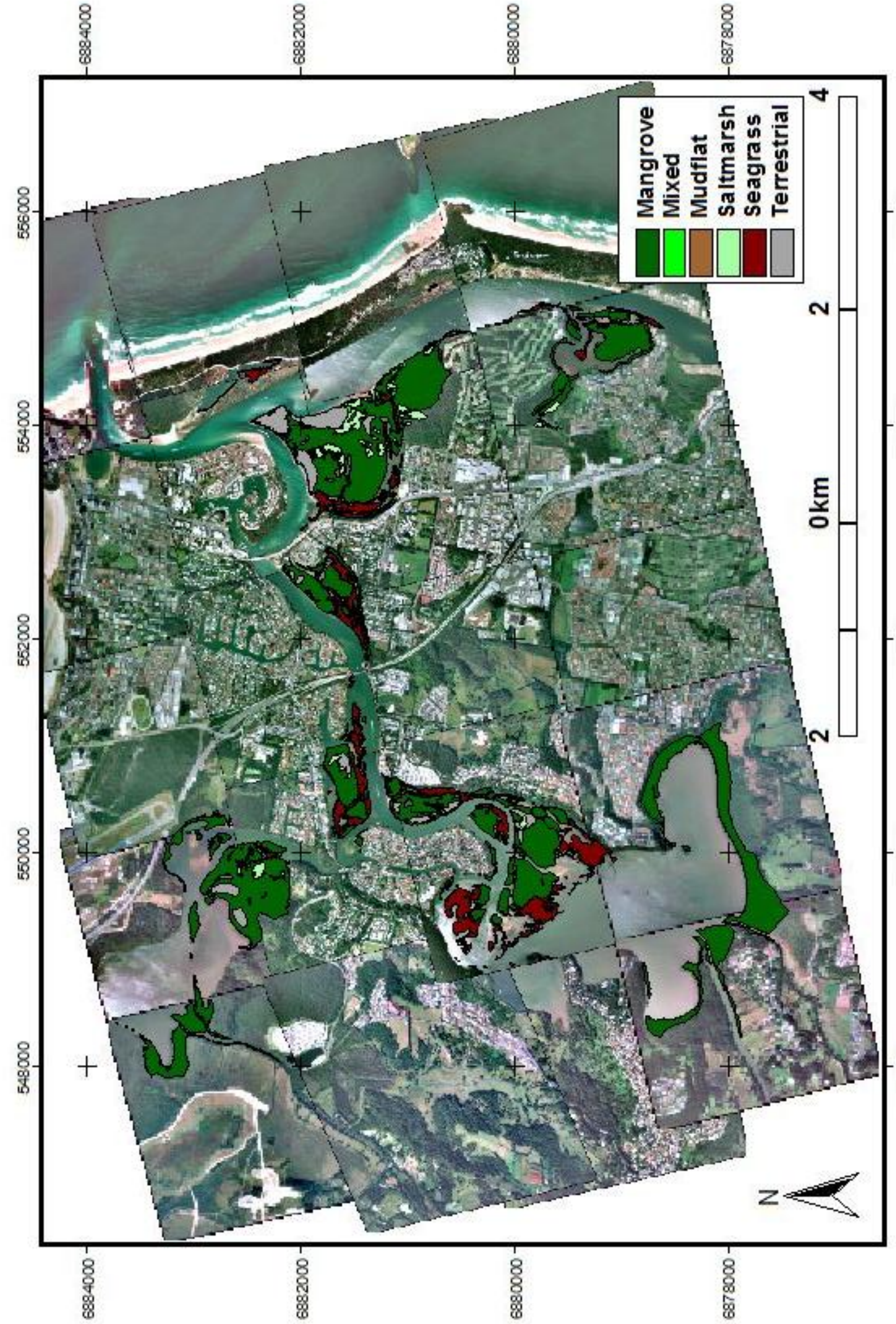
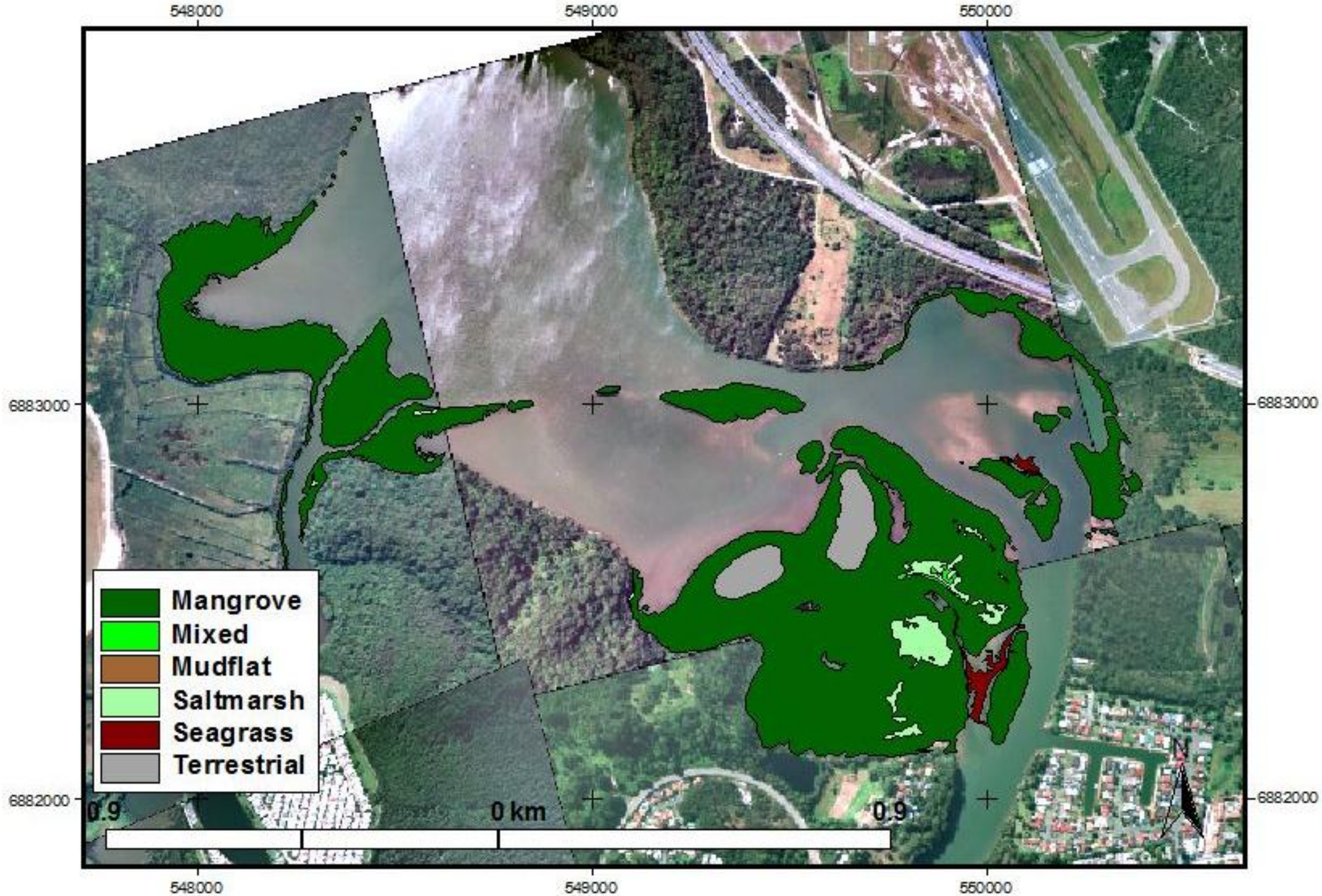
7 Appendix 1: Estuarine Vegetation Maps

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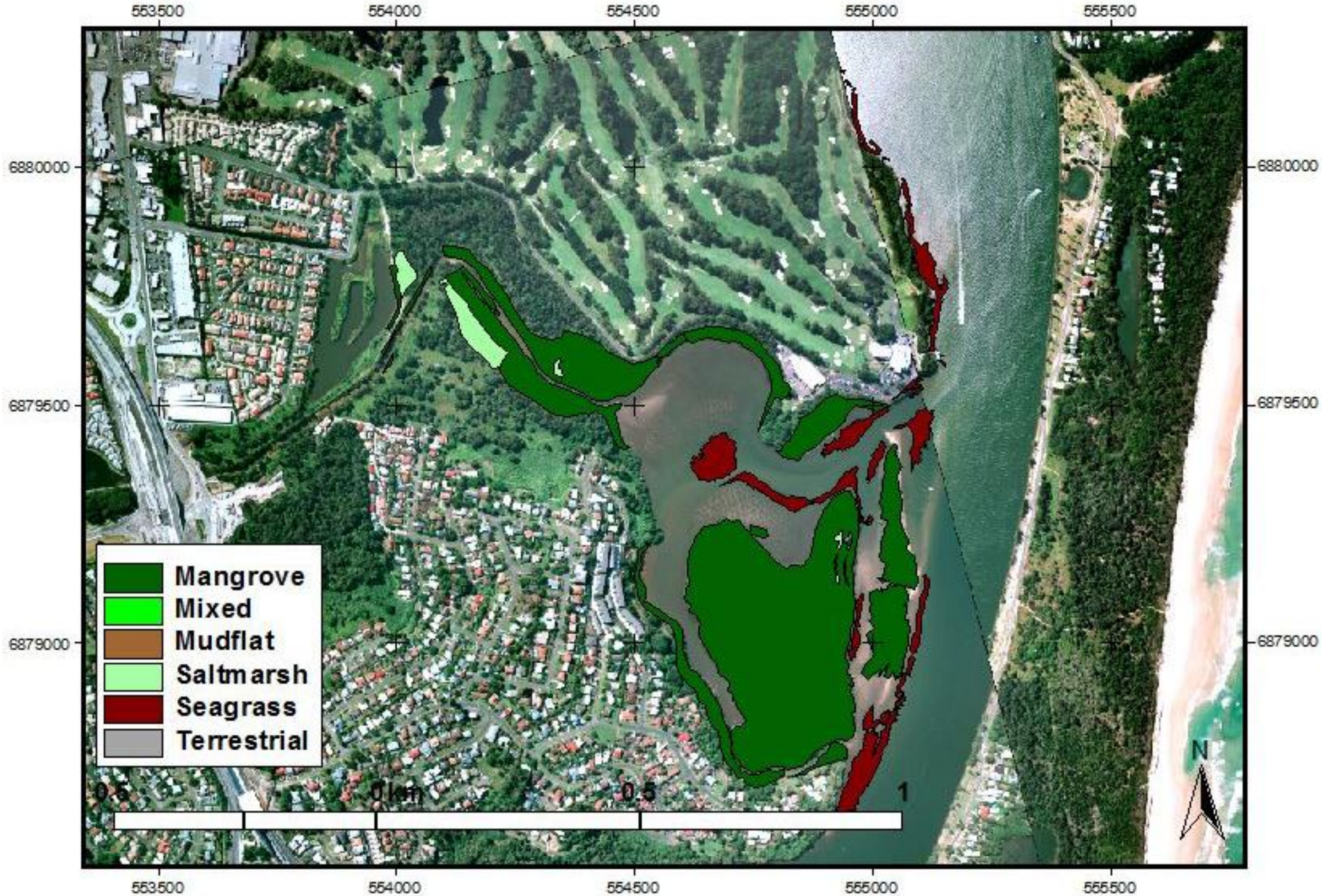
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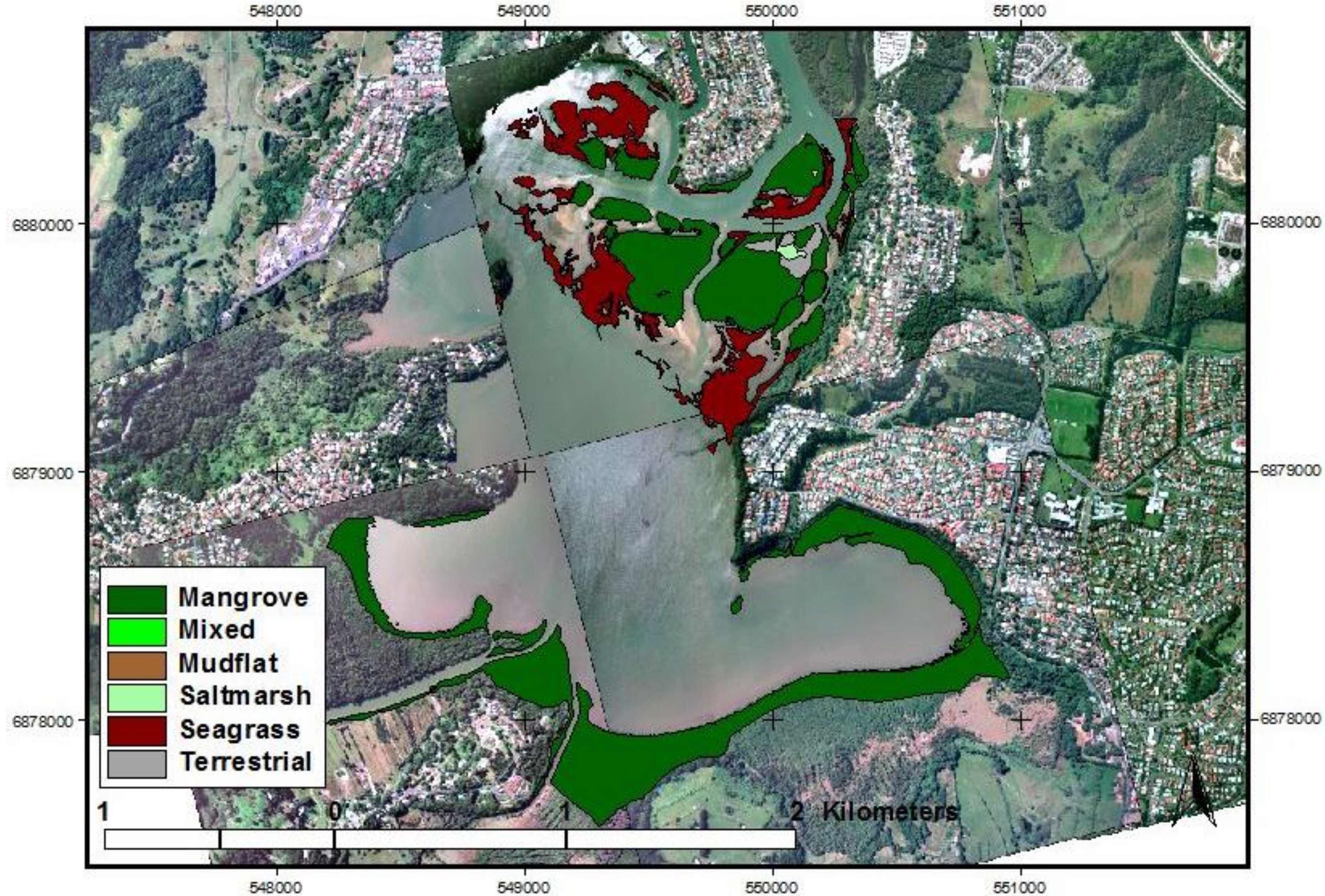
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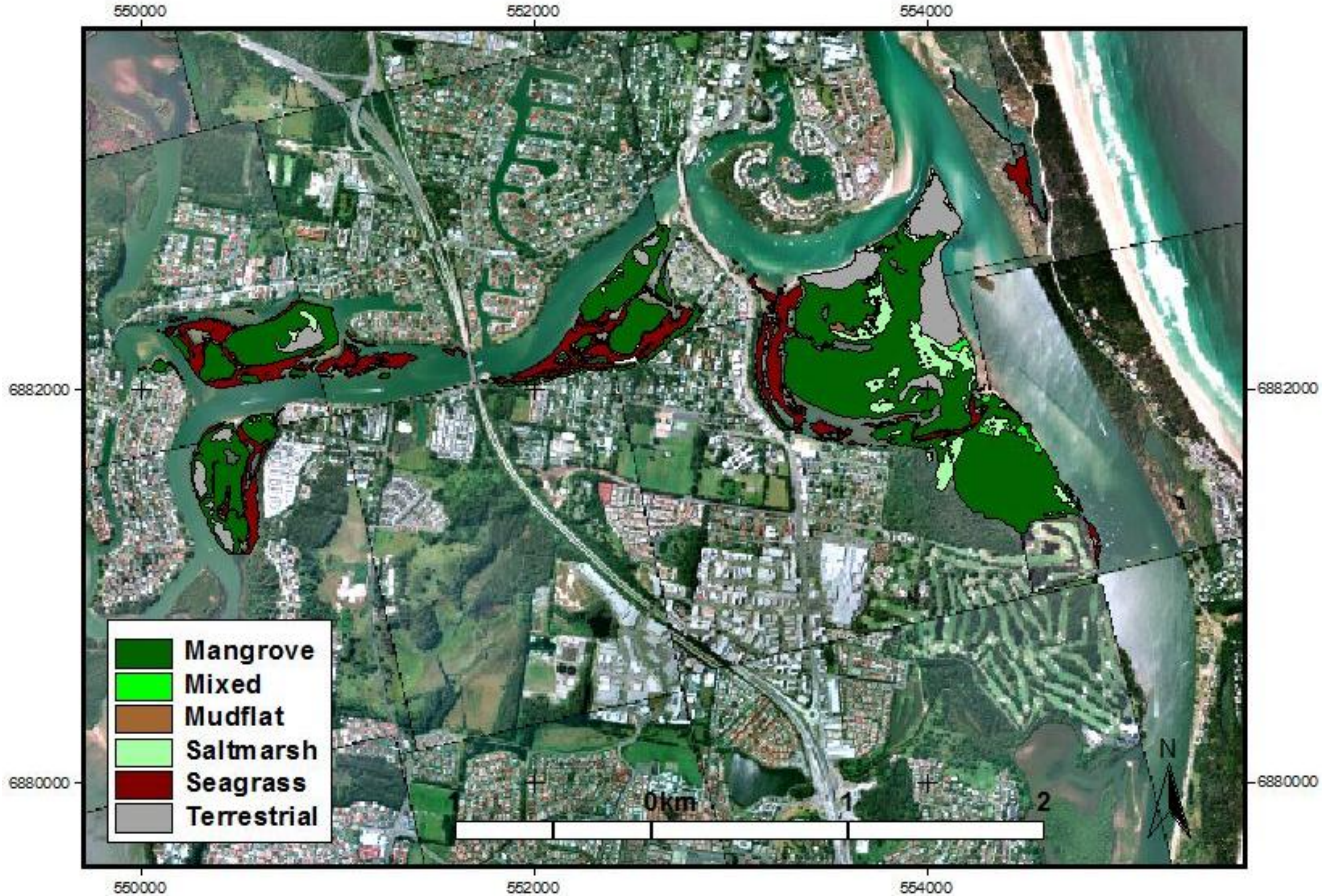
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